Recreational Fishing Initiatives Fund Final Report

Can recreational fishers provide a cost effective means for monitoring artificial reefs?

James R. Tweedley[1], James H. Florisson[1, 2], Thomas A. Bateman[1], Jennifer A. Chaplin[1] & Howard S. Gill[1]

[1] School of Veterinary and Life Sciences, Murdoch University, 90 South St. Murdoch, Western Australia 6150

[2] Recfishwest, Suite 2, 45 Northside Drive, Hillarys Western Australia 6025

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Facing towards   **Facing away**

Pentaceropsis recurvirostris
(Longsnout Boarfish)

Trygonorrhina fasciata
(Southern Fiddler Ray)

Glaucosoma hebraicum
(Western Australian Dhufish)

Anoplocapros amygdaloides
(Western Smooth Boxfish)

Lagocephalus lunaris (Rough Golden Toadfish)
& *Pseudocaranx georgianus* (Silver Trevally)

Seriola hippos (Samson Fish)

Photographs of species from different ecological groups observed in this study using baited remote underwater video (BRUV) systems facing towards and away from the artificial reef modules.

Extract from a BRUV video on the Bunbury artificial reef.

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Non-technical summary of research findings

Artificial reefs have been constructed and deployed in over 50 countries around the world to enhance the productivity of aquatic habitats and fishing experiences. In April 2013, two purpose-built concrete artificial reefs were deployed in Geographe Bay, Western Australia to provide additional fish habitat and increase upwelling and thus enhance recreational fishing opportunities. Due to the relatively high cost of planning, purchasing and deploying these structures, it is important to understand spatial and temporal usage of the reef by fish assemblages, in order to determine the extent to which fishing opportunities are actually enhanced. One potential method to reduce monitoring costs is to utilise volunteers from the general public to collect data, *i.e.* citizen science. The overall objective of this project was to determine whether recreational fishers, through a citizen science program, could potentially provide an effective means for monitoring artificial reefs.

Following a recruitment drive and underwater camera trial, a small number of recreational fishers were provided underwater drop video cameras and asked to record footage on the Bunbury and Dunsborough artificial reefs and also nearby natural reefs. Unfortunately, only very limited amounts of data (~1 hour) were received due to a combination of a lack of participation/engagement, unseasonal weather and the short timeframe of the project. However, enough videos were received to undertake a preliminary analysis of the differences in the characteristics of the fish faunas of the two types of reef, *i.e.* natural *vs* artificial. The results indicate that artificial reefs may potentially harbour greater numbers of species and a larger total maximum abundance. Multivariate statistical analyses did not detect any differences in the fish faunal compositions between natural and artificial reefs, which were likely due to the dominance of King Wrasse (*Coris auricularis*) on both reefs. Furthermore, large amount of variability between replicates caused by the differences in recording time, which, although standardised, was still an artefact in the resultant data and may have masked any 'real' differences among the reef types.

Given the limited data provided by the above monitoring program, a critical review of the citizen science components of the project was completed and a set of key recommendations for use in future projects using recreational fishers to collect video footage provided. These

included: (i) enhancing the methods of contacting and recruiting volunteers to include social media and encouraging communication among participants, (ii) using a GoPro camera mounted on Baited Remote Underwater Video (BRUV) systems to ensure that the footage collected is of high quality (resolution), (iii) providing simplified and consistent instructions and (iv) ensuring regular communication and engagement with volunteers. A global literature review on citizen science and the benefits and limitations using this type of project for research purposes was also undertaken together with a brief description of such programs that have been or are being conducted in aquatic environments in WA.

To test the suggestions the BRUV systems with a GoPro camera constructed from readily available materials could be deployed by recreational fishers as a citizen science artificial reef monitoring tool, Ecotone Consulting built one of these units and deployed it randomly around the Dunsborough artificial reef. The resultant footage was found to be of much higher quality than that obtained using the drop camera. The GoPro videos were analysed to determine whether there was a difference in fish assemblages between artificial reef modules and the surrounding area, *i.e.* videos where the camera directly faced one or more of the artificial reef modules were vs those were no modules could be observed in the camera's field of view. The results demonstrated that mean number of species and the mean number of benthic and epibenthic species were greater on footage recorded when the camera faced the modules. There was also a difference in the faunal composition, with 52.63% more recreational target species being found on artificial reefs than in surrounding areas. It was also concluded that the BRUV technology could be used, by citizen scientists, to monitor the fish faunas of artificial reefs.

Another potential method to reduce the cost of monitoring programs for the fish faunas of artificial reefs is to use citizens to analyse the footage as part of their studies. However, if such a program was to proceed using volunteers with limited experience, *i.e.* undergraduate students, it is important to ensure that the fish data extracted from the video is reliable. Thus, to investigate the impact of observer bias, the BRUV footage collected from the Dunsborough artificial reef was analysed by having multiple observers, with similar levels of marine science training and recreational fishing experience. It was found that whilst

observers recorded similar mean numbers of species and total abundance counts, significant differences in species composition were detected. This was due to observers misidentifying members of particular families, *i.e.* the leatherjackets and trevallies. This suggests that, while observers with limited experience may be able to detect common species, misidentification of less common and/or less distinct species can lead to significant variation in the data due to observer bias. Therefore, if university students are to be used as part of any citizen science monitoring project, it is recommended that participants should receive additional training in species identification, and be subjected to an initial trial where their results are compared to that of a more experienced observer until a minimum similarity of 90% is consistently recorded.

The provision of BRUV footage from the Bunbury artificial reef by Ecotone Consulting allowed statistical analysis of fish faunas on both reefs to be undertaken to identify what level of information could be obtained using the BRUVs. Analysis of the data found both the mean number of species and total abundance were greater at the Dunsborough artificial reef and that there was also a significant difference in species composition. While, more data are required to provide a more accurate picture of any differences, this does highlight the fact that the BRUV footage can be employed to test for differences in the fish faunas of these reefs and possibly also nearby natural reef or other areas.

It is concluded that recreational fishers did not provide an effective means for monitoring artificial reefs during this project. This result, however, is a consequence of a lack of data stemming from an absence of volunteer engagement in a limited pilot project with a short time frame and unseasonal weather. This does not exclude the potential for using citizen scientists to monitor artificial reefs, following some changes in the methodology, technology and management of citizen science protocols, and thus it is possible to utilise recreational fishers as an effective means for monitoring artificial reefs. This project was subjected to restrictive and limiting factors but more importantly, discovered ways to overcome these issues by provided key recommendations on technology, methodologies and community

engagement that should be followed to increase the effectiveness of using recreational fishers to provide sound scientific information in the future and these have been actively employed in a new citizen science program for monitoring the fish fauna of the Bunbury and Dunsborough artificial reefs Reef Vision.

Section 1: Background Information and Overview of Project

Habitat enhancement structures

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

Artificial reefs are one of the most commonly deployed types of HES and have been deployed in more than 50 countries around the globe (Diplock, 2010). These structures vary greatly in type, structure, purpose and ecological function. Artificial reefs can be divided into two main types based on the materials used in their construction, namely 'reefs constructed from materials of opportunity' and 'purpose-built reefs'. Materials of opportunity are pre-existing materials such as concrete blocks and rubble, stones, polyvinyl pipe, tyres, derelict ships, car bodies, oil extraction equipment (such as disused oil rigs) and disused military equipment and vehicles, which are deployed to form the reef (Fig. 1; Sherman *et al*., 2002). Purpose-built artificial reefs are those constructed from particular material and designed specifically for target species/fauna or to mimic particular habitats or create environmental effects, such as upwelling (Department of Fisheries, 2012a; Haejoo, 2015). Purpose-built artificial reefs can be built from metal framework, steel, steel-reinforced concrete or

concrete. Examples include species-specific reefs, such as abalone habitat reefs, larger Offshore Artificial Reefs (OAR) such as the Sydney OAR, a 12 m tall metal structure aimed at facilitating the propagation of pelagic species, and concrete fish homes, such as Fish Boxes[TM] and ReefBalls[TM], which are designed to form habitats for a myriad of different species (Fig. 1; Sherman *et al.*, 2002; Haejoo, 2011, 2013).

Fig. 1. Photographs of different HES produced from (top row) materials of opportunity, from left to right; the Tangalooma Wrecks (www.queensland.com), tyre reef at Moreton Bay, Queensland (www.divingthegoldcoast.com) and disused oil rig of the Gulf of Mexico (www.nytimes.com) and (bottom row) purpose-built artificial reefs, from left to right; Abalone habitat reef, a Fish Box[TM] and the Sydney OAR (www.haejoo.com/purpose-built-artificial-reef).

Study sites and artificial reefs

Geographe Bay, is located on the lower west coast of Australia and ranges from the Bunbury breakwater (33° 18'S, 115° 39'E) in the north to the northwest point of Cape Naturaliste (33° 32'S, 115° 00'E) in the south. It covers an area of ~290 nautical miles[2] and has a maximum water depth of 30 m (Bellchambers *et al.*, 2006). Due to its north facing aspect and being exposed to prevailing south-westerly swell, makes Geographe Bay the southernmost protected embayment on the west coast of Australia (Bellchambers *et al.*, 2006). Geographe Bay exhibits an array of different habitats, ranging from low profile reefs to large seagrass meadows, with limited areas of sandy habitat. At depths of 2-14 m, the bay

is dominated by expansive meadows of the seagrasses *Amphibolis griffithi* and *Amphibolis antarctica* (Walker *et al.*, 1987; Bellchambers *et al.*, 2006), while *Posidonia sinuosa* dominates at deeper depths (Oldham *et al.*, 2010).

The influence of currents on Geographe Bay vary seasonally, with the poleward flowing Leeuwin Current flowing in winter, while a cool equator-ward flowing coastal counter current, the Capes Current, occurs in summer (Pearce and Pattriachi, 1999). When the Leeuwin Current moves offshore between November and March, initiating the Capes Current, there may be localised upwelling, which influences local fisheries (Gersback *et al.*, 1999; Pearce and Pattriachi, 1999). Geographe Bay experiences microtidal conditions, with the mean tidal range being < 1 m resulting in most water movement occurring as a result of winds (McMahon *et al.*, 1997). The bay is a key recreational hotspot for people from the towns of Dunsborough and Busselton and the city of Bunbury, as well as tourists from other regions, particularly the state capital, Perth.

Geographe Bay was chosen as a suitable site for the deployment of the artificial reefs primarily because of the passion of local recreational fishers, who had promoted the deployment of artificial reefs for many years and that such structures might increase tourism into the area (Mark Pagano, Department of Fisheries WA pers. comm., 2015). Furthermore, the artificial reefs could not be deployed north of Bunbury due to large amounts of sediment being flushed from the Leschenault Estuary during winter and the presence of a nearby colony of Little Penguins (*Eudyptula minor*), which could be negatively affected by an increase in boat traffic. Prior to deployment, constraints mapping was employed to analyse any social or experimental limitations on the success of the reefs policy. The design, construction, placement and relationship of artificial reefs with the hydrology, sediment dynamics and surrounding environment were considered throughout the project (Department of Fisheries, 2012a). The Department of Fisheries, together with the South West Artificial Reefs Reference Group, which comprised scientists and environmental and fisheries managers and key stakeholders, used the following criteria to identify possible sites within Geographe Bay - (i) likely to attract key nearshore recreational species, (ii) in close proximity to boat ramps to allow safe access by small vessels, (iii)

situated over predominantly sand substrate to avoid seagrasses, (iv) compliance with state and commonwealth marine park zoning and (v) in water depths of between 20 and 30 m (Department of Fisheries, 2012a).

In April 2013, 60 purpose built modules were deployed to create two separate artificial reefs off the coasts of Bunbury and Dunsborough in Geographe Bay, creating the South West Artificial Reef Trial. Each of the modules (FishBox^{TM}) is constructed from steel-reinforced concrete, is 3 m^3 and weighs 10 tonnes (Fig. 2; Haejoo, 2013). To construct each reef, 30 modules were grouped into six clusters of five modules (Fig. 3), over an area of four hectares (Haejoo, 2013). The Bunbury artificial reef was deployed around 115° 35.900'E, 33° 18.500'S in a water depth of 17 m depth, while the artificial reef at Dunsborough was deployed around 115° 9.980'E, 33° 3.962'S in a water depth of 27 m (Fig. 3; Department of Fisheries, 2013). To ensure that the reefs are easily assessable to recreational fishers both were located within 5 km, as the crow flies, of boat ramps. Each module is designed to promote upwelling, by driving nutrients up the water column, due to the curvature of the concrete cross braces, as well as to provide shelter and variation in environmental effects such as light, temperature and hydrological variables to increase habitat (Haejoo, 2011; Department of Fisheries, 2012a). The primary aim of the artificial reef was to provide additional habitat for key fish species of recreational interest, such as Pink Snapper (*Chrysophrys auratus*), Samson Fish (*Seriola hippos*) and Silver Trevally (*Pseudocaranx georgianus*).

Monitoring and citizen Science

To meet legislative requirements, any artificial reef deployed in WA has to have a dedicated monitoring and management plan, to ensure the structural integrity of the structure (Department of Fisheries, 2012a). During the structural surveys, the Department of Fisheries is also monitoring of the success of the artificial reefs in attracting fish species, increasing fish biomass and altering behaviour (*e.g.* if fish feed and/or reproduce in association with the reefs). The reefs are very popular with the local community and organisations associated with recreational activities (*e.g.* fishing and scuba diving), and the commercial
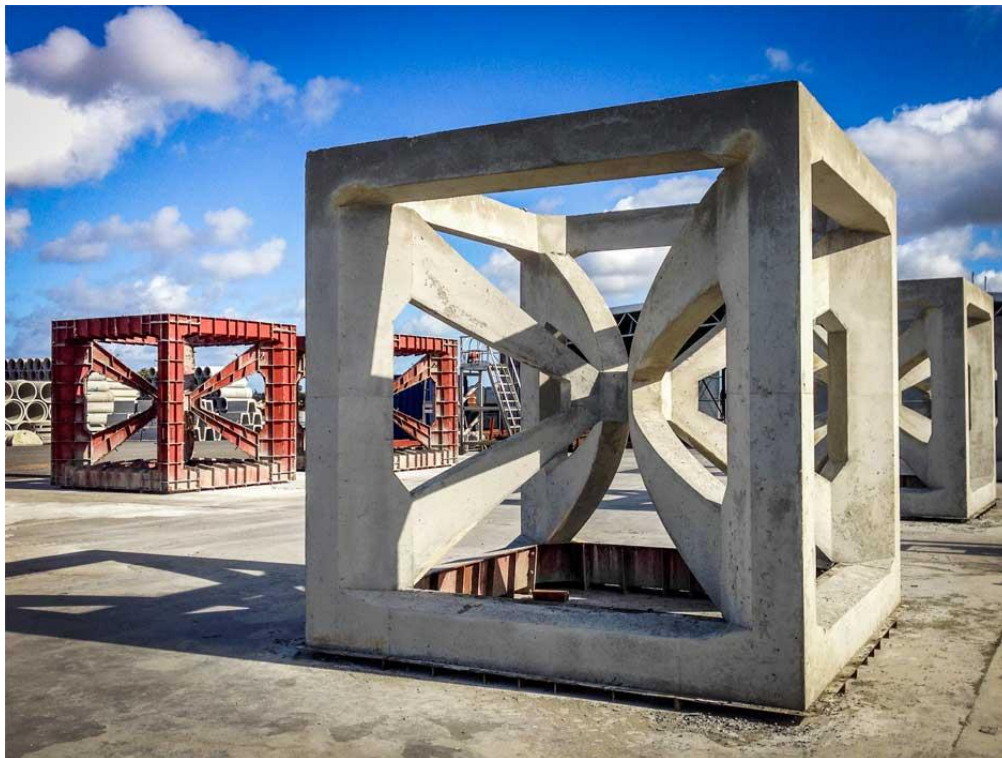
Fig. 2. Some of the 60 Fishbox^{TM} modules being constructed to be deployed in the artificial reefs off Bunbury and Dunsborough. Image courtesy of Haejoo.
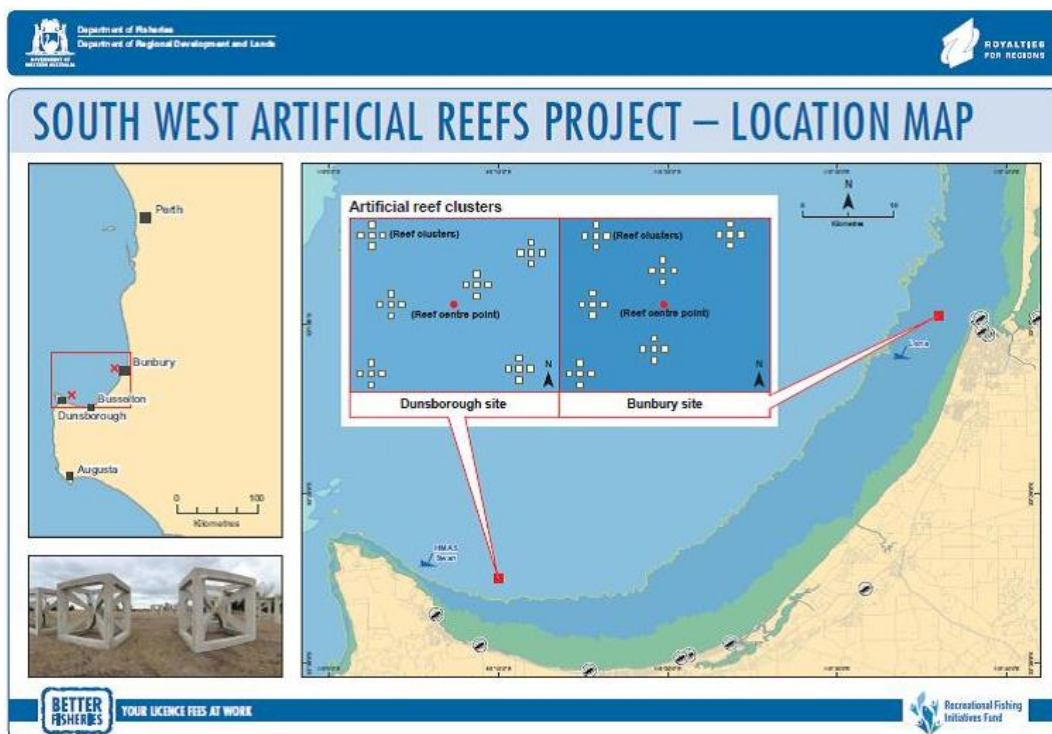
Fig. 3. Maps showing the location and spatial arrangement of the two purpose built artificial reefs in Geography Bay, Western Australia. Image courtesy of the Department of Fisheries WA.

sector are also interested in utilising these structures. However, the high cost associated with designing/selecting, purchasing, deploying and monitoring, *i.e.* at least $2.38 million in the case of the Bunbury and Dunsborough artificial reefs (Department of Fisheries, 2015), can be prohibitive for proponents looking to construct such structures.

One mechanism of reducing the cost of artificial reefs would be to use citizen science to collect monitoring data. Citizen science involves the use of volunteers to conduct research, sampling, data collection and/or analyses or monitoring (Thiel et al. 2014). This has the potential to reduce funding and labour costs to research organisations and thus increase cost efficiency, whilst also providing social benefits to volunteers and the opportunity for the collection of extensive data sets over large spatial and temporal scales. It also result in increased community ownership/stewardship of the project (Pattengill-Semmens and Semmens, 2003; Conrad and Daoust, 2008; Dickinson *et al.*, 2010; Tulloch *et al.*, 2013; Wilson and Godinho, 2013). It is thus not surprising that, in recent years, there has been a marked increase in the use of members of the general public to assist in scientific research (*e.g.* Silverton, 2009; Baltais, 2013; Lambert, 2014; Thiel *et al*., 2014).

Project aims

This study aimed to utilise a small suite of keen recreational fishers, as citizen scientists, to collect underwater video footage from both the Bunbury and Dunsborough artificial reefs, and nearby natural reefs, to help assess whether volunteers could effectively monitor spatial and temporal trends any in fish assemblages on the between the two types of reefs.

Specifically this project had the following milestones

- Identify 12 keen recreational fishers to participate in the study.
- Research a number of potential camera options and conduct a trial using the recreational fishers and incorporate their feedback.
- Design a sampling regime.
- Purchase cameras and train participants.
- Produce log books.

- Analyse videos recorded by recreational fishers.

- Provide training opportunity for Honours students.

The activities of this project can be divided into two phases, as follows.

In the first phase, the project set out to engage 12 recreational fishers and provide them with live action underwater camera to capture footage of the fish faunas of the artificial reefs and also nearby natural reefs (Section 2). During this phase of the project, human and animal ethics approval to conduct the research was obtained (Section 2). Information packs (e.g. information sheet, consent form, log book) for the recreational fishes were also developed (Section 2, Appendix 2.1-2.5). However, only a small amount of video footage was collected from the recreational fishers. This was partly due to a lack of engagement with the recreational fishers and the poor quality of the footage obtained from the cameras that were provided to the recreational fishers (Section 2).

In the second phase, the project attempted to address some of the problems experienced in the first phase. Specifically, a critical review of citizen science elements of the project was undertaken and used to help develop an approach to improve engagement of recreational fishers in the project (Section 3). Furthermore, the suitability of the footage captured by a Baited Remote Underwater video (BRUV) system, designed by Ecotone Consulting, to monitor the fish assemblages of the artificial reefs in Geographe Bay was investigated (Section 4). This was done to assess the potential for using these BRUVs in any future citizen science monitoring of the artificial reefs. In addition, the project also investigated the potential for observers to bias the data recorded on BRUV footage supplied by Ecotone Consulting from the Dunsborough Reef (Section 5). This experiment was undertaken done because any citizen science approach to monitoring the fish assemblages of the artificial reefs in Geographe Bay will necessarily involve multiple people (e.g. volunteers and students) recording data from footage. Consequently, it is important to understand how different people might record the data differently and thereby influence the results. A preliminary investigation of the fish fauna of the two artificial reefs, *i.e.* Bunbury and Dunsborough, was also conducted, using BRUV footage supplied by Ecotone Consulting (Section 6).

This study has provided useful information on workers looking at developing citizen science projects, particularly those using underwater camera (see sections 2-6 and particularly Section 7 which describes the conclusions and recommendations for future citizen science projects monitoring artificial reefs). These lessons have been actively used to the development of Reef Vision (the successor to the current project), which forms part of a larger Fisheries Research and Development Corporation project (2014/005); see Section 8 for more details, together with a review of citizen science methodologies and projects (Appendix 3.1).

Note that the sections of this report describing the outcomes of each of the sections of the project have been taken directly from honours theses written by James Florisson (sections 2-4 and 7) and Thomas Bateman (sections 5-6). However, minor modification has occurred to reduce replication, particularly in the introduction and materials and methods sections. Full copies of these theses can be found at http://researchrepository.murdoch.edu.au/29398/ and http://researchrepository.murdoch.edu.au/29645/, respectively, and the thesis abstracts are provided at the end of the report (additional appendices 1 and 2).

Section 2: Citizen science monitoring of the fish communities on Bunbury and Dunsborough artificial reefs and comparisons to nearby natural reef

Overview

This section of the report is based on Chapter 2 in James Florisson's honours thesis, which was completed in June 2015. It details the first attempt at using a small team of recreational fishers to monitor the fish faunas of artificial (and natural) reefs by collecting footage using an underwater camera with a live feed. Unfortunately, however, only very limited amounts of data were collected due to lack of participation, the short timeframe of the study and unseasonal bad weather. This lack of data severely limited the hypotheses able to be tested and the range of statistical analyses employed. However, a preliminary assessment of the fish faunas of artificial and natural reefs was able to be undertaken and demonstrated that artificial reefs harboured greater a number of species and mean and maximum abundances of fishes. Importantly, the lessons learned about engagement with recreational fishers have been applied to future work (see *Reef Vision* section).

Introduction

In recent years there has been an increase in the number of Habitat Enhancement Structures (HES), *i.e. purpose built structures or materials placed in the aquatic environment for the purpose of creating, restoring or enhancing a habitat for fish, fishing and recreational activities in general, in coastal waters worldwide* (Diplock, 2010; Department of Fisheries, 2012a). Of the many types of HES, artificial reefs are the most common and have been deployed in more than 50 countries around the globe (Diplock, 2010). An artificial reef is an anthropogenically manipulated underwater structure deployed for a range of purposes. While they serve a range of functions *e.g.* engineering solutions for coastal erosion and providing locations for recreational activities, such as surfing and diving (Brock, 1994; Baine, 2001; Ng *et al.*, 2014), these reefs are typically employed to increase the abundance and diversity of marine life within an area by creating additional shelter, food sources and a colonising surface for marine organisms (Svane and Peterson, 2001).

In Western Australia, two purpose-built artificial reefs were deployed off the coasts of Bunbury and Dunsborough, on the lower-west coast of Australia in April 2013. Each reef is made up of six clusters of five modules, each module being three cubic meters and weighing ten tonnes (see Fig. 2). Each module is designed to promote upwelling by driving nutrients up the water column due to the curvature of the cross braces made from re-enforced concrete, as well as provide shelter and variation in environmental effects such as light, temperature and hydrological variables to increase habitat (Haejoo, 2011; Department of Fisheries, 2012a). The primary aim of the artificial reef was to provide additional habitat for key fish species of recreational interest, such as Pink Snapper (*Chrysophrys auratus*), Samson Fish (*Seriola hippos*) and Silver Trevally (*Pseudocaranx dentex*).

To meet legislative requirements, any artificial reef in WA has to have a dedicated monitoring and management plan, to ensure the structural integrity of the structure (Department of Fisheries, 2012a). At the same time, monitoring of the success of the artificial reefs in attracting fish species, and increasing fish biomass (*e.g.* if fish feed and/or reproduce in association with the reefs) is also being measured by the Department of Fisheries during the structural surveys. Although these structures are very popular with the local community and organisations associated with recreational activities (*e.g.* fishing and scuba diving), there is also interest from the commercial sector in utilising these reefs. The high cost associated with designing/selecting, purchasing, deploying and monitoring, *i.e.* at least $2.38 million in the case of the Bunbury and Dunsborough artificial reefs (Department of Fisheries, 2015), are prohibitive. One mechanism of reducing the cost of artificial reefs would be to use citizen science to collect monitoring data, a method which would also result in increased ownership/stewardship of the structures by the community (Pattengill-Semmens and Semmens, 2003; Conrad and Daoust, 2008).

In light of the above, this section of the project utilised a small suite of keen recreational fishers as citizen scientists to collect underwater video footage from both the artificial reefs, and nearby natural reefs, to help elucidate whether volunteers could effectively monitor the differences in fish assemblages potentially caused by artificial reefs. The initial aim was to analyse video footage collected by the recreational fishers to determine whether the fish

communities on the artificial reefs were similar to those on nearby natural reefs and thus whether the artificial reefs were fulfilling their objective of enhancing the surrounding habitat. For various reasons, however, very little footage was obtained from the recreational fishers (see Section 3). In view of the limited footage, the revised goal was to use the footage that was available to make a preliminary assessment of the fish assemblages of the Dunsborough and Bunbury artificial reefs during the first 15 months of their deployment.

Materials and methods

Site description

This study was conducted on the artificial reefs located in in Geographe Bay near Busselton. The artificial reef is located approximately 5 km from the Dunsborough boat ramp at 33° 3.962'S 115° 9.980'E. Full details of the composition and design of the artificial reefs and on Geographe Bay and its environmental characteristics are given in Section 1.

Citizen science

The citizen science methodology employed during this project is shown as a flowchart in Fig. 4.

Citizen Science Methodology for Artificial Reef Monitoring

Project Commences

Obtaining Permits

- Human ethics permit obtained to conduct citizen science project
- Ensured volunteer privacy, anonymity and consent

Contacting Fishers

- 6 fishers needed from each region, 4 obtained from each region
- initially fishers sourced by Recfishwest, after a decrease in volunteer numbers, more were sourced by word of mouth, Busselton Mail newspaper and local tackle stores
- Communication between volunteers and project managers
- To qualify, fishers had to fill out a questionnaire and consent form

Camera Trial

- 4 fishers from each region trialled the 2 cameras
- Testing if cameras suitable for volunteer use as well as suitability for monitoring marine environment

Fishing/Data Collection

- Volunteers given equipment and instructions
- Volunteers to record data on logbooks and camera data in USBs
- Range of methods to transport data from fishers to project managers
- Communication between project managers and volunteers to track progress of the project

General Volunteer Management

- Establish communication network between volunteers and project managers
- Instructions and user friendly equipment supplied to volunteers
- Multiple mid-project volunteer satisfaction surveys
- Post-project volunteer attitudinal survey

Fig. 4. Flowchart detailing the citizen science aspects of the project.

Selection of participants

The project aimed to recruit twelve avid recreational fishers, with six living in close vicinity to, and regularly fishing, the artificial reef in Bunbury and likewise another six for the Busselton-Dunsborough reef. The selection of twelve participants was a trade-off between the need to collect sufficient data and the need to cap project costs, given the relatively high cost of the cameras. This number of volunteers was also chosen to help mitigate against any attrition from the project due to issues with fishers, such as boat malfunctions, personal/family issues (*e.g.* illness) and functions (*e.g.* going overseas on holiday) or participants becoming their disengage and not collecting any footage.

The selection process was initially undertaken by Recfishwest, the peak body for recreational fishing in Western Australia. Firstly an advertisement was placed in Recfishwest's electronic newsletter that is emailed to members every month. This led to 14 applications, of which 8 were enlisted into the project (four in the immediate vicinity of each artificial reef). The remaining four fishers were selected and recruited through direct contact with staff members at Recfishwest. After recruitment into the study, a project manager from Murdoch University contacted the fishers by phone and email and then travelled to Bunbury and Busselton-Dunsborough to speak, in person, to each of the volunteers.

To participate in the project, each volunteer had to complete a questionnaire, which included details on the participant's recreational boating license, boating experience, type of vessel, availability of safety equipment and contact details (Appendix 2.1). The purpose of the questionnaire was to make sure that the volunteers had a clear understanding of the instructions and methodology for the project. It was also used ensure that each volunteer conformed to the relevant marine licencing requirements, *i.e.* licensed and insured vehicles with the required safety equipment, as specified by the Western Australian Department of Transport and meet certain safety requirements, *i.e.* that had ample experience as a skipper, would use a boat that was suitable for travelling to and from the reefs. Each participant was also required to fill out and sign a consent form to confirm that they voluntarily committed to the study and understood the circumstances around instructions, responsibility, and the rights of the volunteer and that they will not be personally identified

in any publication (Appendix 2.2, 2.3). This form was approved by the Murdoch University Human Research Ethic Committee (Permit 2014_005). To maintain volunteer privacy, participants were not to be identifiable by name, only by a volunteer code, and this information was stored separately to any data.

Camera trial

In order to select the most appropriate underwater cameras for use by the recreational fishers, 12 different models were compared in desktop study and the two that were most suitable in terms of their (i) safety and ease of use for fishers (ii) ability to collect footage of adequate quality and (iii) ability to stream live footage back to the fisher on the boat to reduce snagging in the artificial reef modules, were purchased and trialled by the volunteer fishers. The two of the cameras that were deemed the most appropriate were the Sony Charged-Coupled Device (CCD) 700 TVL Underwater Fishing Camera and the EelCam Diving Fishing Camera 1/3'' 800TVL CMOS Fish-shape and 7'' LCD Monitor Kit. Selected volunteers were asked to trial these two cameras and specifically to assess, i) ease of use and ii) potential safety issues. Ease of use is a major facet of citizen science, as user-friendly technology is a contributing factor to overall volunteer satisfaction (Newman *et al.*, 2010), while insurance and workplace health and safety are emerging concerns in many contributory and collaborative projects (Baltais, 2013). Safety considerations included the weight of the equipment and any potential tripping hazards associated with the 50 m of cable required for the camera to comfortably reach the reefs.

Each volunteer involved in the trial selected the Sony CCD 700 TVL, primarily as this camera did not spin when the vessel was drifting and thus the operator had more control over the direction of the cameras field of view, was easier to operate and had less chance of entanglement in the modules due to its shape. Fishers also indicated that the 2 GB SD cards could store only very limited quantities of footage and thus all camera kits were provided with 32 GB SD cards. The feedback from volunteers at this stage was intrinsically important to the project. Firstly, it provided sound advice on the pros and cons of the various cameras, leading to the selection of the most appropriate camera. Secondly, it allowed volunteer

feedback to influence the project methodology, which may mitigate negative interactions between volunteers and the equipment in the future, whilst also giving participants a sense of ownership over the project.

(a)

(b)

Fig. 5. (a) The Sony Charged-Coupled Device (CCD) 700 TVL Underwater Fishing Camera, with 50 m cable and 360° rotating head and (b) the EelCam Diving Fishing Camera with 1/3'' 800TVL CMOS Fish-shape camera and 7'' LCD Monitor Kit.

the project methodology, which may mitigate negative interactions between volunteers and the equipment in the future, whilst also giving participants a sense of ownership over the project.

Following the completion of the camera trial and arrival of the Sony CCD 700 TVL cameras, the project manager travelled to meet each volunteer and give them their camera, verbal and written instructions on how to use them and an information sheet containing written instructions, artificial reef cluster location coordinates and contact details (Appendix 2.1, 2.4). Volunteers were asked to visit each cluster of the artificial reef modules and a nearby natural reef every month and record up to 15 minutes of footage. However, after feedback from several of the participants, the duration of the monitoring was changed from 15 minutes on each cluster and natural reef area, to around five minutes on each, and to at least 15 minutes in total per month on each cluster. For each video, volunteers were also asked to complete a record in a logbook (Appendix 2.5). The logbooks collected information on: (*i*) submersion time of the camera (to check it was for an adequate period of time); (*ii*) whether on natural or artificial reef (to help with metadata analyses and comparing differences between the natural and artificial reefs); (*iii*) which species were caught and their total size (to see if these species were similar to the species sighted in the footage) and any general comments or environmental observations (to help identify any outliers, patterns or different variations in the footage, such as different species in relation to time of day, or change in turbidity after a storm).

Logbooks were large with limited text and were taken on board the vessel during monitoring (Appendix 2.5). Volunteers were asked to transfer data to researchers at Murdoch University. Initially a cloud (internet) storage method, using the Dropbox software package, was trialled as this would automatically download any videos uploaded by volunteers to the researchers. This software, however, proved was too complex for the volunteers to use and thus USB sticks were employed. Once filled with video footage, the USBs could be mailed directly to Murdoch University, picked up by the project manager when visiting the fishers and/or be dropped off at the nearest Department of Fisheries office. While, in addition to the above methods, the logbook could be scanned or

photographed and emailed. The project aimed to collect video footage and the corresponding logbook notes monthly from each participant.

Video metadata

Once received, a suite of metadata were recorded for each video, namely footage code (1-999), footage number (1-999), fisher code (F1-F8), locality (Bunbury or Dunsborough), habitat type (artificial or natural), length of footage (seconds), file size (megabytes) and quality rating. The quality rating was a visual assessment of the clarity of the footage and was assessed across the entire video. The scales ranged from 1 (worst quality) to 10 (best quality) and incorporated factors such as turbidity, water and camera movements, video length and the amount of fish and structures identifiable (Fig. 6).

Observation protocols

For each video, the values of two quantitative variables, namely Max-N and Count-N were recorded for each species observed. The first of these variables, Max-N, is the maximum number of individuals of a species observed simultaneously during the video, *i.e.* the largest number in a single video frame (Priede and Merret, 1996; Willis and Babcock, 2000). This variable is commonly used as an indication of abundance because, by counting the maximum individuals of one species in the field of view at one time, it avoids the possibilities of double counting the same individuals (in a different frame) and gives a conservative estimation of relative fish density (Priede *et al.*, 1994; Cappo *et al.*, 2004; Watson *et al.*, 2005; Gomelyuk, 2012).

The second variable calculated for each species in each video was Count-N, *i.e.* the total number of individuals of a species seen during an observation period (Schobernd *et al.*, 2013; Mallet and Pelletier, 2014; Wartenburg and Booth, 2014). Count-N enumerates and identifies all individuals observed in 'digital transects', effectively imitating an in-situ slate-transect enumeration. Thus, this variable identifies and counts all individual fish that appear on the screen (Wartenburg and Booth, 2014). Each species recorded was also assigned to an ecological group affinity using the Nakamura (1985) classification. Under the Nakamura classification each species is classified based on their typical spatial position with regard to

Quality level 1

Very turbid. Screen has grain-like effect. Camera shaking. Video only lasts 10 seconds.

Quality level 3

Quite turbid but fish and modules visible. Small grain effect on screen. Limited shaking. Video lasts over one minute.

Quality level 5

Fish easily identifiable in close proximity to the camera. Reduced shaking and greater clarity. Epiphytic growth observable on modules. Video lasts over two minutes.

Quality level 10

All fish easily identifiable. No shaking and excellent clarity. All fish easily identifiable. Module growth easy to observe. Footage length over 15 minutes.

Fig. 6. Examples of different quality levels of footage obtained from the Sony CCD 700 TVL Underwater Fishing Camera. Note that footage at quality level 10 was taken from the video footage collected later in the project using a GoPro Hero 4 on the same artificial reefs and is shown here for comparative purposes.

the reef (Tessier *et al.*, 2005; Bortone, 2007). A-type species are found proximate to/or inside holes and crevices on the reef and are thus classified as benthic. B-type species are found closely associated with the reef, but not in direct contact are known as epibenthic and C-type species are loosely associated with structure, often found schooling above it and distinguished as pelagic species (Nakamura, 1985; Bortone, 2007; Wartenburg and Booth, 2014). The number of modules per video was also analysed to analyse whether there was a localised effect on fish assemblages.

Multivariate analysis of fish community composition

The count-N data for each species in each video was standardised by dividing that number by the length of that video (in seconds) and multiplying by 60, to give a count per minute for each species in each video. All videos less than one minute were removed from the data set as they were too short to contain any species. Individuals that were unable to be identified were also removed from the data set (Lek *et al.,* 2011). The data matrix was then square-root transformed to down-weight the contributions of species with consistently relatively high values and balanced them with the values of rarer species and used to construct a Bray-Curtis similarity matrix. This matrix was then subjected to a one-way Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson, 2001) test to determine whether the fish communities on the two reef types, *i.e.* artificial and natural reefs, differed significantly. This test was chosen as it is robust enough to cope with the unbalanced design of 3 samples from natural reefs vs 12 from artificial reefs (see Anderson *et al.,* 2008). The above Bray-Curtis similarity matrix was then subjected to non-metric Multi-dimensional Scaling (nMDS; Clarke, 1993) to produce an ordination plot to explore visually any trends among reef types.

A shade plot, derived from the square-root transformed fish fauna data for each video, was used to visualise the trends exhibited by the counts (per minute) of the various fish species across the artificial and natural reefs. This plot is a simple visualisation of the frequency matrix, where a white space for a species demonstrates that the taxon was never collected, while the depth of shading from grey to black is linearly proportional to the density of that taxon (Clarke *et al.,* 2014a; Tweedley *et al.,* 2015a).

Results

Video metadata

Of the eight participants, video footage was successfully obtained from three. Moreover, those three fishers recorded only 17 videos, with a total duration of just over one hour. Video length varied from 10 seconds to 13 minutes 24 seconds, with an average length being 3 minutes 45 seconds per video. The general reef location was not specified for the vast majority of videos from both artificial reefs (85%) and natural reefs (100%) *i.e.* the logbook data were incomplete (Table 1). Moreover, only four of the 17 videos (24%) were recorded over natural reef.

Table 1. The number (#) and percentage (%) of videos recorded from artificial and natural reef off Bunbury and Dunsborough.

	Artificial Reefs		Natural Reefs		Total	
	#	%	#	%	#	%
Bunbury	0	0	0	0	0	0
Dunsborough	2	15	0	0	2	12
Unknown	11	85	4	100	15	88
Total	**13**	**100**	**4**	**100**	**17**	**100**

The quality of the footage was generally low and ranged between 1 and 5 on the 1-10 scale (Fig. 7). The average quality of artificial reef footage (3.8) was similar to that of the quality of footage obtained over natural reef (3).
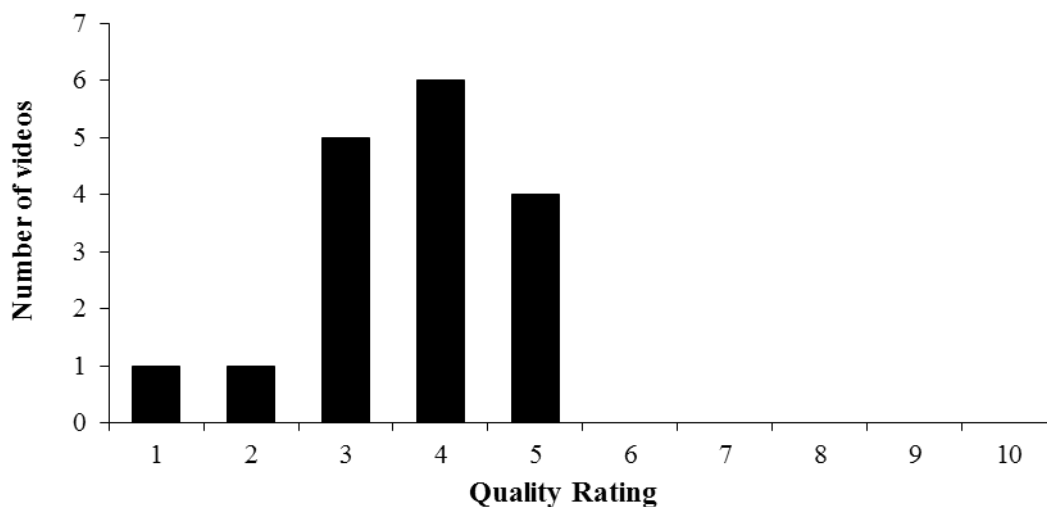
Fig. 7. The quality rating of the 17 videos collected on the natural and artificial reefs.

Univariate metrics

Of the thirteen species that were recorded across the 64 minutes of footage from the 17 videos (see Table 2), nine (69%) belonged to the 'B Type' ecological group indicating that they were epibenthic (Fig. 8). The remaining four species were equally assigned to the A (benthic) and C (pelagic) types.
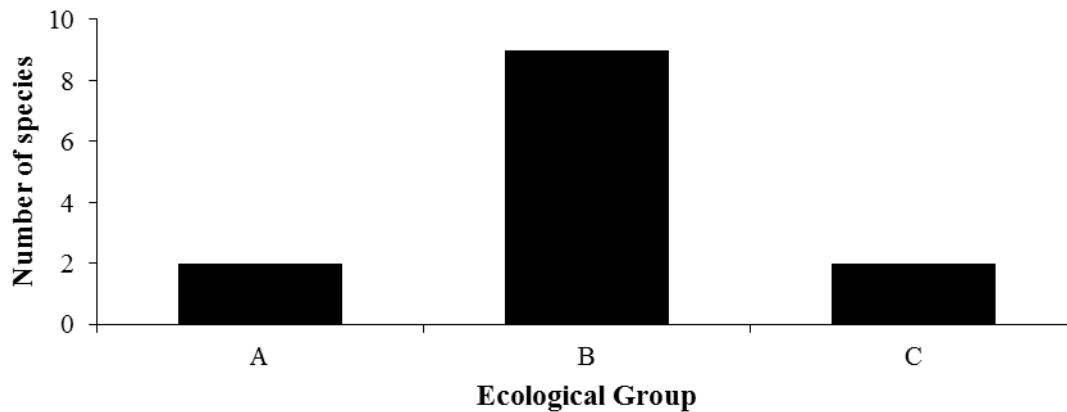
Fig. 8. The numbers of species recorded representing each of the three ecological groups defined by Nakamura (1985).

To test whether a larger amount of reef modules observed in the footage, had an effect on the fish assemblages, fish ecological groups as well as average mean abundance and average number of species was tested. In just over half (54%) of the 17 videos two or more of the five artificial reef modules could be sighted, while 23% of videos captured footage of one or two modules and the final 23% of the videos were filmed on natural reef (Fig. 9). Videos in which more than two modules were sighted contained larger numbers of mean individuals (numbers of individuals observed per minute of footage) of fish assigned to Type B ecological group (epibenthic species), while natural reefs had more Type A (benthic) and Type C (pelagic) species. The most abundant group overall was Type B, followed by C and A respectively (Fig. 10).
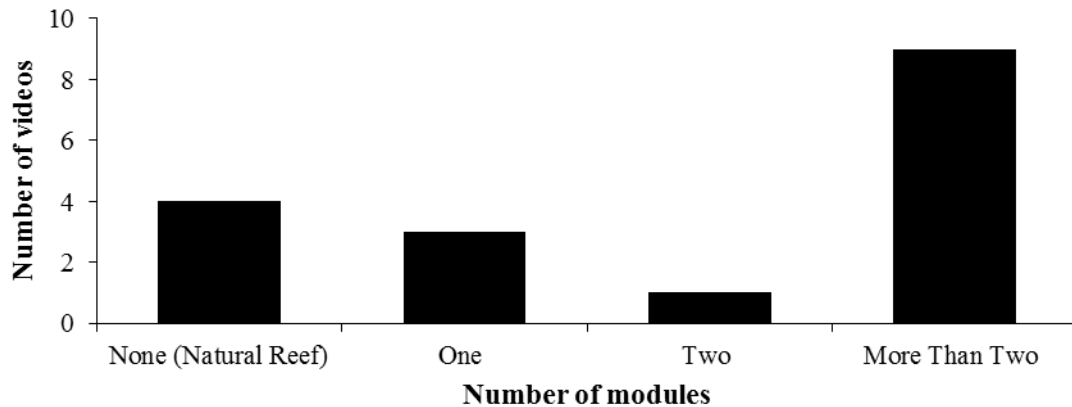
Fig. 9. The number of videos in which none, one, two or >2 modules were observed.
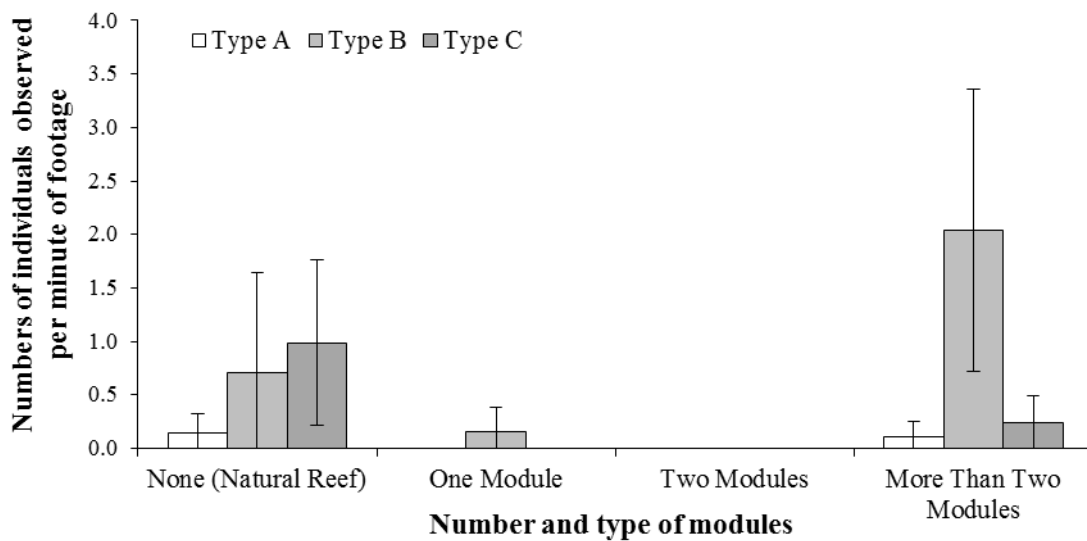
Fig. 10. The number of mean individuals (Max-N) from each ecological fish type (Nakamura classification) per minute of footage observing each number and type of modules. There is no data for two modules, as although fish were observed, they were unidentifiable. There error bars show the large variability between fish observed and the differing lengths of footage recorded (for example, there was 6.39 minutes for footage with one module and 38.24 minutes or over half the footage for more than two modules.

Average mean abundance (Max-N – mean abundance averaged to the amount of videos that exhibited each amount of modules) was far greater in videos that sighted more than two modules, rather than those recorded on natural reef or that sighted one module or two modules, *i.e.* ~10 *vs* ~3 and~0.3, mean individuals per video, respectively (Fig. 11). Average number of species was slightly higher (1.2) in videos with more than two modules, than those on natural reef or with one module (1; Fig. 11).
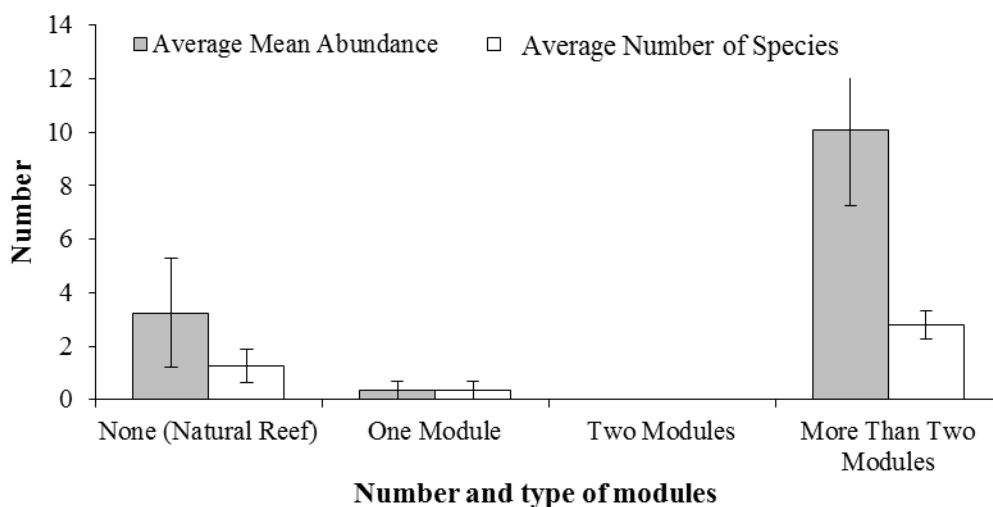
Fig. 11. The average mean abundance (Max-N – mean abundance averaged to the amount of videos that exhibited each amount of modules) and average number of species for differing numbers of modules encountered in the videos recorded on the natural and artificial reef. The error bars signify the variability of average mean abundance and average number of species in the differing amounts of modules.

Overall, thirteen identifiable fish species were recorded across the 17 videos and for each species in each video a max-N and count-N were recorded (Table 2). The Max-N ranged from 0 - 18, but was almost invariably < 5, while the number of identifiable species in a single video ranged from 0 - 6 and was typically ≤ 3 (Table 2a). Four species were recorded over natural reefs and 11 over the artificial modules, however, it should be noted that far fewer videos were recorded over natural reefs.

Among the fish species, the Western King Wrasse *Coris auricularis* was the most abundant, representing ~31% and ~47% of the maximum number of individuals on natural and artificial reefs, respectively (Table 2). While *C. auricularis* represented ≥ 5% of the total fish individuals (based on Max-N) on both reef types, six other species, representing more ≥ 5%

of the total fish individuals, occurred almost exclusively on only one or the other of the reefs types. These other species were the Southern Silver Belly *Parequula melbournensis*, Magpie Perch *Cheilodactylus nigripes* and Spinefoot *Siganus fuscescens* on natural reefs and the Footballer Sweep *Neatypus obliquus*, Sand Trevally *P. georgianus* and Rough Bullseye *Pempheris klunzingeri* over artificial reefs (Table 2a).

When considering species based on Count-N, far larger numbers of individuals per minute were recorded over artificial than natural reefs, *i.e.* ~10 and ~6, respectively (Table 2b). While, the Max-N of *C. auricularis* was the largest overall, and this species ranked 1st on artificial reefs (representing ~39% of all individuals), it only ranked 3rd over natural reefs, representing ~15% of the fish fauna. The most abundant species recorded over natural reefs was *P. melbournensis*, which although contributed almost 50% to the total number of fish recorded over natural reefs, was recorded on only 1 of the 13 videos over artificial reefs and represented < 4% of the total fish fauna. In contrast, *N. obliquus* contributed 28% to the fish fauna over artificial reefs, but was never recorded over natural reefs (Table 2b). Unidentifiable species, *i.e.* those that could be counted but not accurately assigned to a species, made up substantial contributions to the fish fauna of both reef types, representing 31.86% of the individuals observed on natural reefs and 18.81% of the individuals observed on artificial reefs.

Table 2. (a) Max-N and (b) Count-N values for each species recorded in each video. Note that Max-N values are for a single frame, while Count-N values are average for 1 minute of video footage. # = the count of values and % the percentage contribution made by that species to the total fauna of that video. Relatively abundant species, *i.e.* those that represented ≥ 5 % are shaded in grey. R = rank based on %. The number of species, individuals and length of each video is also provided.

(a) Max-N

Species Name	| Natural Reefs 1	2	3	4	#	%	| Artificial Reefs 1	2	3	4	5	6	7	8	9	10	11	12	13	#	%	| Total #	%	R
Coris auricularis		4			4	30.77	6	1			1		3		14				18	43	46.74	51	43.22	1
Neatypus obliquus											4	9							5	18	19.57	18	17.14	2
Parequula melbournensis	3	4			7	53.85													3	3	3.26	17	16.19	3
Pseudocaranx georgianus							4	2												6	6.52	6	5.71	4
Pempheris klunzingeri												5					1			6	6.52	6	5.71	4
Pentaceropsis recurvirostris									2	2										4	4.35	4	3.81	6
Anoplocapros amygdaloides							2		1	1										4	4.35	4	3.81	6
Choerodon rubescens								1					1						1	3	3.26	3	2.86	8
Anoplocapros lenticularis							1			1										2	2.17	2	1.9	9
Siganus fuscescens																			2	2	2.17	2	1.9	9
Cheilodactylus nigripes				1	1	7.69																0	0	2	1.9	9
Upeneichthys vlamingii		1			1	7.69														0	0	2	1.9	9
Chelmonops curiosus																			1	1	1.09	1	0.95	13
Number of species	1	3	0	1	4		4	3	2	3	3	1	2	0	1	0	1	0	6	11		13		
Number of individuals	3	9	0	1	13		13	4	3	4	10	9	4	0	14	0	1	0	30	92		105		
Length of footage	2:44	2:18	0:10	1:58	7:10		13:24	2:19	11:48	11:31	2:42	1:26	2:02	0:11	1:37	1:17	4:52	1:36	1:47	8:32		63:42		

(b) Count-N

Species Name	| Natural Reefs 1	2	3	4	X	%	| Artificial Reefs 1	2	3	4	5	6	7	8	9	10	11	12	13	X	%	| Total X	%	R
Coris auricularis		3.91			0.98	15.35	3.58	0.86			0.75		2.46		21.39				27.48	4.35	38.57	3.56	35.13	1
Neatypus obliquus											12.22	16.47							12.34	3.16	28	2.41	23.85	2
Unidentifiable species	0.37	5.22		2.54	2.31	31.86	3.73	1.29	0.58	0.53	2.96	1.4	0.98		1.24	1.56	0.82	1.25	11.21	2.12	18.81	2.1	20.75	3
Parequula melbournensis	5.49	6.96			3.11	48.8													5.47	0.43	3.73	1.05	10.41	4
Pseudocaranx georgianus							3.73	3.22												0.53	4.74	0.41	4.04	5
Pempheris klunzingeri							0.52		0.85	0.87										0.17	1.53	0.13	1.3	6
Pentaceropsis recurvirostris											1.85						0.25			0.16	1.44	0.12	1.22	7
Anoplocapros amygdaloides								0.43					0.49						0.57	0.11	1.18	0.09	0.87	8
Choerodon rubescens									0.76	0.35										0.85	0.76	0.07	0.65	9
Anoplocapros lenticularis							0.75			0.17										0.78	0.63	0.05	0.53	10
Siganus fuscescens				0.58	0.15	2.29																0.03	0.34	11
Cheilodactylus nigripes																			0.57	0.44	0.39	0.03	0.33	12
Upeneichthys vlamingii		0.43																	0.57	0.44	0.39	0.03	0.33	12
Chelmonops curiosus		0.43			0.19	1.75																0.03	0.25	14
Number of species	2	4	0	2	5		5	4	3	4	4	2	3	0	2	1	2	1	7	12		14		
Number of individuals	16	38	0	6	6.38		156	13	16	13	48	25	8	0	36	2	5	2	103	11.27		10.12		
Length of footage	2:44	2:18	0:10	1:58	7:10		13:24	2:19	11:48	11:31	2:42	1:26	2:02	0:11	1:37	1:17	4:52	1:36	1:47	8:32		63:42		

Multivariate analysis of fish community composition

One-way PERMANOVA demonstrated that there was no significant difference between the fish faunas recorded from video data collected over the two reef types (Table 3). This conclusion is supported by the nMDS ordination plot, were the three points representing the natural reefs were intermingled amongst those representing the artificial reefs (Fig. 12). Moreover, the shade plot shows that there was no clear division between the fish faunas of the artificial and natural reefs (Fig. 13). This was due to some of the few species that were recorded on natural reefs also being present on natural reefs, *i.e. C. auricularis* and *P. melbournensis*, but also the high degree of variability between the fish compositions of the artificial reefs.

Table 3. Mean squares (MS), Pseudo-*F* (p*F*) values and significance levels (*P*) for a one-way PERMANOVA test, employing a Bray-Curtis resemblance matrix constructed from the square-root transformed count-N data from the 15 videos recorded over artificial and natural reefs, which were obtained from recreational fishers.

	df	MS	p*F*	P
Reef type	1	3233	1.60	0.170
Residual	13	2033		
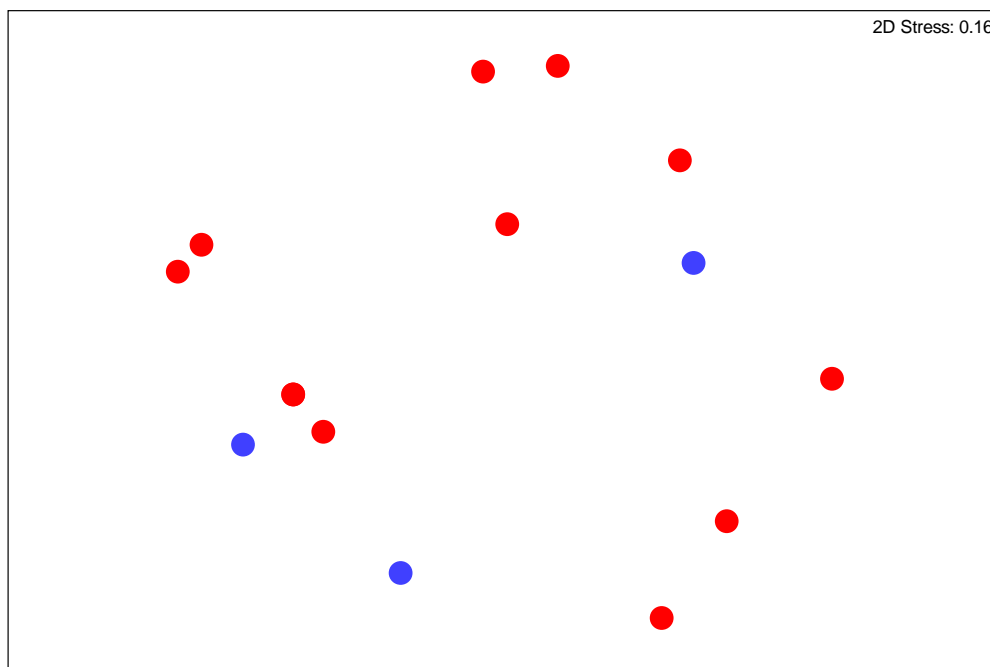
Fig. 12. nMDS ordination plot derived from a Bray-Curtis similarity matrix constructed from the square-root transformed count-N data from the 15 videos recorded over artificial and natural reefs, which were obtained from recreational fishers. ● Natural reefs. ● Artificial reefs.
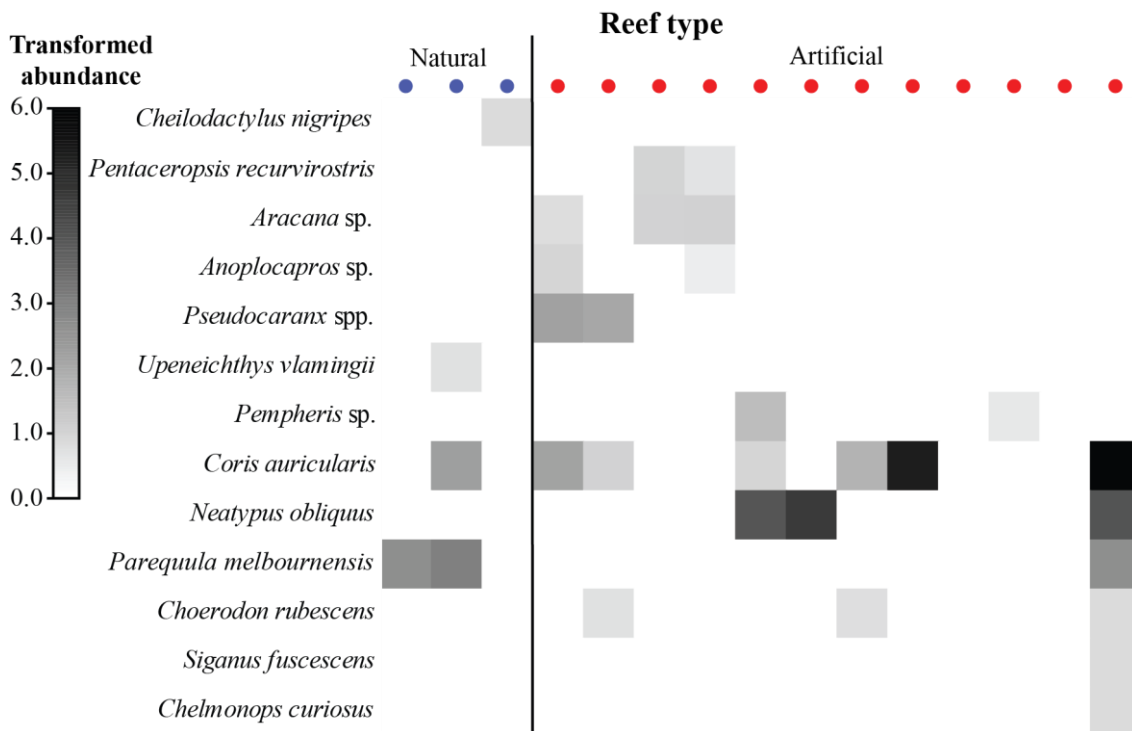
Fig. 2.13: Shade plot of the square-root transformed count-N data from the 15 videos recorded over artificial and natural reefs, which were obtained from recreational fishers. Grey scale represents the transformed counts of each species per minute. ● Natural reefs ● Artificial reefs.

Discussion

Data quantity and quality

Any discussion of the results of this section of the report should consider that the study was severely limited by a lack of data. Reasons for this are considered in detail later, but, in brief, were a lack of volunteer participation, poor quality of the video footage, the short timeframe of the project and unseasonal weather. As a result of these issues, only three of the recreational fishers submitted videos and these had a total duration of ~64 minutes. This limited amount of data is far less than was anticipated. Initially each of the eight fishers was asked to collect 15 minutes of footage on each cluster and 15 minutes on nearby natural reef, at least once a month for a four month period (which was later extended by another two months). This, even without the additional months, would have equated to 4 hours and 20 minutes of footage per fisher, giving a total of 34 hours and 40 minutes. However, the amount of footage received from the citizen scientists was only 3.1% of this initial figure.

Following the feedback that the memory capacity of the camera and SD cards were not sufficient to record the amount of video footage requested, the methodology was changed to ≥ 15 minutes per month on any cluster and 15 minutes on a nearby natural reef. Had this methodology been adopted for the entire data collection period, 16 hours of footage should have been required, which was 16x more footage than was received. In addition, of the 17 videos received, only two were accompanied by location metadata. A critical evaluation and suggestions on how to improve volunteer management of citizen science projects such as this are given below.

In terms of picture quality of the videos received from the fishers, all 17 had a quality rating of ≤ 5 (out of ten). This lack of quality was due to a grain-like effect limiting clarity, a small field of focus, glare and turbidity. For example, one video had approximately 30% of the screen covered by 'pink fuzz' for the entire duration caused by glare. As a result ~20% of all individual fish encountered, when standardised to the maximum abundance per minute of footage, were unable to be identified. These fish were unable to be identified due the quality of the footage as well as the distance from the camera and in some cases, high levels of turbidity. Although some individuals could be identified as far as the Family level, they could not accurately be identified to species level and thus were included as unidentified species. Although it's not rare to observe unidentifiable fish, ~20% is an abnormally large number to encounter (discussed below), and is likely due to the quality of footage, glare and turbidity. A study by Ebner *et al.* (2009) looked at whether remote underwater video can be used to investigate in-stream behaviour of small fishes and decapods in Cottie River, Australian Capital Territory. The study found 9.36% of individuals unidentifiable. Another study by Fischer *et al.* (2007), assessed the role of habitat complexity for fish using a small, semiportable, 3-D underwater observatory in Lake Constance, Germany. This study classified 10% of individual fish as unidentified because they could not be identified to species level. To fix the issue of identifiability, several studies only include the unidentified fish data in certain parts of the analysis, such as overall abundance measures (Gledhill *et al.,* 1996; Ebner *et al.,* 2009). A study assessing reef fish populations (Gledhill *et al.,* 1996) in the Gulf of Mexico included unidentified fish in estimates of general reef fish abundance, however excluded unidentified fish data from a species table for frequency of fish

occurrence 0.5 m or more above the bottom. This aforementioned study didn't divulge the number of unidentified individuals, just that they were observed on the video tapes. It should be noted that univariate and multivariate analyses and results (except Table 2) disregarded unidentifiable species as outliers, as they could not contribute to a Max-N or total species values or belong to a specific ecological group, although they could be grouped as unidentifiable individuals in a Count-N analysis. It should be discussed that this is therefore, another limitation to the data that ~ 20% of all individual fish encountered, when standardised to the maximum abundance per minute of footage could not be included in the preliminary results. This large percentage of fish, could have potentially altered the differences between abundance and number of species in footage from artificial and natural reefs.

Comparisons between the fish faunas of artificial and natural reefs

Many studies globally, have compared the fish assemblages of artificial and natural reefs. Of several studies analysed, the large majority of papers found both number of species and abundance to be significantly higher in fish assemblages on artificial reefs rather than natural reefs (Bohnsack *et al.,* 1994; Bombace *et al.,* 1994; Arena *et al.,* 2007; Booth and Fowler, 2013; Folpp *et al.,* 2013; Koeck *et al.,* 2014) while less papers found the number of species and abundance to be significantly higher on natural reefs (Burchmore *et al.,* 1985; Car and Hixon, 1997). Some studies found there is no difference between the structures (Fowler and Booth, 2012) and that natural reefs have a higher number of species but lower abundance (Hackradt *et al.,* 2011; Granneman and Steele, 2015). In a general sense, it is a challenge to identify a trend throughout the results of the papers due to the variation in research projects, *i.e.* spatial and temporal variation in fish assemblages and structures, distance to natural reef, dimensions of the artificial and natural reefs being compared and amount of habitat complexity each reef exhibits amongst other factors.

Although limited data were available, preliminary comparisons between the characteristics of the fish faunas of artificial and natural reefs were able to be undertaken. In terms of both Max-N and Count-N, greater numbers of fish were recorded on artificial rather than natural

reefs. While there were also seven more species recorded on artificial reefs, it is important to consider that more footage was collected over artificial reef than natural reef and, generally, those videos on the artificial reefs were longer. Thus, further sampling and analysis should be conducted to determine whether this is a *bone fide* finding or a sampling artefact. However, a potential explanation for the greater abundances and number of species recorded on artificial reefs may be due to the upwelling effect, vertical profile, range and complexity of the habitat, and growth on the modules (Bohnsack *et al.,* 1994; Kellison and Sedberry, 1998; Rilov and Benayahu, 2000; Svane and Peterson, 2001; Hunter and Sayer, 2009; Department of Fisheries, 2012a; Granneman and Steele, 2014).

Surveys of the substrate of Geographe Bay demonstrate that high profile reefs only represent a small proportion of the benthic habitats in the nearshore waters of the embayment, as the majority of the substrate comprises low profile reefs, sand and seagrass beds (McMahon *et al.,* 1997). It is therefore possible that the increased number of species and abundances of fish on the artificial reefs could be due to the relatively large vertical profile (3 m). This is supported by the findings of a study by Kellison and Sedberry (1998) who compared the abundances of fish on low and high vertical profile artificial reefs in Charleston, South Carolina in America. These authors found that the abundance of finfishes were significantly greater on the reefs with higher vertical profile. Moreover, research conducted by Harman *et al* (2003), on natural reefs in Hamelin Bay south-western Australia, found a significant difference between the numbers of species of sites location on high and low vertical profile reefs in the same area, with more species being found on reefs with higher vertical profile.

It is also possible that the combination of the two habitats, *i.e.* the artificial modules and the surrounding natural habitat, predominantly sand and seagrass, could create an 'edge effect', possibly resulting in species segregation, potentially driven by predation or competition (Dorenbosch *et al.,* 2005). Generally, edge effects are changes in community structure (fish assemblages) that occur at the boundary of two habitats (Harris, 1988). Depending on underlying mechanisms, the transition of different habitats may result in an 'edge effect' where species can potentially increase or decrease in abundance and biodiversity (Ries and

Sisk, 2004; Dorenbosch *et al.,* 2005). These increases or decreases of abundance and biodiversity along the boundary of two habitats can be caused by migration of individuals and fish schools between habitats, the presence of predators and the availability of food (Dorenbosch *et al.,* 2005).

The presence of sand and seagrass with the relatively high vertical, albeit artificial reef and the natural low profile reef also increases habitat complexity. Moreover, artificial reefs, such as those deployed in Geographe Bay, are designed to provide complex spaces and areas varying in water flow and light shade. These reefs can also provide cryptic spaces and shelter for a range of organisms including fish and invertebrates (Kellison and Sedberry, 1998; Charbonnel *et al.,* 2002; Hunter and Sayer, 2009). As a result they can have a positive ecological effect, often facilitating the development of highly diverse marine communities with characteristics (such as the recruitment, colonisation, succession and development of sessile biota) that reflect those of natural reefs (Svane and Peterson, 2001). A study by Hunter and Sayer (2009) tested species diversity and abundance on natural reefs, simple artificial reefs and complex artificial reefs, with the complex artificial reefs harbouring 2-3 times greater number of individuals for most species. This finding led the authors to conclude that 'enhanced habitat availability produced by the increased structural complexity delivered through specifically designed artificial reefs may have the potential to augment faunal abundance while promoting species diversity' (Hunter and Sayer, 2009).

Although the individual artificial reef modules are only three meters high, their unique cross brace design promotes not only shelter for fish habitats, but also potentially increases upwelling (Haejoo, 2011). Such a feature aims to 'force' water currents of colder, more nutrient-rich water from close to the substrate up and into the water column, thus providing a food source for plankton and larval fish, which, in turn, attract larger fish. This theory was tested in Bungo Channel in the Seto Inland Sea, Japan by Yanagi and Nakajima (1991), who deployed an artificial reef with the aim to induce upwelling. Field observations preformed before and after the deployment demonstrated that concentration of nutrients and chlorophyll *a* (the latter a surrogate for phytoplankton biomass) and biomass of zooplankton all increased after deployment.

Having been deployed 15 months before the start of this study, the artificial reef modules had had the opportunity to be colonised by a range of sessile organisms (see Fig. 6). The growth of these sessile organisms on artificial structures has been shown by Bailey-Brock, (1989), to provide food for some reef fish and eventually increase cover by adding to the three-dimensional structure of the reef. It is thus relevant that, compared to initial surveys at the deployment sites, after two years, four times more fish species have been recorded on the artificial reefs (Paul Lewis, Department of Fisheries, pers. comm.).

From a fish community perspective, PERMANOVA did not detect a significant different in the compositions of the fish fauna recorded over artificial and natural reefs. Shade plot analysis demonstrated that the lack of difference between the two reef types was due to the high levels of variability on the fish compositions within a reef type and the fact that several species were recorded in both environments. This highlights the fact that the above analysis should be approached with caution, due to the limited amount of data available and that more video footage is required to statistically analyse, in a robust quantitative manner, the fish faunas of the two types of reefs.

Future work

Due to the low amounts of footage received from the participants, the results detailed in this section should be considered preliminary. This lack of data (particularly the number of videos [samples]) reduced the suite of hypotheses available to test. However, if greater amounts of footage were received from the participants then it would have been possible to compare the fish faunas on the two artificial reefs (*i.e.* Bunbury and Dunsborough) in addition to the artificial *vs* natural reefs comparison. As the fish faunas of natural reefs around the world have been shown to change seasonally (Sale, 1980; Holbrook *et al.,* 1994; Felix-Hackradt *et al.,* 2013; Henriques *et al.,* 2013; Lopez-Perez *et al.,* 2013), it would be useful to see whether the fish fauna artificial reef changes temporally and, if so, whether it follows the same pattern of changes as natural reef. This would also identify the species which utilise the reef for large periods of time, *i.e.* resident species, and those more 'transient' species, which may utilise the reefs for shorter periods of time.

Information Sheet: Monitoring of Bunbury and Dunsborough Artificial Reefs

Monitoring of Bunbury and Dunsborough Artificial Reefs by Recreational Fishers

Information Sheet

Thank you for expressing an interest in participating in the artificial reef monitoring project.

Background

Artificial reefs were deployed in waters off Bunbury and Dunsborough in 2013. These reefs were deployed with the intention of attracting and providing new habitat for recreationally important species, such as pink snapper, silver trevally and samson fish (for more details refer to http://www.fish.wa.gov.au/Documents/artificial_reefs/south_west_artificial_reefs_flyer.pdf and http://www.recfishwest.org.au/fishing-in-wa/artificial-reefs.html)

Monitoring of the artificial reefs is required to understand the types and abundance of fish that visit and colonise the reefs, how these factors change through time and how the fish assemblages of the artificial reefs compare to those of nearby natural reefs.

This project
Participants will be given Fishing Inspection Cameras and asked to use these cameras to monitor the fish fauna of the artificial reefs and nearby natural reefs. The monitoring will occur between March 2014 and June 2015. One of the main goals of the project is to determine if recreational fishers can provide a cost effective means for monitoring artificial reefs. The project will also deliver important information about the composition and evolution of fish assemblages on artificial reefs.

1

Information Sheet: Monitoring of Bunbury and Dunsborough Artificial Reefs

The research team
This monitoring project is being run by Drs Howard Gill and Jennie Chaplin in the Centre for Fish and Fisheries at Murdoch University. It is being run in conjunction with Recfishwest and the Department of Fisheries and is funded through the Recreational Fishing Initiative.

Contact details

Howard Gill	*Jennie Chaplin*	*Recfishwest*
Centre for Fish and Fisheries Research, Murdoch University South Street Murdoch. WA. 6150	Centre for Fish and Fisheries Research, Murdoch University South Street Murdoch. WA. 6150	*PO Box 34* *North Beach* *W.A. 6920*
H.Gill@murdoch.edu.au	jchaplin@murdoch.edu.au	recfish@recfishwest.org.au
04 06 995 036	04 38 991 949	(08) 9246 3366

Information for participants
If you are selected to participate in the monitoring project, you will be given a Fishing Inspection Camera and asked to deploy the camera at either the Bunbury or Dunsborough artificial reef to record information on the fish fauna at this reef. You will also be asked to deploy the camera at a nearby natural reef to record the fish fauna there. The cameras will be deployed on a cable, which comes with the camera.

Participants will be asked to deploy the cameras as a part of their normal boating/fishing activities rather than to make special trips. The camera can be deployed at any time, before during or after fishing at the discretion of the participants.

All participants will be responsible for all of the costs that they incur in the conduct of the monitoring, except that one Fishing Inspection Camera will be provided to each participant. Participants can keep the camera after the project is completed in recognition of their valuable support. Participants will not be liable for any damage done to the Fishing Inspection Camera.

Participation in the project is voluntary. Participants can withdraw from the monitoring at any time. Participants will be asked to monitor the artificial reef and one nearby reef for at least one 15 minute period each per month. However, ultimately, the amount and timing of any monitoring done by a participant is at the discretion of the participant.

The identity of participants will not be disclosed to the public unless the participant volunteers this information e.g. if they volunteer to become involved in some of the project promotions run by Recfishwest and Murdoch University. The name and identity of participants will be stored separately from the data, and these will be accessible only to the investigators. All data provided by a

2

participant will be analysed anonymously using code numbers. Any inadvertent footage (i.e. any filmed footage other than fish or reef) will be deleted.

Selection criteria

Participants will need to regularly go boating/fishing in the vicinity of one of the reefs. Safety is of paramount concern in the conduct of the monitoring. All participants must have a minimum of a Western Australian recreational skipper's ticket and a suitable boat and safety equipment to participate. Participants must also have experience in boating in the general vicinity of the artificial reef or equivalent. Under no circumstances should a participant attempt to conduct any monitoring if the weather conditions are not suitable or in an unseaworthy boat or in a boat without appropriate safety equipment. All participants must agree to comply with all relevant legal and safety obligations. Refer to the following website for details http://www.transport.wa.gov.au/imarine/recreational-boating.asp

All participants are reminded that they are responsible for their safety and actions at all times - neither Murdoch University nor the research term nor Recfishwest are liable for any consequences arising from your participation in the project.

The next step

If you think you might be interested in participating in the monitoring, please complete the following questionaire and return to the research team or Recfishwest (contact details above).

Questionaire

Where appropriate, please circle the correct answer.

1. Do you have a minimum of a Western Australian recreational skipper's ticket?

    Yes.   *(proceed to question 2)*

    No.   *thank you for your interest, but it is not possible to participate in the monitoring without a minimum of a Western Australian recreational skipper's ticket.*

    Unsure.   *thank you for your interest, but it is not possible to participate in the monitoring without a minimum of a Western Australian recreational skipper's ticket.*

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Information Sheet: Monitoring of Bunbury and Dunsborough Artificial Reefs

2.   What type of skipper's ticket do you have, what is the ticket number, date of issue and expiry date (if applicable)?

Type:

Number:

Date of Issue:

Expiry Date :

3.   How many years of experience do you have as a skipper of boat in marine waters?

4.   Do you have a West Australian recreational fishing licence?

Yes.   *Please provide number and expiry date below.*

No.

Unsure.

Licence Number (if applicable):

Licence Expiry Date (if applicable):

5.   Provide details of the type of boat you would be using when monitoring if you were selected to participate.

Type & Size:

Motor(s):

Registration Number:

Registration Expiry Date:

Information Sheet: Monitoring of Bunbury and Dunsborough Artificial Reefs

6. What safety gear will be carried on the boat you would be using when monitoring if you were selected to participate? Circle equipment that applies.

    Lifejackets (minimum of one for each person on board)

    EPIRB

    Marine Radio

    Flares

    Anchor

    Bailer or bilge pump

    Fire extinguisher

    Navigational lights

    Other (please specify)

7. Would you follow the Department of Transport recommendations for voyage planning for any boating trips that involve the monitoring if you were selected to participate? (refer to http://www.transport.wa.gov.au/imarine/voyage-planning.asp for details)

    Yes

    No

    Unsure

8. Which artificial reef do you want to monitor?

    Bunbury

    Dunsborough

9. Briefly describe the nature of your experience of boating in the general area of the artificial reef that you want to monitor (e.g. number of years, as skipper, as a passenger, locations regularly visited etc).

Information Sheet: Monitoring of Bunbury and Dunsborough Artificial Reefs

10. Indicate approximately how often you fish in the vicinity of the area of the artificial reef that you want to monitor for each of the following time periods. Circle the nearest answer.

Summer	Autumn
about once per week	about once per week
about once per month	about once per month
less than once per month	less than once per month
Winter	Spring
about once per week	about once per week
about once per month	about once per month
less than once per month	less than once per month

11. Please provide your full name, address and contact details.

Name:

Address:

Phone:

E-mail:

This study has been approved by the Murdoch University Human Research Ethics Committee (Approval 2014/005). If you have any reservation or complaint about the ethical conduct of this research and wish to talk with an independent person, you may contact Murdoch University's Research Ethics Office (Tel. 08 9360 6677 or e-mail mailto:ethics@murdoch.edu.au). Any issues that you raise will be treated in confidence and investigated fully, and you will be informed of the outcome.

— Consent Form —

Monitoring of Bunbury and Dunsborough Artificial Reefs by Recreational Fishers

CONSENT FORM

Thank you for agreeing to participate in the artificial reef monitoring project.

1. I agree voluntarily to take part in this study.

2. I have read the Information Sheet provided and been given a full explanation of the purpose of this study, the procedures involved and of what is expected of me.

3. I understand that I will be given a Fishing Inspection Camera and asked to deploy the camera at either the Bunbury or Dunsborough artificial reef to record information on the fish fauna at this reef. I will also be asked to deploy the camera at a nearby natural reef to record the fish fauna there.

4. I understand that I will be responsible for my own safety during the monitoring and will comply with all relevant legal and safety obligations.

5. The researcher has answered all my questions and has explained possible problems that may arise as a result of my participation in this study.

6. I understand I am free to withdraw from the study at any time without needing to give any reason.

7. I understand I will not be identified in any publication or publicity arising out of this study, unless I choose to be.

8. I understand that my name and identity will be stored separately from the data, and these are accessible only to the investigators. All data provided by me will be analysed anonymously using code numbers.

1

Consent Form

9. I understand that all information provided by me is treated as confidential and will not be released by the researcher to a third party unless required to do so by law.

Name of participant: _____

Signature of Participant: _____   Date: ..../......./.......

I confirm that I have provided the Information Sheet concerning this study to the above participant; I have explained the study and have answered all questions asked of me.

Signature of researcher: _____ Date: ..../......./.......

Appendix 2.3. Ethical consent statement

This was provided in the information pack for participants along with the camera.

Thank you for participating in the monitoring of the Bunbury and Dunsborough artificial reefs by recreational fishers

In borrowing the Fishing Inspection Camera, you are consenting to comply with all relevant legal and safety obligations set down for participants in the project (see http://www.transport.wa.gov.au/imarine/recreational-boating-and-water-sports.asp). You are also consenting to Murdoch University using the underwater footage that you collect.

Artificial Reef Research Camera Guide

How to use the Sony CCD 700 Underwater Fishing Camera

Step 1: Charging the camera battery

- Connect the battery to the charger via the battery charging port as shown in blue below and plug in the charger.
- A red indictor light on the charger indicates the battery is charging.
- When battery is fully charged indicator light will turn green.
- IMPORTANT: To increase the life of the battery, allow the battery to charge fully before use and allow the battery to completely drain before recharging again.
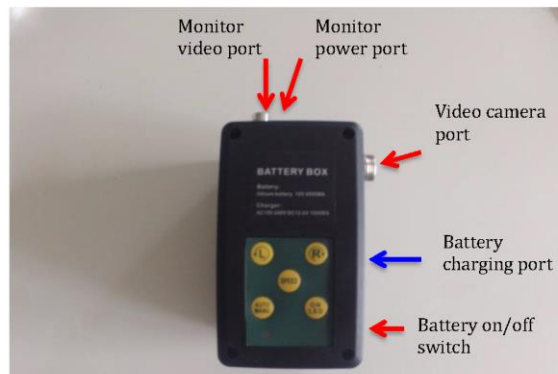
Step 2: Connecting up the camera and monitor

- To provide power to the monitor and camera ensure all three cords (two from the monitor and one from the camera), are properly connected and the camera cord is screwed on to secure it as shown below.
  - Yellow cord from monitor connects to monitor video port on the battery box.
  - Black cord from monitor connects to monitor power port on the battery box.
  - Black 'screw in' cord from camera connects to the video camera port on the battery box.

1
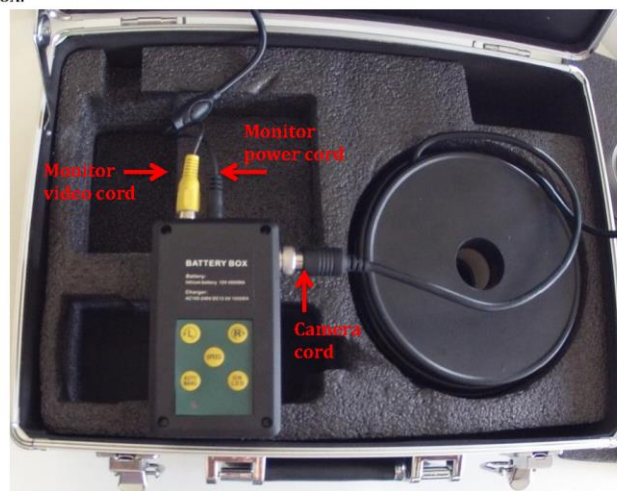
Step 3: Inserting and removing SD card

- In order to save and store footage from the camera the SD (memory) card needs to be inserted into the monitor (and possibly formatted).
- The SD card slot is located on the bottom left hand side of the monitor
- To access the SD card slot, the monitor needs to be released from the case by pressing the release clip on the left hand side of the monitor as shown below.
- To insert the SD card, gently push the card fully into the slot until it clicks in. To remove the card, press the card further into the slot until it clicks and the card will pop out.
- IMPORTANT: Ensure the monitor is turned off and disconnected form the battery when inserting or removing the SD card.
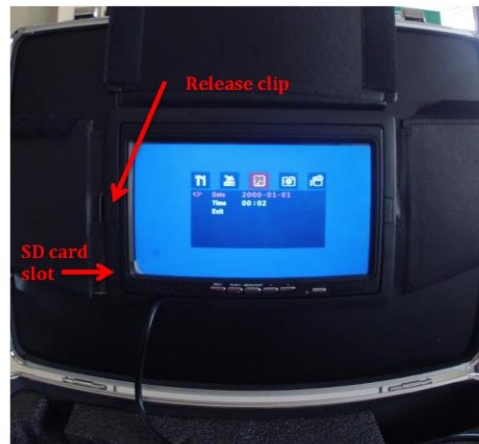
Step 4: Recording footage

- Once all cords are connected to the camera and the SD card is inserted the camera is ready to begin recording.
- Turn the battery on/off switch, located on the right hand side of the battery box, to ON.
- Then press the 'ON LED' button. The camera will turn on, automatically rotate within the casing turn and light up the leds. Pressing the button once more will turn the led lights off.
- To begin recording press the 'REC' button on the monitor. If the camera is recording, the top left of the monitor will display a flashing red "Record"
- To stop recording press the 'MENU/EXIT' button on the monitor
- If this is the first time recording with the SD card, you may be prompted to format the card. Use the " > <" buttons to select "Yes" followed by the "MENU/EXIT" key to format the card. Note: formatting the SD will erase any video footage on the card and should only be done the first time the card is used.

2

Step 5: Monitoring technique

- To provide consistency throughout the monitoring ensure the camera is rotating to the right at the slowest of the two rotation speed. The rotation speed and direction can be changed via the battery box as shown below.
- The 'ON LED' button turns led lights on. Use the led lights when required, but note using them will drain the battery.
- To ensure the proper depth for recording the camera should be lowered gently until it reaches the sea floor, and then raised roughly 1 metre off the bottom.
- Recording for each drop should begin as the camera enters the water and stopped as it is being retrieved to the boat. Each recording session should last approximately 15 minutes, and the date and time of the recording should be noted in the logbook provided.
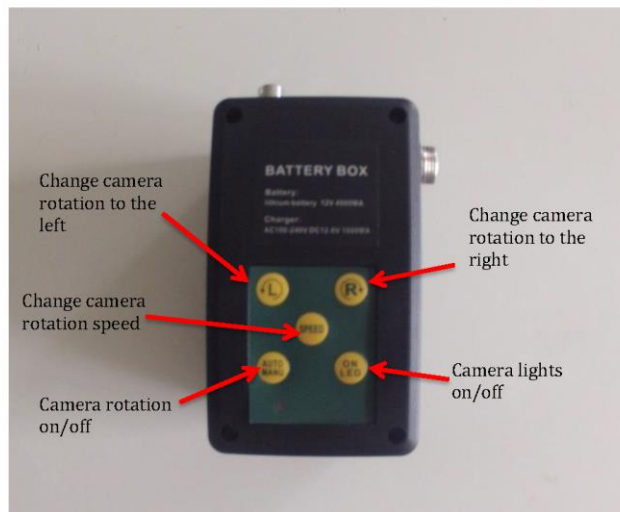
Step 6: Transferring data From the SD card to the USB

- Remove the SD card from the monitor following instructions in Step 3.
- Insert the SD card and a USB stick into the computer via the appropriate slots.
- Open the SD card folder on the computer and select all the video files on the card. Right click on the highlighted files and select '*Copy*' from the options
- Next, open the USB folder, which should be empty unless you have previously transferred video onto the USB. Right click inside the USB folder and select '*Paste*' from the options.
- Data will now begin transferring from the SD card to the USB. Time of transfer will depend on the number and length of videos recorded.
- Please only delete the files once they have been sent to Murdoch University and receipt has been acknowledged.

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Appendix 2.5. Pages from the Artificial Reef Logbook given to participants

Coordinates and locations for the Dunsborough artificial reefs

Thank you for your participation!

PLEASE NOTE PRIOR TO INITIAL CAMERA DROP: When deploying the cameras over the artificial reefs, it does not matter how long you wish to record for. HOWEVER we ask that you do each individual drop for the same amount of time as your first drop for that day. For example, if you record for 5 minutes on the first drop, please do not go for more or less than 5 minutes on each of the other drops, including on natural reef. Any confusion or questions, please ring (email) Howard Gill on **0406995036** (**h.gill@murdoch.edu.au**) or James Florisson on **0410320663** (James@recfishwest.org.au).

NW cluster
33 33.917
115 09.928

W cluster
33 33.961
115 09.943

SW Cluster
33 33.998
115 09.931

Cent Cluster
33 33.941
115 9.983

NE cluster
33 33.933
115 10.026

SE cluster
33 33.999
115 10.017

115°9.917E

115°10'E

33°33.917'S

33°34'S

Module clusters

Site boundary

0 10 20 40 m
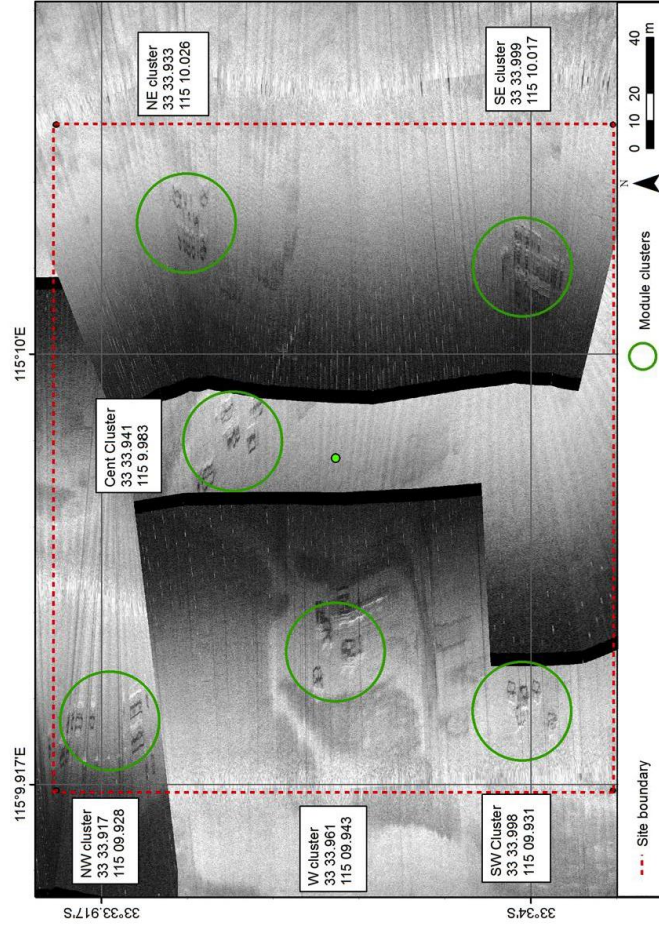
EXAMPLE SHEET (BUNBURY)
Date : November 7 2014

Reef module	Camera in (time: hrs/mins)	Camera out (time: hrs/mins)	Total (time: hrs/mins)	Species caught (cm), other comments:
NW cluster 33 18.454 115 35.859	10:25am	10:30am	5 minutes	No fish, few decent bites.
Cent cluster 33 18.479 115 35.898	10:52am	10:57am	5 minutes	2 pink snapper (35 and 40cms)
NE cluster 33 18.472 115 35.941	11:30am	11:35am	5 minutes	No bites, school of dolphins
W cluster 33 18.499 115 35.853	11:47am	11:52am	5 minutes	Strong gusts, no bites
SW cluster 33 18.545 115 35.857	12:04am	12:09am	5 minutes	1 Baldchin groper (62cm) released
SE cluster 33 18.529 115 35.895	12:31am	12:36am	5 minutes	2 Dhufish, 42 and 53cm, smaller released
Natural reef	01:35am	01:40am	5 minutes	Saw a bronze whaler, hooked 3 pink snapper, released 2 under 40cm. kept one at 76cm.

General comments, e.g., wind, swell current strength and direction etc: Gusty at times, wind changed direction from NE to SW around 10-15knots. Not much swell.

EXAMPLE SHEET (DUNSBOROUGH)
Date : November 7 2014.

Reef module	Camera in (time: hrs/mins)	Camera out (time: hrs/mins)	Total (time: hrs/mins)	Species caught (cm), other comments:
NW cluster 33 33.917 115 09.928	10:25am	10:30am	5 minutes	No fish, few decent bites.
Cent cluster 33 33.941 115 09.983	10:52am	10:57am	5 minutes	2 pink snapper (35 and 40cms)
NE cluster 33 33.933 115 10.026	11:30am	11:35am	5 minutes	No bites, school of dolphins
W cluster 33 33.961 115 09.943	11:47am	11:52am	5 minutes	Strong gusts, no bites
SW cluster 33 33.998 115 09.931	12:04am	12:09am	5 minutes	1 Baldchin groper (62cm) released
SE cluster 33 33.999 115 10.017	12:31am	12:36am	5 minutes	2 Dhufish, 42 and 53cm, smaller released
Natural reef	01:35am	01:40am	5 minutes	Saw a bronze whaler, hooked 3 pink snapper, released 2 under 40cm. kept one at 76cm.

General comments, e.g., wind, swell current strength and direction etc: Gusty at times, wind changed direction from NE to SW around 10-15knots. Not much swell.

Section 3: Critical review of the citizen science methodology and recommendations for future recreational fisher monitoring programs

Overview

This section of the report is based on Chapter 3 in James Florisson's honours thesis, which was completed in June 2015. It provides a critical discussion and review of each of the steps in the citizen science aspects of the methodology and a suite of recommendations. This review was undertake as only limited amount of data were collected in the pilot project (see Section 2), due to a number of environmental, operational and communicational issues.

Contacting and recruiting fishers

There are several ways in which the previous method of contacting and recruiting fishers could be enhanced, to try and recruit a higher and more engaged level of volunteers, *i.e.* citizen science champions. The scope of the promotion and advertising campaign should be greater and more thorough, to generate a larger pool of applicants from which the best candidates can be selected. Such a media campaign should include both traditional and non-traditional media elements.

Traditional media elements would be centred on a press release (from project partners including Recfishwest and Murdoch University), followed by active engagement with interested parties, such as print, audio and visual media outlets. Audio platforms such as ABC Southwest and talkback radio, *i.e.* 6PR, would be ideal for this promotion of the project and generating interest among potential volunteers. Targeted interviews could also be conducted on pre-existing fishing radio programs, such as John Curtis's fishing reports on ABC Radio, as these shows are well known amongst recreational fishers. Similarly, the filming and inclusion of segment about the project on a Western Australian fishing program (such as Fishing Western Australia) would reach a large audience and offer the chance to show visually, what potential volunteers could partake in. Articles could also be published in popular fishing magazines such as Western Angler and the West Australian Fishing Magazine. By combining with organisations such as Recfishwest and/or the Department of

Fisheries, the project could develop a media release with the Minister for Fisheries, which would increase the chances of TV stations doing a segment for the news. Advertisements and promotions would also be conducted through state newspapers such as The West Australian/Weekend West and Sunday Times and local newspapers, such as The Busselton-Dunsborough Mail, South Western Times, Bunbury Mail and the Bunbury Herald. Hardcopy advertisements and information sheets could also be put up on local bulletin and notice boards and given to tackle and camping stores within relatively close proximity to the artificial reefs and boat ramps.

Given the increasing influence of social media in recent years, any media campaign should include Facebook, Twitter and Instagram. There are a number of 'group', 'community' and 'pages' on the Facebook on which the project could be promoted *e.g.* Fishing Busselton and South West WA, Busso 4x4 Camping and Fishing, Busselton Fishing WA, Geographe Bay Yacht Club, Bunbury and Districts Power Boat and Fishing Club, Fishing Bunbury, Bunbury Fishing and Diving, Bunbury 4x4 and Fishing, Fishing South West WA, South West Artificial Reefs Community Facebook Page (Fig. 14) and Recfishwest. While, the current project did utilise Recfishwest's electronic newsletter (E-news), which is send to over 50,000 recreational fishers in Western Australia, the Department of Fisheries have a similar newsletter Catch! (See www.fish.wa.gov.au/fishing-and-aquaculture/recreational-fishing/catch-e-newsletter/Pages/default.aspx), which is emailed to all fishers who have any current fishing licence and those that subscribe separately, that could also be utilised.

The material released during the media campaigns should focus on the relatively simplistic nature of the data collection and the fact that fishers deploy the cameras during their normal fishing activity and thus don't have to do separate trips or decrease their fishing experience and/or opportunities. Secondly, the releases should seek to instil a level of ownership of the artificial reefs and stewardship for the marine resources in the area, to engage the volunteers and give them a sense of purpose for the project and its relevance for the local marine environment. Finally, the last message that could be included would be the social benefits from contributing to a citizen science projects.
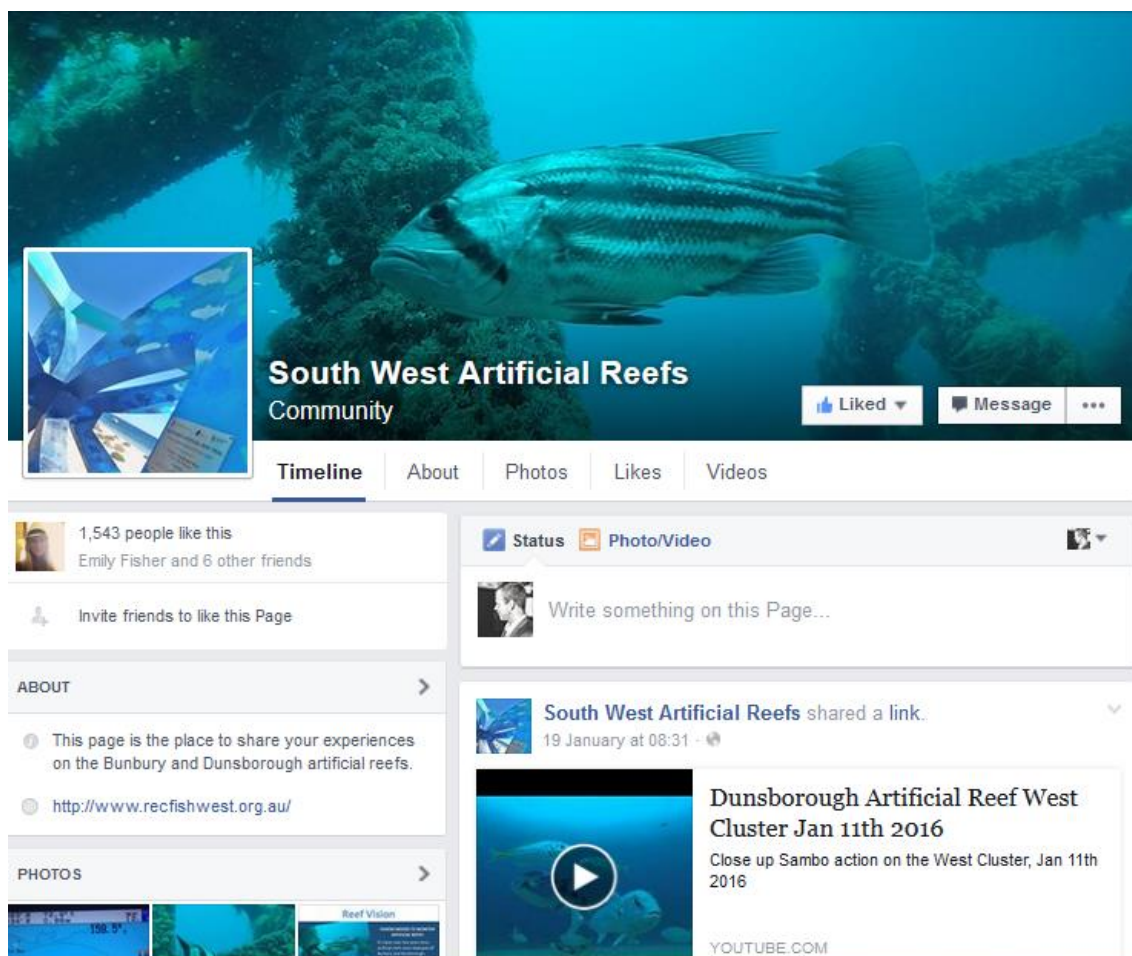
Fig. 14. The South West Artificial Reefs Community Facebook Page.

The main purpose of the media campaigns would be to recruit a sizable pool of volunteers. By acquiring a large suite of potential participants, filters can then be applied to select the most appropriate of those participants, *i.e.* champions. The greater the proportion of highly motivated and engaged volunteers the more data likely to be collected. A higher level of recruitment of volunteers on each of the artificial reefs may be beneficial, *i.e.* recruiting backup fishers, in the case that participants leave the project for any reason. Similarly, a continued source of volunteers, as the result of engagement through regular media releases or updates would also be beneficial if the volunteer attrition rate increased.

Camera trial

While the camera trial was successful and no doubt increased the level of engagement with the volunteers, there were a number of issues with the camera (see above). Essentially, the

quality and quantity of data gathered from the Sony CCD 700 TVL cameras was not statistically or scientifically adequate to test the hypothesis regarding the efficacy of citizen science monitoring of artificial reefs. In future, a trial involving a greater number of different types of camera should be conducted to ensure the quality of the video footage recorded is high enough for robust scientific analyses. Of course, this may led to a greater number of cameras being purchased for the trial and more expense, but better quality cameras would increase the value and accuracy of the project, noting too that this would increase the cost of the project.

One camera that should be trialled in future projects of a similar nature is the GoPro Hero 4[TM]. This camera was initially excluded from the selection process as it did not feature a live feed back to the boat. This was considered a critical part of the criteria as it would enable the fishers not to get the camera equipment snagged in the artificial reef modules. However, its likely GoPros attached to buoys will have a lesser chance of entanglement than the live feed cameras. This is because they aren't attached to a drifting boat and the only chance of entanglement is getting dropped directly on top of the modules. The chance of this happening is minimal, however can be rectified by retrieving the snagged equipment by pulling from a direction against the current or snagged position to unsnag the equipment. The GoPro camera is smaller and more user friendly, it also records better quality footage (than the other tested cameras) which can increase the accuracy in the results of the data analysis. For example, a comparison of 10 minutes of footage on the same artificial reef yielded 20 more species on the GoPro than the Sony CCD 700 TVL, as only fish at a close proximity could be accurately identified in the footage collected using the latter camera (J. Florisson unpublished data; see later).

Data collection

The process of data collection should be changed to make the project more applicable and desirable for fishers, to decrease the level of bias, to make it operationally and logistically simpler for volunteers and to collect better qualitative and quantitative data from the locations. Thus, a new methodology is proposed. Fishers will be asked to deploy a Baited Remote Underwater Video system (BRUV) in a set randomised zone near one cluster of

artificial reef modules for 40 minutes. A BRUV system uses either a single camera or two cameras (stereo-video to accurately measure distances) filming the area around a bait used to attract fish, the bait bag is placed close to the camera at a distance ranging between 0.5 and 1.5m (Ellis and DeMartini, 1995; Willis and Babcock, 2000; Heagney *et al.,* 2007; Mallet and Pelletier, 2014). BRUVs are most commonly used to survey variations in fish assemblages between sites, changes in assemblages over time (for example, diurnal variations) and interactions of species attracted to the baits and how these species interact with the surrounding ecosystem, thus overcoming previous limitations to these types of sampling. Each fisher will also deploy the same BRUV setup in an area of nearby natural reef for 20 minutes on the same day. This methodology follows that developed by Recfishwest in their monitoring program. The BRUV setup will consist of a GoPro Hero 4^{TM} camera on a pipe sled (filled with 5kg of lead), attached to a buoy with 35m of rope. Fishers will also be asked to use a similar logbook as in the initial phase, the only difference being the addition of new locational information including the grid and randomised deployment coordinates.

A lack of clear and consistent instructions, like those given in the initial phase of this project, can increase error and spatial and temporal sampling biases and result in selective data collection (Dickinson *et al.,* 2010), for example a volunteer only recording footage from one of the five clusters of artificial reef modules. To reduce spatial bias, it is recommended that volunteers will only be required to sample in one square on a grid, which encompasses a single artificial reef module cluster. The grid size will be standardised and each individual cell numbered and randomly assigned to a specific volunteer(s). This will reduce spatial bias by ensuring all reef clusters are sampled equally, theoretically at least. Likewise, if the same area of natural reef is monitored by all fishers, this would not be representative of natural reef fish assemblage composition due to lack of sampling location diversity. Natural reefs will be sampled for a period of 20 minutes, preferably on the same day as the artificial reefs. The location of the reef does not need to be known as the fishers would not feel comfortable in disclosing that information, and its unlikely fishers would monitor the same natural reefs as they would all likely have their own favourite areas of natural reef. Sampling sites should be representative of the surrounding region to be unbiased, if not, this can also introduce levels of bias to citizen science research. If the habitat types surrounding sampling

sites are not representative of the larger regional landscape, then differences in species occurrences or abundance may reflect spatial sampling bias rather than true geographic differences in population size (Lawler and O'Connor, 2004; Niemuth *et al.,* 2007).

Temporal biases caused by lack of standardisation across sampling occasions during the current project were caused by unseasonal bad weather and timing delays with camera importation and variability from changes to instructions and guidelines. To mitigate this, participants in the future will be required to deploy the BRUV for at least 20 minutes, but no longer than 30 minutes (including a standard error time period of ± 10%) once a month in their set grid cells. The recording time of at least 20 minutes will allow the bait plume to travel far enough and attract a sufficient number of species for robust statistical analyses. There will not be a restriction on the number of replicate recordings collected in a grid cell in each month. The reason for only having a minimum level of replication is that the stricter the instructions the greater the chance of losing volunteer interest and participation. The presence of this minimum level of participation is that Dickinson *et al.,* (2010) found that 'when programs have no prerequisites for minimum effort (that is, any type of effort is allowed), samples may be highly biased, resulting in inaccurate data collection.

With the original data collection method, volunteers are required to stay in the vicinity of their camera while filming, however, the BRUV may be attached to a buoy with a rope, rather than the camera being attached to the monitor on the boat (as with the original method). This would allow the volunteer to leave the immediate drop zone, and therefore they can actively fish for the period while the camera is deployed. This is considered attractive to the participants as they can actively fish and target specific species, rather than focusing on a small monitor screen for 15 minutes while drifting (as they did in the initial phase). A stationary benthic BRUV attached to the buoy is also likely to have a smaller chance of being snagged in the artificial reef module. This is because it isn't moving and drifting with a boat, instead being stationary on the ocean floor, thus mitigating risk in relation to drifting into modules. Although the use of bait with a BRUV could be viewed as a selective attractant increasing bias, all animals passing through the field of view, in response to the effect of bait or not, can be recorded (Armstrong *et al.,* 1992). The lack of size

selection, and the powerful sampling replication afforded by multicamera (BRUV) units avoids false negatives (Tyre *et al.,* 2003) and allows standardised sampling at any depth, time of day and type of benthic topography (Cappo *et al.,* 2007).

General volunteer management

Volunteer management is an important facet of any citizen science project. The benefits of correct volunteer management include low attrition rates, thus increasing cost efficiency by not having to promote and advertise for more volunteers, increased quality in the data set by having engaged and passionate volunteers and smoother communication and volunteer engagement throughout the project. A good relationship between volunteers and researchers can also give the volunteer the ability to discuss the research and wider ecological issues with scientists, experience something unique, see animals and habitats they didn't know existed in the area, master new skills and develop an appreciation of the effort involved in collecting ecological data (Wilson and Godinho, 2013).

It is recommended that the way volunteers were managed in the first phase of the project be altered to achieve more desirable project results and better relationships with volunteers. One way to develop a better rapport between the volunteers and the project managers would be for communication to occur at least once a week by phone and once a month in person (depending on project funding, this option may not be viable, or instead could be undertaken by a 'champion' volunteer, the most engaged and effective participant with good communication skills). The purpose of the phone call would be to check for any change in attitude from the volunteer towards the project, check that the equipment is functioning correctly and answer any questions the participants have, as well as disseminating results back to the fishers. Volunteers should be seen once a month by a project manager or engagement officer (or champion volunteer) to discuss aspects of the project and any issues and to collect copies of the video recordings. Such a meeting would eliminate the data collection and transport issues encountered during this study. It also shows the volunteers that the coordinators and engaged and involved in the project and presents an opportunity for the two way dissemination of information between the two parties as well as an opportunity for the presentation of any project findings to the

participants. If the costs associated with this level of engagement are beyond the scope of the project, face to face data collection and engagement could be completed by project partners in regional governmental offices, such as staff from the Department of Fisheries who have offices in both Bunbury and Busselton. The contact should be at a standardised time for each fisher, and fishers should be able to have phone contact during office hours and email contact outside office hours. This would potentially foster positive engagement, for the volunteers to know that they have this level of support.

To make it easier for the fishers and to reduce error in the data collection, fishers would be given a clear, concise and simple set of standardised written instructions. The instructions would also have contact details for project managers and local safety information. These instructions would be have large pictures to show the steps, large text and be water proof so that they can be utilised while monitoring. Four times a year there would also be a gathering of volunteers and project managers. This would aim to increase relationships and the quality of the volunteer network, to discuss the project and for project managers to disseminate project results to that date. The gathering could also be extended to involved organisations such as Recfishwest and the Department of Fisheries as well as the general public. This could help increase attendance, sustain interest and engage the general public to give the local community a sense of stewardship over the project and the artificial reefs. It's also another opportunity for volunteers to discuss any issues they are encountering with the project structure and equipment. A short film of the best segments of footage captured from the cameras would also be shown to keep volunteers interested, engaged and passionate about the project. After the project there would also be several other events, these would include a community seminar to discuss the findings to stake holders, local fishers, end users and the general public. A post project survey or interview would also be conducted with volunteers to gauge attitudinal variation at the end and throughout the project, what skills and knowledge they obtained and how they felt the project went. The purpose of this exercise would be analyse social and emotional variation in the volunteers to help with future citizen science projects, and to see if the volunteers would be interested in contributing to similar projects in the future.

Recommendations for future research

While this citizen science project yielded only small quantities of data (see Section 2), this was most likely due to unseasonal weather patterns, lack of volunteer communication and logistic difficulties with importing the cameras. Although this limited success could be interpreted as a setback in the case for using recreational fishers as cost-effective means to monitor artificial reefs, it's important to consider that this project is a pilot study, which had an evolving methodology. The following dot points represent key considerations that should be incorporated into any future project to employ citizen science to monitor artificial reefs.

- The method of contacting and recruiting volunteers should be enhanced, by using traditional and social media, with a greater scope for promotion and advertising to recruit a large quantity of better quality volunteers.

- Smaller GoPro cameras should be utilised on BRUV structures to maximise the quality and quantity of data as well as simplify the equipment and procedure for fishers.

- Clear and concise instructions and monitoring protocols will decrease volunteer attrition rates as well as spatial and temporal biases, while increasing the accuracy and quality of the footage.

- Positive outcomes of correct volunteer management can be optimised by adequate communication and engagement with the volunteers.

Introduction

As the human population increases, so does the range and extent of deleterious anthropogenic activities and associated perturbations. As a result, there is a growing need to monitor these influences to ensure ecosystem sustainability. The collection of robust scientific data by government organisations and tertiary educational institutions can be expensive and prohibitive. Thus, for example, the cost of monitoring several fisheries is more than the income the government receives from these fisheries (Leyland Campbell, Recfishwest, pers. comm.). In an effort to reduce costs and engage the general public many organisations are turning to citizen science. Citizen science is defined by Open Scientist (2011) as "*the systematic collection and analysis of data; development of technology; testing of natural phenomena and the dissemination of these activities by amateur scientists, the public or researchers on a primary avocational basis*". This term encompasses a variety of aspects of volunteering in scientific research including community-based monitoring, community science and volunteer monitoring (Sbrocchi, 2013). The different types, research aims, capabilities and opportunities in citizen science are vast and varied, for example: counting numbers of stars in distant galaxies, determining the timing of flowering events, monitoring the health of coral reefs and recording information on bird migrations (Gollan, 2013). The success of many of these projects has resulted in decision makers and non-government organisations increasing their use of citizen volunteers to enhance their ability to monitor and manage natural resources, track species at risk and conserve protected areas (Conrad and Hitchey, 2011).

Citizen science is not a modern facet of science. In the past many scientists have conducted research, with their studies being avocational or unpaid and thus essentially being a form of citizen science. For example, Benjamin Franklin was a printer, diplomat and politician and Charles Darwin sailed on *HMS Beagle* as an unpaid companion to Captain Robert FitzRoy, rather than as a professional naturalist (Silverton, 2009). The restrictions facing modern research (such as costs, funding cuts and collecting large amounts of data across large spatial and temporal ranges) are fuelling exponential growth in the area of citizen science.

Silverton (2009) and Baltais (2013) both commented that as of January 2009, the ISI Web of Knowledge database only contained 56 citizen science research articles, with 80% being published in the last 5 years. However, there are hundreds of scientific publications investigating patterns and processes that are based upon data gathered by citizen scientists. As of April 2015, the ISI Web of Science contained 355 citizen science articles, 299 more than in 2009, however it's likely that there are many more articles included in the collection based on data procured through citizen science. Though citizen science can be applied to most scientific disciplines (from drug trials in medicine to observations in astronomy), it is also commonly used in the analyses of ecological patterns and processes. Many ecological processes occur over large spatial and temporal scales, including migration patterns, disease spread and species range changes. Gathering sufficient data on such processes can be difficult using traditional research methods, particularly given limitations in time and funds (Bonney *et al.,* 2009; Dickinson *et al.,* 2010; Tulloch *et al.,* 2013). Recruiting volunteers from the general public into citizen science projects potentially offers a low cost way to expand the reach and frequency of data collection, although this can be dependent on context (Lambert, 2014). This background to citizen science aims to critically review the benefits and limitations resultant of using citizen science for research purposes. It also aims to assess the range of types of citizen science projects, as well as document the citizen science projects that have been or are being conducted in aquatic environments in Western Australia.

Benefits

The number of citizen science projects is expanding both in Australia and throughout the world due to the benefits it provides both to the project managers (such as cost efficiency) and the participants (such as social values). The major benefit of citizen science to the project managers and/or researchers is its cost effectiveness and efficiency and increasing stakeholder capacity (Wiersma, 2010; Sullivan *et al.,* 2014). The use of volunteers helps reduce the overall cost of the research by i) reducing fieldwork and/or data collection costs, ii) reducing staffing costs and iii), by reducing the above costs, and may also reduce the cost of indirect or 'hidden' charges such as oncosts and overheads.

Volunteers can collect data over large spatial scales, creating large longitudinal data sets which have led to new quantitative approaches to emerging questions about the distribution and abundance of organisms across space and time (Dickinson *et al.,* 2010). One of the best examples to illustrate the power of citizen science in obtaining large amounts of ecological data is eBird. This project, which was established by the Cornell Lab of Ornithology in 2002, collects information on bird distribution and abundance through the presence or absence of species and through checklist data. Through a combination of community engagement and partnerships, eBird has created a global network of volunteers who submit an average of three million observations per month (Lambert, 2014; Sullivan *et al.,* 2014).

Similarly, in Australia, the Range Extension Database and Mapping Project (REDMAP) was developed and launched in 2009. This is a web-based citizen science initiative where community members submit photographic observations of species found outside of their native range, which are then verified by expert scientists (Pecl *et al.,* 2014). REDMAP was created after it was identified that range shifts globally, are one of the most frequently reported impacts of climate change (Pecl *et al.,* 2014). Detailed examination of whole assemblages or ecosystems suggest that between 20% and 85% of species are shifting where they live in response to changes in temperature (Chen *et al.,* 2011; Wernberg *et al.,* 2011). To date, REDMAP has had over 1,060 reports of species outside of their previously known and recorded ranges, verified by over 80 expert scientists (Pecl *et al.,* 2014).

The cost-saving and efficiency of successful citizen science projects can be very large, for example, two studies by Dickinson *et al.* (2010) and Sullivan *et al.* (2014) both analysed the cost effectiveness of 'Project Feeder Watch', to find it was extremely cost effective at collecting large amounts of data. Dickinson *et al.* (2010) suggests that the Cornell Lab's Project Feeder Watch contributes $3 million per year worth of observer effort, and Sullivan *et al.* (2014) noted that the cost per datum on eBird in 2008 was only 3 cents (Wiersma, 2010). It's likely that in most citizen science projects, the value of the project increases with the number of participants and the amount of data provided by those people (depending on the context of the project). It is important to consider that, while citizen science can save

financial resources in a number of different facets, they do require initial and continued expenditure.

Many citizen science projects have developed data platforms and portals such as websites and smart-phone applications that are user friendly and easy to input large amounts of data. The design and development of such software can be expensive, however, having the end users enter the data saves costs in the long term by preventing the data being manually entered by researchers and for the ability for large amounts of free data, *i.e.* numbers, photographs and videos, to be uploaded. Moreover, the development of software, *e.g.* a smart-phone application, may increase the accuracy of the resultant data over paper recording, *e.g.* a logbook, by i) promoting the end user to look at potentially erroneous data and, where necessary, modify and ii) by standardising data by forcing the end user to choose from a small list of options and iii) automated data from the device, *e.g.* location / time data, rather than data entered by the end user (Kerry Trayler, Swan River Trust, pers. comm.).

While the costs of citizen science surveys can be high, Goldstein *et al.* (2014) found that this method was more cost effective and efficient on a per detection basis for the purpose of recording the presence of the species being studied. These authors stated that "*in the face of increasing ecological and economical costs of biological invasions we recommend straight forward citizen science surveys, over indirect field surveys, to managers and researchers seeking to efficiently track progressing invasions of readily observable animals cost-effectively*".

One of the less obvious benefits of citizen science is fostering collaboration between organisations to share data, funding, resources, volunteers and reach a wider audience by promotion through alternative networks. One example of this multi-organisational collaboration is Prawn Watch in Western Australia (Trayler *et al.*, 2015). Prawn Watch receives shared funding from the Swan River Trust (WA government agency that manages the Swan Canning Riverpark) and Recfishwest (WA peak body for recreational fishing), shares data with Murdoch University, Swan River Trust and Recfishwest and has a large range of alternative networks through Murdoch University, Recfishwest, Department of

Fisheries and the Swan River Trust (Leyland Campbell, Recfishwest, pers. comm.). Another active example of this was the creation of the Reef Citizen Science Scoping Study by the Great Barrier Reef Foundation. This enhanced collaboration between citizen science groups across the reef, promoted and raised the credibility of citizen science and optimised the use of citizen science data by scientists, reef managers, conservation groups and communities (Great Barrier Reef Marine Park Authority, 2013). A further benefit is that due to large temporal and spatial ranges combined with observer effort, citizen science appears to be particularly effective at finding disappearing native species, rare organisms, new organisms and invasive organisms. This is demonstrated in many studies, two examples include the Lost Ladybug Project (lostladybug.org) finding extremely rare native ladybugs by the public analysing ladybug species compositions (Dickinson *et al.,* 2010) and FeralScan (www.feralscan.org.au) in which the public map feral animal sightings in their area, which is an Australian initiative that now has over 25,000 community recordings (Lambert 2014).

The major benefit of citizen science from the citizen's perspective is the social values of volunteer involvement. Volunteers, by engaging in the project, are able to become a 'scientist' for a certain period of time helping to contribute and collect data and samples. A citizen science based project in Melbourne, designed to describe the distribution and habitat preferences of bats, found that the benefits to volunteers included i) discussing research and wider conservation issues with scientists, ii) experiencing something unique, *e.g.* seeing animals and habitats that they didn't know existed in the area, iii) gaining an understanding and appreciation of the issues facing the organisms and their importance in ecosystems as well as mastering new skills and iv) developing an appreciation of the effort involved in collecting ecological data (Wilson and Godinho, 2013).

Involvement in citizen science programs can promote active engagement, encourage pro-environmental/ecological attitudes and behaviours and increase the public's scientific literacy, awareness of issues and ecological knowledge (Lambert, 2014). The evaluation by Jordan *et al.* (2011), of an invasive plant monitoring project determined that volunteers' knowledge of invasive plants increased on average by 24%. Similarly, following engagement into a prawn monitoring project, participants' knowledge of the rules of the recreational

fishery increased on average from 50% to 91% (Tweedley *et al.,* 2014, 2015b; Trayler *et al.,* 2015). Furthermore, volunteers involved in the 'Monarch Butterfly (*Danaus plexippus*) Larvae Monitoring Program' reported that the project had led them to take an active role in habitat improvement (Oberhauser and Prysby, 2008; Lambert, 2014). Other social benefits of engagement in citizen science projects include improved communication leading to shared goals between diverse stakeholder groups and increased engagement and participation in local issues and community development, all of which influence policy-makers (Fernandez-Gimenez *et al.,* 2008; Conrad and Hilchey, 2011; Lambert, 2014).

Limitations

Although citizen science has many benefits, it also has several limitations. These can be broken into three main groups, namely organisational issues (including volunteer participation issues), data collection issues and data use issues. Conrad and Hilchey (2011) stated that many of the challenges for community based monitoring occur at the organisational level.

Organisational issues include occupational health and safety (Baltais, 2013), funding (Whitelaw *et al.,* 2003), information access challenges (Milne *et al.,* 2006) and a lack of volunteer interest (Conrad and Daoust, 2008). As legislation and regulations are consistently changing, especially in relation to occupational health and safety and insurance, the reviewing of policies and insurances needs to be continually undertaken by organisations to ensure adequate compliance. Legislation is an issue as if it's not adhered to, projects can lose funding. The Wildlife Preservation Society of Queensland (a major citizen science organisation) stated that insurance and workplace health and safety are emerging concerns and 'many contributory and collaborative projects offer no insurance to those projects through their own organisations' (Baltais, 2013).

Another major operational issue is funding. Funding issues vary between organisations, projects, locations and funding priorities, however, they can have dire consequences on citizen science projects. This is particularly problematic in relation to the timeframe around

funding. For example, long-term projects are more susceptible to funding variations and issues, especially when the projects are funded by multiple short term grants from different organisations. Projects that rely on short term grants can present a barrier to long term sustainability (Crall *et al.,* 2010). While corporate sponsorship is an option, active searching for funding opportunities, good communication techniques and enhancing relationships with funding bodies could potentially help alleviate funding issues, instead of depending on corporate sponsorship.

A final organisational issue is generating, managing and maintaining volunteers and volunteer interest. Generating and maintaining volunteer interest is a key challenge of citizen science and it is especially difficult as it's hard to establish clear links between citizen science projects and their influence on participant behaviour and attitudes (Lambert, 2014). Managing volunteers and volunteer interest requires qualified staff, usually a volunteer coordinator and can also be helped by having well established user friendly technology. 'Volunteers motivations are complex, change throughout the project life cycle and are strongly affected by personal interests and are thus an issue for citizen science project management' (Rotman *et al.,* 2012).

The second major limitation of citizen science is issues with the collection of data. These issues include error and bias due to variation in observer quality and/or participant objectivity and bias from variation in sampling effort over time and space. Many of the error and bias are due to the fact that the skills of citizen scientists are often, as expected, much lower than those of research staff. Citizen scientists vary in ability, experience and the type of training they have been exposed too (Dickinson *et al.,* 2010). These authors reported that, a lack of training can increase the error and bias in the misidentification of species, incorrect reporting and selective data collection. Age is also an important factor to consider. For example, a study undertaken by Delaney *et al.* (2008) found that 80% and 90% of students in, respectively, grades 3 (8-9 years old) and 7 (12-13 years old) had the ability to differentiate between two species of invasive crabs, while older volunteers, who had at least two years of university education, were able to correctly identify both species and the age of the crabs with a success rate of 100%. For many projects, most of the variation in

observer ability is due to new participants, affecting short and medium term projects. For example, a short term project will likely have a larger variation in observer skill than a long term project as new participants in a short term project do not have a long timeframe in which to learn, whereas, conversely, observers in longer projects have more time to be trained and learn accurate and consistent methods to conducting observations in longer projects. Several studies of volunteer based monitoring programs conducted over many years have documented 'learner' or 'first year' effects, where observers become better data collectors over time (Jiguet, 2009; Shmeller *et al.,* 2009; Dickinson *et al.,* 2010). An example of this can be seen in the French Breeding Bird Survey, in which the average increase in the detected abundance of bird species between the first and all subsequent years of volunteer participation was 4.3% (Jiguet, 2009).

Bias from variation in sampling effort over time and space is a common issue in citizen science and varies with method, effort, species and environments sampled. Bias caused from variation in spatial and temporal sampling effort usually stems from lack of standardization. To limit bias, most scientific projects have strict standardisation protocols in relation to intervals, repeated tests, guidelines and benchmarks. However, when these protocols are too demanding or strict, there is a chance of loss of volunteer participation and interest. For example, it might be easy to recruit volunteers to record data from wilderness environments during warm, dry summer months, but less so during colder, wetter months or vice versa depending on the climate of the environment. This can be minimised, to some extent, by having a large number of participants and using some of the more experienced volunteers, *i.e.* champions, to undertake more intensive roles.

The less control that programs have over effort, the greater the potential for bias in the resultant data, however, as specified before, a high level of control and standardisation can severely impact volunteer participation (Dickinson *et al.,* 2010). These authors considered that the data collected by citizen science programs that have no prerequisites for the minimum level of sampling effort required may be highly biased. For example, in a program where participants are asked to record species they see in a particular area, this can result in the over-reporting of rare species, under-reporting of common species, and failure to report

repeated sightings, because they are not deemed as 'interesting' by the observer. Moreover, some volunteers even stop reporting when there are no interesting species recorded, this can lead to analyses and conclusions that reflect variation in effort more than actual biological patterns and processes (Dickinson *et al.,* 2010). Thus, projects with no framework for standardizing effort may not necessarily present inaccurate data, but varying numbers of participants, count durations and inclusion of effort measurable, needs to be taken into account in the analyses of the project. The amount of effort expended should be considered an important variable that should be accounted for in analysis (Link and Sauer, 1999).

Spatial biases in sampling effort may also occur when resulting data are not representative of the habitat/location, the sampling method is not standardization and/or when large data sets are not filtered appropriately (Dickinson *et al.,* 2010). If the habitat types surrounding sampling sites are not representative of the larger regional landscape, then differences in species occurrences or abundance may reflect spatial sampling bias rather than true geographic differences in population size (Lawler and O'Connor, 2004; Niemuth *et al.,* 2007). This can be accounted for by sampling in more locations, with more replications to try and increase the level of representation to the larger landscape. Irrespective of sampling methods, sampling sites should be representative of the surrounding region to be unbiased, if not, this can also introduce levels of bias to citizen science research. When managing large citizen science data sets (such as lots of recordings, samples or observations from ranging temporal and spatial scales), filters are extremely beneficial. Filters are a tool to select or omit specific data out of a larger data set and can be used in the data entry process to ensure all required protocol information is accurately entered as well as to extract specific data from large general data sets, post data entry (Hochachka *et al.,* 2012). An example of filter use is in Project FeederWatch in which automated filters are used to identify potential errors in bird observations submitted by participants by the use of historical data and if a species had not been reported by at least 4% of participants in the last season (Bonter and Cooper, 2012). Some projects like eBird, get people to report for all species, but code birds that aren't targeted as absent, in presence-absence studies, thus just extracting data on the specific target species while still collecting a broad range of data (Bonney *et al.,* 2009).

The final category of citizen science issues are those relating to data usage. These issues are centred on the perceived lack of quality and distrust of citizen science data as well as access rights to that data. In light of the issues discussed above, data collected by citizen scientists may not be taken seriously by decision makers and scientists (Conrad and Daoust, 2008 and Wiggins and Crowston, 2011). Thus, many researchers can potentially find that their data is not considered for use in the decision making process or published in scientific peer-reviewed journals, either due to data collection concerns or difficulty getting their data to the appropriate decision-maker or journal (Milne *et al.,* 2006; Conrad and Daoust, 2008). The values of certain citizen science groups and volunteers may also impact data use, for example, purposely targeting or avoiding certain species to get a desirable outcome. These concerns led to the US Congress, in 1994, calling for the National Biological Survey to exclude data gathered by volunteers because of the belief that their 'environmentalist agenda' would lead to biased data collection (Root and Alpert, 1994; Conrad and Hilchey, 2011). Citizen science projects may also encounter issues around intellectual property rights and data ownership policies. For example, Only 64% of the invasive species monitoring programs reviewed by Crall *et al* (2010), generated species distribution maps and only 23% made their data publically available, due to concerns about privacy and data sensitivity'(Lambert, 2014). This is likely due to some citizen science initiates not being adequately shared or analysed with other groups, as few projects inform volunteers about intellectual property rights or have clear data ownership policies (European Commission, 2014).

Types of citizen science

There are many different classifications and types of citizen science projects. These projects can vary from small scale localised studies (PrawnWatch) to global research projects (eBird). While citizen science has the potential to contribute to a plethora of research projects, it is best suited to studies where, i) data collection is labour intensive and involves fieldwork, ii) quantitative data are required, iii) the spatial and/or temporal extents are broad, iv) the methodology is well designed, simple and easy to execute, v) guidance material and/or

professional assistance are available and vi) data submission can be done electronically (Gommerman and Monroe, 2012).

Citizen science is rapidly becoming more popular with people taking part in projects all over the world. Volunteers can now participate in projects on population ecology, conservation biology, ecological restoration, climate change and various types of monitoring. Throughout the rapid expansion of citizen science's popularity, a single universal classification for different typologies has not yet evolved, instead having various classification systems for project types. Dickinson *et al.,* (2010), puts projects into organismal monitoring; classifying projects by taxonomic group, environmental monitoring; classifying projects by environmental variables and non-ecological projects which classify projects by their field of inquiry. Although approaches are diverse, two commonly accepted typologies are those proposed by Bonney *et al.,* (2009) and Wiggins and Crowston (2011). Bonney *et al.* (2009) proposed a typology that classifies projects according to their degree of public participation, and Wiggins and Crowston (2011) classifies projects based on their goals (Lambert*,* 2014). These two typologies are provided in Table A1 with examples.

Table A1. Project Typologies, modified from Lambert (2014).

Bonney *et al.* (2009 typology)			
Type	Description	Example Project	Purpose
Contributory	Designed by scientists, volunteers primarily contribute data	ClimateWatch	Monitoring phenology (seasonal life cycles)
Collaborative	Designed by scientists, volunteers contribute data, refine project design, analyse data, disseminate findings	Coastal Walkabout	Monitoring coastal biodiversity
Co-created projects	Co-designed by scientists and volunteers	Streamwatch	Monitoring local stream health
Wiggins and Crowston (2011) typology			
Type	Description	Example Project	Purpose
Action	Citizens collaborate with scientists in action research approaches, often to address local environmental concerns	Sherman's Creek Conservation Association	Protecting local creek
Conservation	Focus on protecting and managing natural resources whilst educating the general public	Invasive Plant Atlas of New England	Mapping invasive plants
Investigation	Focus on testing specific research hypotheses	eBird	Collecting bird observations
Virtual	May have similar goals, but all activities are carried out remotely, using online platforms	Explore the Sea Floor	Classifying marine organisms
Education Projects	Primarily conducted to achieve educational goals (scientific rigour may be less important)	Biodiversity snapshots	Biodiversity surveys

Aquatic citizen science projects in Western Australia

With its ability to provide large data sets on a range of variables cost effectively and inform and engage the public, numerous citizen science projects been employed in a Western Australia. These projects vary from tagging, biological donations, logbooks and monitoring and identifying movements, patterns and range shifts and are covered in the following section.

Tagging

There are various citizen science projects that use tagging as a research tool. Tagging fish are part of what is known as the capture-mark-recapture sampling method (CMR). In CMR experiments, animals are captured, marked, released and recaptured many times by repeat sampling (Pradel, 1996), In WA, recreational fishers tag fish as well as submit recapture data, such as location, length and the health of the specimen, usually in logbooks (see 3.2.5.3 *Logbooks and monitoring*). Key species are tagged all over the state for various projects such as Dhufish (*Glaucosoma herbraicum*), Baldchin Groper (*Choerodon rubescens*), Pink Snapper (*Chrysophrys auratus*), Breaksea Cod (*Epinephelides armatus*) and Samson Fish (*Seriola hippos*) by Australian National Sportfishing Association WA, Westag and Infofish Australia. The Department of Fisheries also tags Tailor (*Pomatomus saltatrix*), Pink snapper, Samson Fish and blue swimmer crabs (*Portunus armatus*). Western Australian universities, gamefishing associations and fishing clubs also tag many species. Different species get tagged for varying purposes, for example, pelagic and migrating species such as Southern Bluefin Tuna (*Thunnus maccoyii*) are tagged to discover where the fish migrates to, its varying distributions and its growth if recaptured. All these species and many more are tagged and caught by citizen scientists in WA, with data going towards research on recruitment, movement and migration, stock structure, monitoring and mortality. The Department of Fisheries have created a tagging iPhone application for reporting recaptures which helps citizen science through being a user friendly basic vessel to transport tagging data.

Biological donations

Citizen scientists can also assist by helping sampling or donating their catch (or part of it). One of the largest and most successful of these projects in WA is known as Send Us Your Skeleton (SUYS), ran by the Department of Fisheries. SUYS asks recreational fishers to voluntarily donate fish frames belonging to a number of key recreational species such as: Herring (*Arripis georgianus*), Dhufish, Baldchin Groper, Pink Snapper and Bight Redfish (*Centroberyx gerrardi*) from their catch to allow biological data extraction by scientists to produce age structures and conduct stock assessment analyses (Fairclough *et al.,* 2014).

Some examples of the biological data extractable includes dietary analyses from the fish guts, sexual analyses from the gonads, genetic analyses from tissue samples and ageing from the otoliths (structure in the inner ear) or vertebrae of species (Fairclough *et al.,* 2014). A multi-organisational project looking at restocking western school prawns into the Swan River Estuary has a citizen science component known as PrawnWatch. PrawnWatch has 135 volunteer citizen scientists (as of October 2014) that have participated in the broodstock collection events and contributed to the collection of 580 gravid females that produced 12.5 million eggs (Tweedley *et al.,* 2015b). A project run by Murdoch University in the south-west of WA is based on fishers providing squid samples, has had over 3152 samples collected with over 28% coming from recreational fishers. The samples are aged and data being collected will contribute to biological information, as well as a stock assessment on this species. Biological samples are also taken by many recreational fishers when catching game fish such as Tuna and Mackerel (Scombridae), Dolphinfish (Coryphaenidae) and Billfishes (Xiphiidae and Istiophoridae) to help with research. Fin clips and tissue samples (in some cases used when collecting samples but releasing fish after) can be used for DNA and genetic analyses, hard parts such as otoliths and Chondrichthyes (sharks and rays) vertebrae can be used for ageing, guts can be used dietary and internal parasite analyses and gonads can be used to determine sex and sexual maturity (Pepperell Research and Consulting, 2010). These differing biological samples can be used in studies to help analyse local and global genetics and distributions, biology, parasite analyses and ecology of these species (Pepperell Research and Consulting, 2010).

Logbooks and monitoring

Another method for obtaining data from citizen scientists/recreational fishers is through the adoption of survey techniques or a fishing logbook. Surveys involve verbal contact with the participant and asking them a range of questions to collect data, while logbooks involve fishers themselves recording information on their catches to later be submitted to an organisation for analyses. A project by the Western Australian Department of Fisheries on blue swimmer crabs has over 100 recreational fisher volunteers issue logbooks to measure the size, sex and distribution of the crabs in the Swan-Canning, Peel-Harvey and Leschenault

estuaries. The Western Australian Department of Fisheries also administers the Research Angler Program which involves anglers filing out logbooks to provide data on a whole range of variables on a large amount of recreational species. These variables can include population structure, movement, growth, mortality, abundance and diversity on species such as Tailor (*Pomatomus saltatrix*), Herring (*Arripis georgianus*), Squid (Order *Teurhoidea*), Dhufish (*Glaucosoma herbraicum*), Baldchin Groper (*Choerodon rubescens*), Pink Snapper (*Chrysophrys auratus*) and many others (Department of Fisheries, 2012c). The Department of Fisheries also conduct a survey known as the isurvey, where volunteers keep a 12 month diary for a biennial survey of recreational catch and effort. One of the other purposes of Prawn Watch is also monitoring. Prawn catches are monitored and data is collected through a mobile phone application to analyse location information, type and number of prawns, gravidity of the prawns and bycatch information (Trayler *et al*., 2015).

One of the more common types of citizen science approaches adopted as a marine research tool is monitoring. Monitoring generally means observing a system or species and recording any variability that is observed in the system or species. There are currently a number of marine based citizen science monitoring projects that use recreational fishers as volunteers. Stocked and tagged fish are monitored to ensure the health of the stock. This is currently being done by many different organisations in different projects such as monitoring tagged Black Bream (*Acanthopagrus butcheri*) in rivers and estuaries such as in the Peel-Harvey and Swan-Canning systems. Restocked bream are also monitored to assess the successfulness of the stocking activity in systems such as the Blackwood River Estuary in south-western WA. Fishers are asked to report the lengths of these restocked species to assess their growth rate as well as the number caught, to assess their size class and their contribution to the overall population. Restocked fish can be differentiated from natural cohorts as they generally have stained otoliths. Staining mediums such as alizarin complexone are used for staining the otoliths, initiated by emerging hatchery-reared juveniles in the stain, the stained otolith is still visible to the naked eye years later (Jenkins *et al.,* 2006). Restocked Mulloway (*Argyrosomus japonicus*) and Barramundi (*Lates calcarifer*) are also monitored using the same method in the west coast and Kimberley regions of WA respectively.

Monitoring can also be used to analyse the effects and successfulness of habitat enhancement structures such as FADs (Fish Attraction Devices) and artificial reefs.

Identifying movements, patterns and range shifts

Citizen scientists also play a key role in identifying movements, patterns and range shifts of migratory, invasive, rare and common species. The Department of Fisheries have the Pestwatch Application, in which hundreds of citizen scientists have reported sightings of invasive marine species such as the Asian date mussel (*Musculista senhousia*), northern pacific sea star (*Asterias amurensis*) and European fan worm (*Sabella spallanzanii*) and freshwater species such as Redfin perch (*Perca fluviatilis*), Carp (*Cyprinus carpio*) and Mosquitofish (*Gambusia holbrooki*) (Department of Fisheries, 2012b). Aquatic pests, aquatic diseases (including fish kills) and illegal fishing activities are all reportable to FISHWATCH on the phone number: 1800 815 507. Citizen scientists can also log species sightings when the species are rare or not usually found in the area to show movements, patterns and distribution shifts such as in the REDMAP project. The Range Extension Database and Mapping Project (REDMAP) is a web-based citizen science initiative where community members submit photographic observations of species found outside of their native range, which are then verified by expert scientists (Pecl *et al.,* 2014). To date, REDMAP has had over 1,060 reports of species out of their respective ranges verified by over 80 expert scientists (Pecl *et al.,* 2014).

Summary

Citizen science is scientific research or analyses conducted by, or contributed to, from the general public or nonprofessional scientists. Applicable to most scientific disciplines, citizen science is increasing in popularity and is used for many different purposes such as collecting samples, observational monitoring and recording information on specific anomalies. Citizen science can generally be seen as a cost effective way of collecting, and in some cases analysing data, however it does have several more benefits as well as some notable setbacks.

There are many benefits to using citizen science in scientific research. One of the major benefits is its cost effectiveness and efficiency, due to reducing fieldwork and data collection costs, reducing staffing costs and reducing indirect costs such as overheads. Another benefit is that volunteers can collect data over large spatial and temporal ranges. For example, eBird, collects data from over 80 countries, has been for 13 years and as of August 13th, 2012 had 100,333,837 observations (Cornell University, 2012). Other benefits include organisational benefits in relation to sharing data, funding, resources and volunteers, as well as the benefit of enhancing social values attributable to volunteer involvement. There are also several issues with citizen science including organisational, data collection and data usage issues. Organisational issues can include legislation and insurance, funding, and variations in volunteer interest. Data collection issues include error and bias due to variation in observer or sampler quality and/or participant objectivity as well as bias stemming from variation in sampling effort over time and space. Final issues involve those in relation to data usage. These issues are based on the perceived (and in some case, potentially misconstrued) lack of quality in, and distrust of citizen science, as well as issues surrounding data access rights.

Citizen science is used globally to analyse organisms, objects, patterns and phenomena, from logging comets and asteroid showers (Fireball-global) and collecting bird observations (eBird-global) to locating and managing invasive plants (Invaders of Texas-America) and monitoring water, air, soil, biodiversity, bugs and the climate (OPAL-United Kingdom) (Lambert, 2014). In Western Australia, one of the main disciplines citizen science is used in, is biology and ecology (however, it is also used in many others such as medicine and anthropology). Citizen science in WA is used to monitor and sample many different ecosystems from terrestrially locating invasive fauna (FeralScan) to logging marine species observed out of their natural distribution while fishing, snorkelling or diving (REDMAP). Citizen science is used as a research tool in many aquatic projects in WA, including projects that utilize tagging data, biological donations, logbooks and monitoring techniques and those that identify movements, patterns and range shifts.

Section 4: Using Baited Remote Underwater Video systems to field test artificial reef monitoring technology and methodologies suited to citizen science

Overview

This section of the report is based on Chapter 4 in James Florisson's honours thesis, which was completed in June 2015. The results were important in the context of the broader study because they demonstrated that a custom-designed Baited Remote Underwater Video system, which could easily be deployed by recreational fishers, is suitable for recording the fish on the Geographe Bay artificial reefs. Moreover, the results presented here suggest that any monitoring of the fish assemblages of the artificial reefs needs to take into account the direction the camera is facing, *i.e.* towards or away from the reef. This section also provides preliminary data on the types of fish present on the Dunsborough artificial reef, which include some important recreational species, such as Silver Trevally and Pink Snapper.

Introduction

Monitoring of marine environments by resource and environmental managers and/or researchers can provide robust quantitative data that are of sufficient quality to inform management decisions. However, a drawback of using governmental and tertiary education providers to undertake research programs is that these projects can be expensive and time consuming. One method to reduce some of these costs is to utilise citizen scientists to undertake community monitoring, as such programs can cover a larger area, in less time at a lower cost (Hill and Wilkinson, 2004; Silverton, 2009; Dickinson *et al.*, 2010; Wiersma, 2010; Baltais, 2013; Wilson and Godinho, 2013; Sullivan *et al.*, 2014). In recent years there has been an increase in the use of citizen science, particularly for obtaining data over large spatial and temporal scales cost effectively (Silverton, 2009; Dickinson *et al.*, 2010; Baltais, 2013; Lambert, 2014), and there are currently several marine-based citizen science projects being undertaken in Western Australia (*e.g.* Department of Fisheries, 2012c; Fairclough *et al.*, 2014; Lambert, 2014). As mentioned in the Overall introduction, following the

deployment of the two artificial reefs in Geographe Bay there is a legislative requirement to monitor the structural integrity of the reefs on an annual basis, as a condition of government approvals to deploy the reefs. In a similar manner to the work undertaken in Section 2 a citizen science monitoring regime for the reefs is currently being designed by Recfishwest, using Baited Remote Underwater Video (BRUV) systems to monitor artificial reefs, rather than the a 'drop camera' method tested earlier.

The overall aim of this component of the study was to determine i) the effectiveness of another method of video capture of fish on artificial reefs, *i.e.* BRUVs, and ii) the effect of randomly placing the BRUVs in the vicinity of the artificial reef clusters. Specifically, the second aim investigated whether the direction of the camera, i.e. pointing towards or away from the modules, had any effect on fish fauna captured on the BRUV footage. The starting hypothesis was that the characteristics of the fish fauna recorded from BRUVs directly facing the artificial reef modules would be different from those recorded from BRUVs facing away from the modules. Thus, the results of this study will provide an indication was to whether randomised BRUV deployment is a viable method to employ in citizen science monitoring program for artificial reefs.

Materials and methods

Study site

This study was conducted on the artificial reefs located in in Geographe Bay near Busselton. The artificial reef is located approximately 5 km from the Dunsborough boat ramp at 33° 3.962'S 115° 9.980'E. Full details of the composition and design of the artificial reef and on Geographe Bay and its environmental characteristics are given in Section 1.

Sampling regime

Forty seven underwater videos, each of ~17 minutes in duration, were obtained from a Baited Remote Underwater Video (BRUV) system (Fig. 15) deployed around the Dunsborough artificial reef on 10th and 19th of March 2015 by staff from Ecotone Consulting.

BRUVs are weighted frames that contain single or multiple cameras to film an area around a bait bag, which is used to attract fauna, these systems can be orientated horizontally or vertically and be deployed on the seafloor or in the water column (Mallet and Pelletier, 2014). On each sampling occasion, four BRUVS were deployed in succession to collect video footage. The first BRUV was deployed close to the artificial reef centre point with each subsequent camera deployed along a spiral path through the artificial reef area, using a GPS for navigation. Note that this sampling design was developed by staff from Recfishwest and Ecotone Consulting and involved no input from staff and students at Murdoch University. The methodology was chosen to replicate, in part, the movements of recreational fishers and sample randomly areas in and around the artificial reef modules to test the validity of a randomised BRUV deployment method for potential future use with citizen scientists.

Fig. 15. Construction of the custom made BRUV. From right to left: Cementing pipe fixtures with weights already inside the legs (skids), the finished BRUV frame trialling with camera position, and final product about to be deployed on the artificial reef.

Once deployed, each camera was submerged for ~20 minutes before being retrieved. Upon retrieval, the video footage was extracted and GPS coordinates of the location recorded. The BRUV was then rebaited and redeployed in a random location along the spiral trajectory. Sampling lasted for around six hours on each day.

The BRUVS employed in this study were designed and constructed from readily available materials. The frame for each BRUV, which covered an area of around 580 mm x 450 mm, was constructed from class 9 Polyvinyl Chloride (PVC) irrigation pipe, which is rated to 8.88 atmospheres and thus able to withstand pressures associated with water depths to at least

78 meters. Lengths of pipe and the associated fittings are glued together with green PVC cement, traditionally employed for gluing pressurised water pipes. The frame is stabilised by two skids/platforms, each filled with four 680g lead weights, making the BRUV negatively buoyant, with a total weight of 5.5 kg. Pipe brackets were used to mount a camera, a rope tie point (both on top) and the bait arm suspended underneath. The bait arm (or boom) is suspended 150 mm above the substrate and has a length of 600 mm from the BRUV central point, with a bait bag placed 500 mm from the camera (Ellis and DeMartini, 1995; Willis and Babcock, 2000; Heagney *et al.*, 2007). The bait bag, which was 180 mm x 100 mm, was constructed from plastic mesh.

Before each deployment, 500 g of Australian Sardine *Sardinops sagax*, or congeneric species *Sardinops* spp., was placed into the bait bag. These species are widely used in similar studies due to their soft oily flesh, which is known to attract fish (McLean *et al.*, 2010; Watson *et al.*, 2010; Bassett and Montgomery, 2011; Goetze *et al.*, 2011; Mallet and Pelletier, 2014). Moreover, Dorman *et al*. (2012) tested various bait types in BRUVs and concluded that the use of Australian Sardine, as standardised bait for BRUVs, is justified for use along the west coast of Western Australia.

A GoPro Hero 4 Silver Action Video Camera [TM] was mounted to the BRUV and used to record the video footage. This camera was chosen as it has an ultra-wide angle lens and is able to recorded video footage with resolution of 1080p at 60 frames per second. To make the camera more suitable for use in the study the standard housing was replaced with waterproof housing to increase the depth rating from 40-60 m and Battery BacPac [TM] was used to extend battery life to around over three hours.

Video metadata

Once footage was uploaded, it was classified and grouped for video metadata analyses. To assist with classification and analyses, videos attributes were recorded including footage number, whether the camera was facing a) one or more of the modules or b) none of the modules, quality rating and observational notes. The footage quality was rated using the same methods as in Section 2, using a scale of 1-10 (Fig. 6). Of the 47 videos collected, a

random subset of 15 facing the modules and 15 facing away were selected for data extraction.

Observation protocols

For each of the 30 videos, the Max-N, for each species, *i.e.* the largest number of individuals of a species on a single frame of footage, was calculated (Priede and Merret, 1996; Willis and Babcock, 2000). This measure of abundance was employed as it avoids the possibilities of fish double counting and gives a conservative estimation of relative fish density (Priede *et al.*, 1994; Cappo *et al.*, 2004; Watson *et al.*, 2005; Gomelyuk, 2012). Unlike in with the drop camera footage, Count-N was not calculated, as in the earlier study this was used to estimate the number of fish that were unable to be identified, and such problems determining the identify of species in this section were greatly reduced by the higher resolution of the footage (see later). Although all videos were approximately the same length, *i.e.* ~17 minutes, to ensure direct comparability among videos a standardised viewing time of five minutes was established between 7 and 12 minutes. This 5 minute period was analysed for extracting the number of modules observed, various metadata, number of species, ecological group affinities and mean abundance of individuals (Max-N).

Each species recorded was also assigned to an ecological group affinity using the Nakamura (1985) classification (see inside cover). Under this scheme, each species is assigned to a type based on their typical spatial position with regard to the reef (Tessier *et al.*, 2005; Bortone, 2007). Thus, A type species are found proximate to/or inside holes and crevices on the reef and are thus classified as *benthic*. B type species are found closely associated with the reef, but not in direct contact are known as *epibenthic* and C type species are loosely associated with structure, often found schooling above it and constitute *pelagic* species (Nakamura, 1985; Bortone, 2007; Wartenburg and Booth, 2014).

Statistical analyses

A data matrix containing the Max-N for each species in each video was subjected to the DIVERSE routine in Primer v7 (Clarke *et al.*, 2014b) with the PERMANOVA+ add on

(Anderson *et al.*, 2008) to calculate the number of species and 'total' number of individuals. The data for each of the biotic variables was used to construct a Euclidean distance matrix and subjected to one-way Permutational Analysis of Variance (PERMANOVA; Anderson, 2001) to determine whether the values for each of those measures differed significantly between the videos recorded from BRUVs facing towards the artificial reef modules and those facing away. The null hypothesis that there was no significant difference was rejected if the significance level (*P*) was ≤ 0.05. Prior to undertaking these analyses, the data for the number of individuals were square-root transformed, while the number of species did not require transformation. The arithmetic means and associated 95% confidence intervals were calculated and graphed to visually determine the cause of any significant differences.

To undertake multivariate analyses, the untransformed data matrix used above was fourth-root transformed to down-weight the contributions of species with consistently relatively high values and balanced them with the values of rarer species and used to construct a Bray-Curtis similarity matrix. This matrix was then subjected to the same one-way PERMANOVA test described above, only this time operating a multivariate sense. The above Bray-Curtis similarity matrix was then subjected to non-metric Multi-dimensional Scaling (nMDS; Clarke, 1993) to produce an ordination plot to explore visually, any trends among the fish compositions on the video recorded facing different directions.

Finally, a shade plot, was produced from the fourth-root transformed fish fauna data for each video, averaged for those 15 samples facing towards and those 15 samples facing away from the modules. This plot was used to visualise the trends exhibited by the Max-N abundances of the various fish species on the video recorded facing different directions. This plot is a simple visualisation of the frequency matrix, where a white space for a species demonstrates that the taxon was never collected, while the depth of shading from grey to black is linearly proportional to the density of that taxon (Clarke *et al.,* 2014a; Valesini *et al.,* 2014).

Results

Video metadata

From the total of 47 videos, 30 videos were randomly selected, with the camera in 15 of those videos facing one or more of the modules (*i.e.* facing modules), whereas in the other 15 videos the no modules were observed in the footage (*i.e.* facing away). Each of the videos ranged between 17 and 20 minutes in duration, with a five minute section between 7 and 12 minutes analysed qualitatively for quality using the scale shown in Fig. 6. The quality of the footage ranged between 7 and 9 (out of ten; Fig. 16). The average quality level of all the videos was 8.13 and was similar in videos facing modules (8.40) and those facing away (7.86).
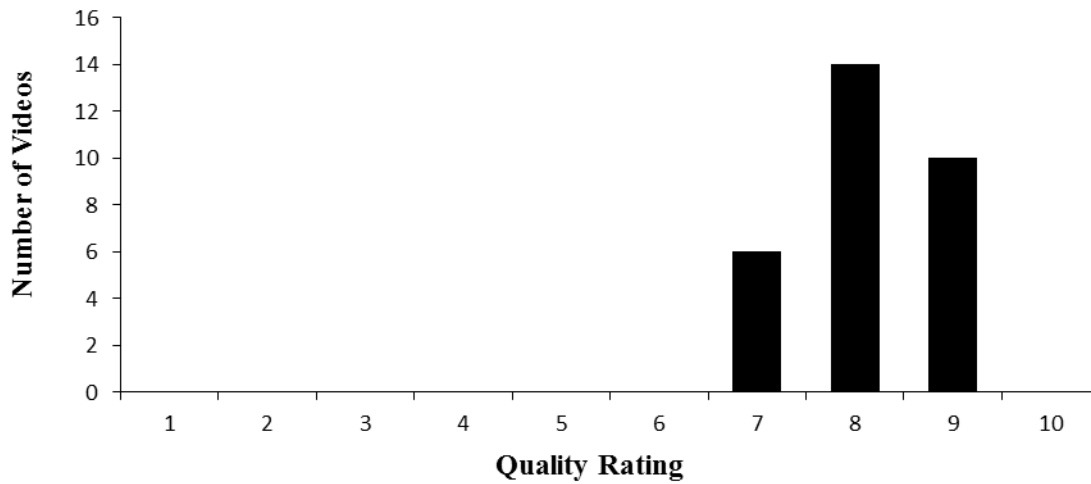
Fig. 16. The quality rating of the 30 videos collected using BRUVS on the Busselton artificial reef.

Of the 30 videos collected 50% was footage observing areas with no artificial reefs. Out of the other 15 video that captured at least one of the artificial reefs modules in the field of view, 11 (~73%) observed one module, 4 (~27%) observed areas with two modules and none filmed areas with more than two (Fig. 17).
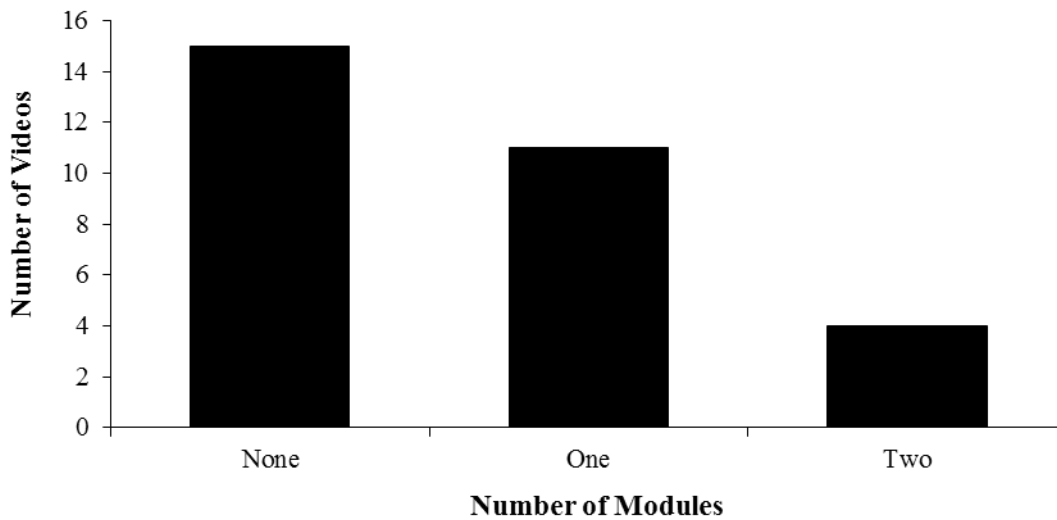
Fig. 17. The number of modules observed in each of the 30 videos analysed. Note: half of the videos intentionally observed no modules.

Descriptive metrics

A total of 33 species of fish and one species of mollusc were identified from the five minute sections of footage from the 30 videos (*i.e.* 2 hours and 30 minutes in total) and together represented each of the three Nakamura (1985) ecological group affinities (A, B and C). The 44% of the species recorded (15) constituted the 'A Type' as they were benthic, while the next most numerous affinity was B (epibenthic), which was represented by 12 species (Fig. 18). Thus, together species that were cryptic and closely associated to structure respectively species made up 79% of the total number of species were thus more speciose than the pelagic fauna (C type), which comprised seven species. It should be noted that while *Sepioteuthis australis* (Southern Calamari) is not a teleost or elasmobranch, it has been included in the data sets as it is a species targeted by recreational fishers.
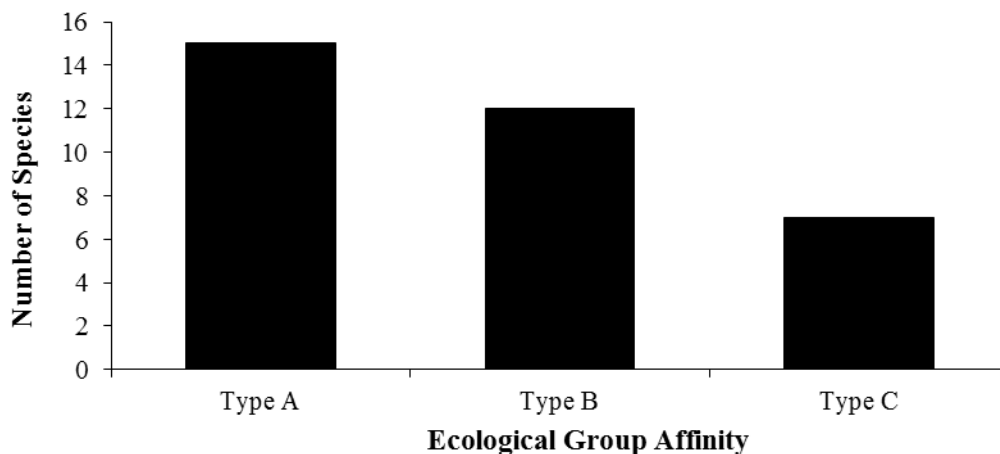
Fig. 18. Numbers of species assigned to each of the three Nakamura (1985) ecological group affinities, *i.e.* A (benthic), B (epibenthic) and C (pelagic).

The greatest mean number of species was recorded in footage where two modules were observed in the field of view (Fig. 19). In such footage, species belonging to type B were more numerous (2.75) than those in types A (1.25) or C (1). In contrast, the lowest mean number of species was recorded on videos where no modules were observed and on these videos there was little difference between the mean number of species in each of the three ecological groups (all ~ 0.5 species/video). Footage in which, one module was observed fell between the two 'extremes', with slightly greater mean numbers of species in types A and B (both ~1) than C (0.45; Fig. 19). Cameras facing modules, *i.e.* those with one of two modules

in the field of view) had approximately 36.3% more A species and 50% more B species than camera footage not facing modules (Fig. 20). However, cameras not facing modules had a higher level of 'Type C' (pelagic) species, recording 25% more than footage observing modules.
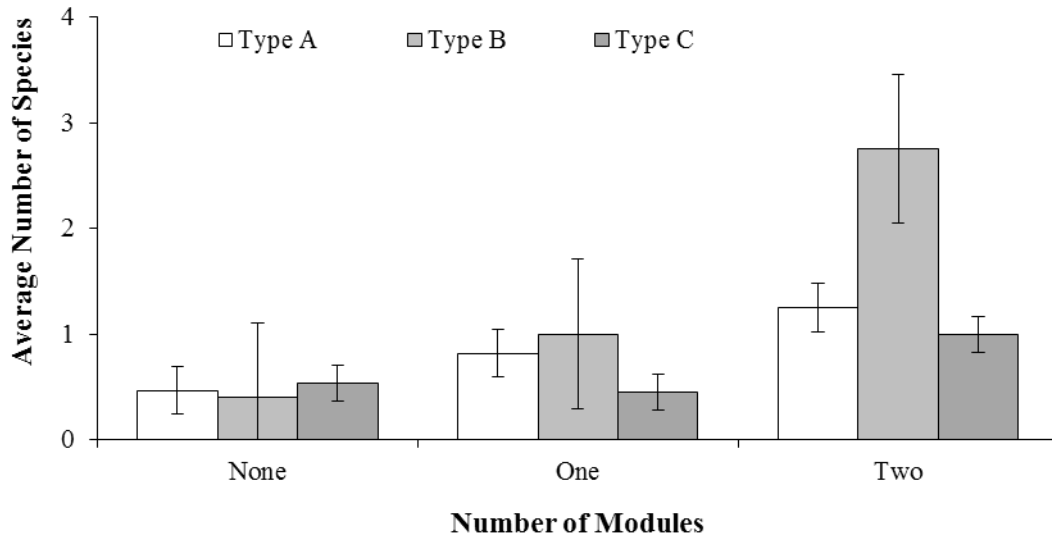
Fig. 19. The average number of species recorded belonging to each three Nakamura (1985) ecological group affinities, *i.e.* A (benthic), B (epibenthic) and C (pelagic), observed in each video with different numbers of modules in the field of view. Error bars represent ± 1 standard error.
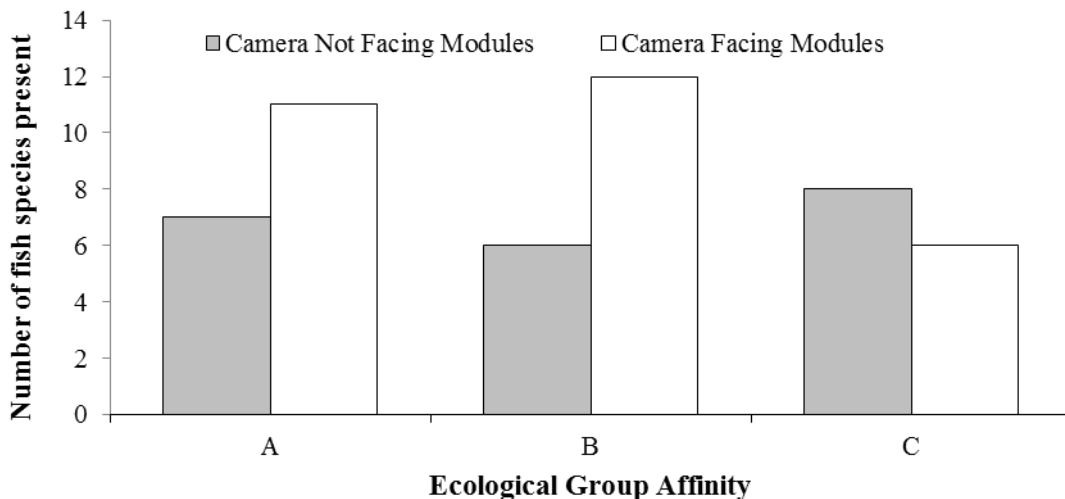
Fig. 20. The number of species present in each of the three Nakamura (1985) ecological group affinities, *i.e.* A (benthic), B (epibenthic) and C (pelagic) in videos where the camera was facing towards or away from the artificial reef modules.

The mean number of species increased sequentially with the amount of modules in the field of view of the camera, with by far the greatest values recorded for two modules (5) than either 1 (2.3) or none (1.4; Fig. 21).
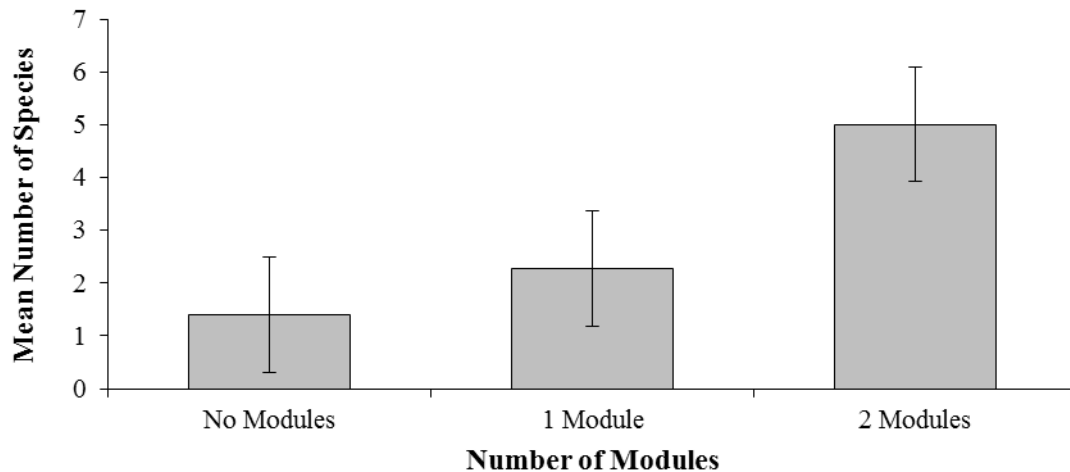
Fig. 21. The average number of species observed in videos with different numbers of modules in the field of view. Error bars signify the variability of mean number of species in the differing amounts of modules.

Average mean abundance (calculated from the total Max-N averaged across a suite of videos) increased sequentially with the number of modules in the field of view. Thus, the lowest average mean abundance was recorded for camera facing away from the modules (~27) and 31% than the greatest average mean abundance of (~39) recorded from videos in which to two modules could be seen (Fig. 22).
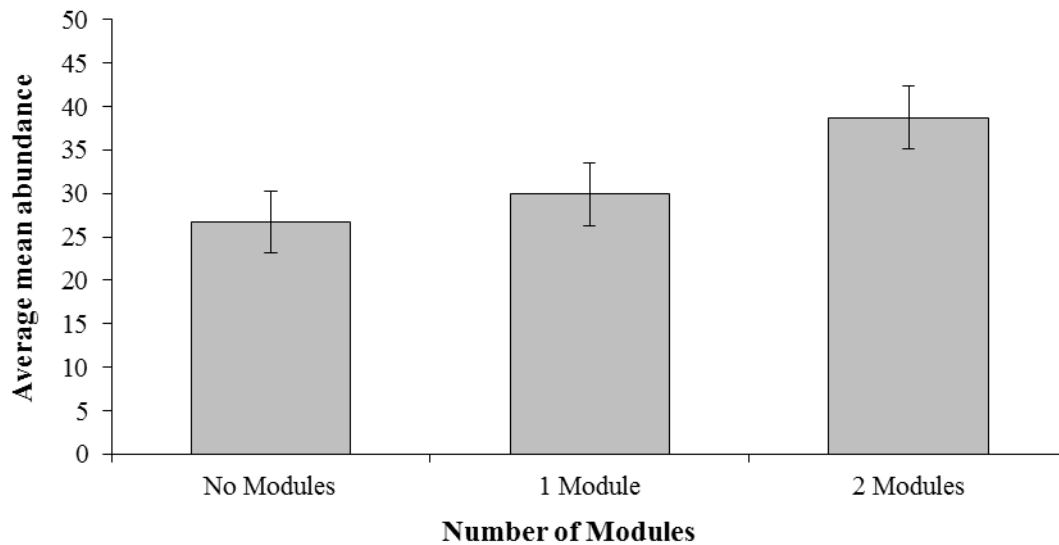
Fig. 23. The averaged mean abundance observed in videos with different numbers of modules in the field of view. Error bars signify the variability of average mean abundance in the differing amounts of modules.

A total of 34 species were identified from the 30 videos analysed in this study (Table 4). Of those species, 29 were recorded in footage observing artificial reef modules, 21 species were recorded in footage were no modules were observed and 17 species (50%) were recorded in both areas. It is also noteworthy that 12 species (~35%) were recorded only in footage that observed artificial reef modules, while 5 species (~15%) were recorded only in footage that contained no artificial reef modules (Table 4). Relatively similar total number of individuals was also recorded with 484 footage with modules and 401 on footage without modules.

A suite of ten species contributed over 90% to the total number of individuals recorded around the Busselton artificial reef. Of those ten, three were particularly abundant namely *P. georgianus* (Sand Trevally), *C. auricularis* (Western King Wrasse) and *N. obliquus* (Footballer Sweep), with each species representing not only more than ~5% to the total number of individuals overall, but also on the sets of videos facing towards and away from the modules. Such was the dominance of *P. georgianus* that is represented almost 60% of the total fish fauna and almost 70% on the videos facing away from the modules.

While the seven top ranked species were present there were some differences in abundance with greater counts of particularly *C. auricularis*. Species such as *Chromis klunzingeri* (Blackhead Puller), *Trachurus novaezelandiae* (Yellowtail Scad) and *Trachinops noarlungae* (Yellow Head Hula Fish) all represented >1% of the total number of individuals recorded when the camera was facing the modules, but were absent on videos where the camera faced away. Although none of the four species only recorded on footage facing away from the modules contributed >1% to the total number of individuals, it is noteworthy that those species comprised the two of the three elasmobranch species, *i.e.* *D. brevicaudata* (Smooth Stingray) and *T. personata* (Masked Stingaree) and the recreationally important *C. auratus* (Pink Snapper).

Table 4. Average individual mean abundance (#), percentage composition (%) and rank (R) of individual species recorded in footage facing modules and not facing modules. Total number of species and individuals are also provided. Grey shading indicates species that contributed ~>5% to the total number of individuals.

Species Name	Facing Modules			Not Facing Modules			Total		
	#	%	R	#	%	R	#	%	R
Pseudocaranx georgianus	231	48.53	1	278	69.33	1	509	57.51	1
Coris auricularis	83	17.44	2	39	9.73	2	122	13.79	2
Neatypus obliquus	33	6.93	3	20	4.99	3	53	5.99	3
Parequula melbournensis	14	2.94	5	10	2.49	4	24	2.71	4
Anoplocapros amygdaloides	14	2.94	5	10	2.49	4	24	2.71	4
Austrolabrus maculates	15	3.15	4	1	0.25	10	16	1.81	5
Pempheris klunzingeri	9	1.89	7	7	1.75	5	16	1.81	6
Chromis klunzingeri	13	2.73	6				13	1.47	7
Trachurus novaezelandiae	13	2.73	6				13	1.47	7
Seriola hippos	5	1.05	9	6	1.50	6	11	1.24	8
Diodon nicthemerus	8	1.68	8	2	0.50	9	10	1.13	9
Trachinops noarlungae	8	1.68	8				8	0.90	10
Upeneichthys vlamingii	5	1.05	9	2	0.50	9	7	0.79	11
Sepioteuthis australis	2	0.42	11	4	1.00	7	6	0.68	12
Ophthalmolepis lineolatus	5	1.05	9	1	0.25	10	6	0.68	12
Myliobatis australis	3	0.63	10	3	0.75	8	6	0.68	12
Trygonorrhina fasciata	2	0.42	11	4	1.00	7	6	0.68	12
Glaucosoma hebraicum	2	0.42	11	3	0.75	8	5	0.56	13
Dasyatis brevicaudata				4	1.00	7	4	0.45	14
Chelmonops curiosus	3	0.63	10				3	0.34	15
Lagocephalus lunaris	2	0.42	11	1	0.25	10	3	0.34	15
Anoplocapros lenticularis	2	0.42	11	1	0.25	10	3	0.34	15
Pentaceropsis recurvirostris	2	0.42	11				2	0.23	16
Chrysophrys auratus				2	0.50	9	2	0.23	16
Cheilodactylus nigripes	2	0.42	11				2	0.23	16
Halichoeres brownfieldi	2	0.42	11				2	0.23	16
Eubalichthys mosaicus	2	0.42	11				2	0.23	16
Parazanclistius hutchinsi	1	0.21	12				1	0.11	17
Pseudolabrus biserialis	1	0.21	12				1	0.11	17
Parapercis haackei	1	0.21	12				1	0.11	17
Trygonoptera personata				1	0.25	10	1	0.11	17
Suezichthys cyanolaemus				1	0.25	10	1	0.11	17
Neosebastes bougainvillii	1	0.21	12	0	0.00		1	0.11	17
Parpercis ramsayi	0	0.00		1	0.25	10	1	0.11	17
Total number of species		29			21			34	
Total number of individuals		484			401			885	

Statistical analyses

One-way PERMANOVA demonstrated that there was a significant difference between the mean number of species recorded from video collected from cameras facing towards or away from the modules (Table 5). On average, cameras facing towards the modules recorded ~7.5 species, compared to 5 on videos where the camera was not facing modules (Fig. 24a). In contrast to the number of species, mean number of individuals (the total Max-N for each video) was shown by PERMANOVA not to differ significantly between the two

types of videos. In both cases, ~30 individuals were observed within the five minute period (Fig. 24b).

Table 5. Mean squares (MS), Pseudo-*F* (p*F*) values and significance levels (*P*) for a one-way PERMANOVA test on (a) number of species and (b) mean total number of individuals (the total Max-N for each sample) calculated from the 30 videos recorded with camera facing towards or away from the artificial reef modules.

(a) Number of species	df	MS	p*F*	P
Camera direction	1	43.2	6.88	0.013
Residual	29	6.3		

(b) Number of individuals	df	MS	p*F*	P
Camera direction	1	1.94	1.78	0.212
Residual	29	1.09		
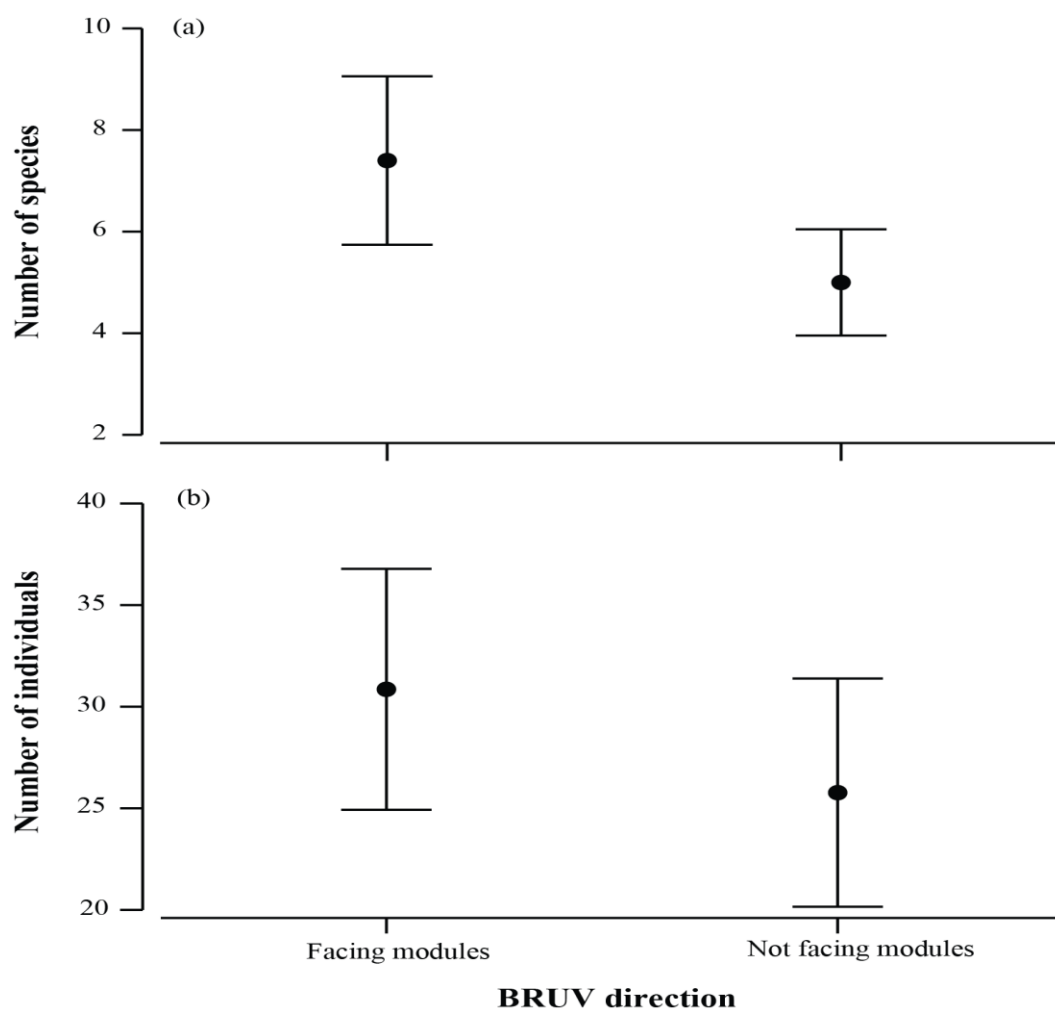
Fig. 24. (a) mean number of species and (b) mean total number of individuals (the total Max-N for each sample) calculated from the 30 videos recorded with camera facing towards or away from the artificial reef modules. Error bars represent 95% confidence limits.

One-way PERMANOVA detected a significant difference between the fish faunas recorded with the camera facing towards *vs* away from the artificial reef modules (Table 6). This difference is illustrated on the nMDS ordination plot, where the points representing the two camera angles are broadly separated on opposite sides of the plot. Thus, those samples obtained from cameras facing the modules are located on the left hand side of the ordination and only intermingle with five of the samples obtained from cameras facing away from the plot (Fig. 25). Note that each point represents a single sample and that the magnitude of the differences exhibited on the plot maybe increase if the samples were averaged.

Table 6. Mean squares (MS), Pseudo-*F* (p*F*) values and significance levels (*P*) for a one-way PERMANOVA test, employing a Bray-Curtis resemblance matrix constructed from the fourth-root transformed Max-N data calculated from the 30 videos recorded with camera facing towards or away from the artificial reef modules.

	df	MS	p*F*	P
Camera direction	1	3858	3.29	0.005
Residual	29	1174		
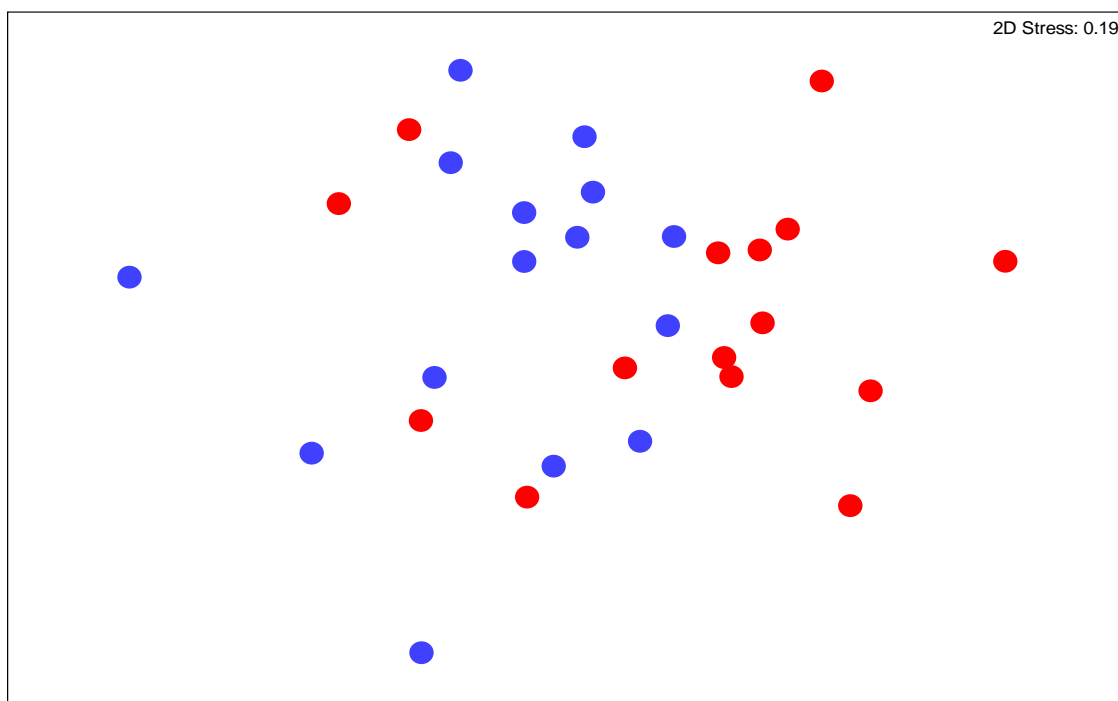
Fig. 25. nMDS ordination plot derived from Bray-Curtis similarity matrix a constructed from the fourth-root transformed Max-N data calculated from the 30 videos recorded with camera facing towards ● or away from the artificial reef modules ●.

Interpretation of the shade plot, which was constructed from the same pre-treaded data used to produce the nMDS plot, demonstrated that the faunas were dominated by *P. georgianus* and that it occurred in approximately equal abundances regardless of the camera direction (Fig. 26). There was also a suite of five species that were relatively abundant in both groups of samples, *i.e. C. auricularis*, *N.obliquus, A. maculates Parequula melbournensis* (Southern Silver Belly) and *Anoplocapros amygdaloides* (Western Smooth Boxfish), but were present in greater numbers on videos recorded facing the modules. Several species such as *D. brevicaudata, C. auratus*, *T. personata*, *Suezichthys cyanolaemus* (Bluethroat Rainbow Wrasse) and *Parpercis ramsayi* (Sand Perch) were only found on videos facing away from the modules, whereas the reverse was true for fishes, *e.g. C. klunzingeri*, *T. novaezelandiae, Pentaceropsis recurvirostris* (Longnose Boarfish), *Cheilodactylus nigripes* (Magpie Perch) and *Halichoeres brownfieldi* (Brownfields Wrasse), however, in almost all cases the abundances of these species were low (Fig. 26).
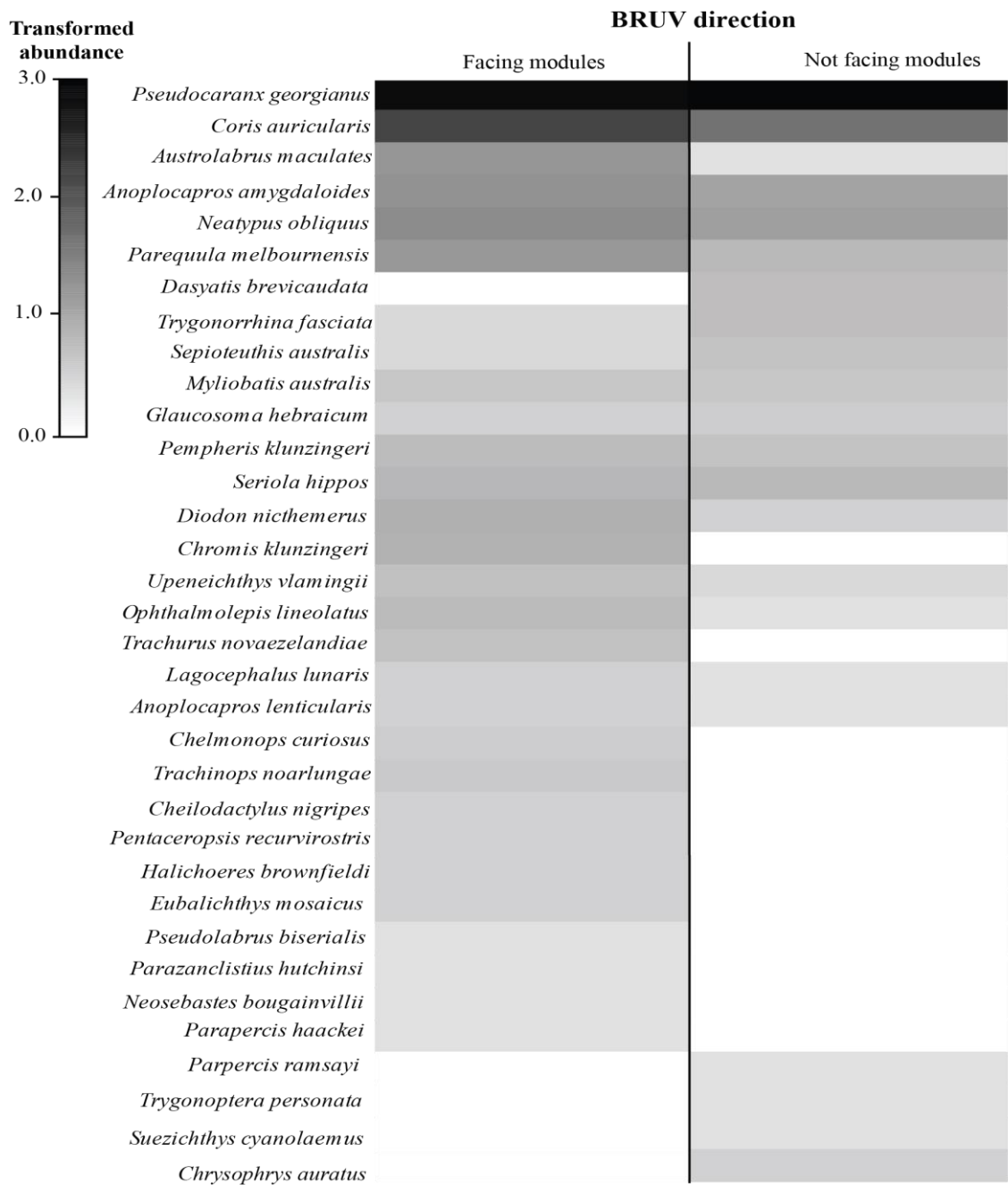
Fig. 26. Shade plot of the fourth-root transformed Max-N data calculated from the 30 videos recorded with camera facing towards or away from the artificial reef modules.

Discussion

The characteristics of the fish faunas living in and around the artificial reef in Busselton were quantified by recording the maximum abundance of each species identified in 30 videos obtained from Baited Remote Underwater Video (BRUV) systems. Of these videos, 15 were obtained when the camera was facing one or more of the modules, whereas the other 15 were obtained when the camera was facing away from those modules. The resultant data were used to test the hypothesis that the characteristics of the fish fauna (diversity, abundance and faunal composition) would change depending on the direction the camera was facing. A secondary aim was also to test the effectiveness of another method of video capture of fish on artificial reefs.

Video metadata

The video footage was collected by a researcher (not affiliated with this project or Murdoch University), who followed a standardised methodology for each replicate and thus the duration of all videos was approximately equal at ~17 minutes. This was in stark contrast to the citizen science approach (detailed in Section 2), in which video length ranged from 10 seconds to 13 minutes. As a result, a less biased approach to standardisation was able to be applied, *i.e.* comparing a set length of footage from defined start and end points (this section) *vs* calculating an average count for each species per minute (Section 2), which of course would bias diversity measures based, in some part, on the number of species (Clarke *et al.,* 2014b). The qualitative index for quantifying video quality scored the videos in this section with an average rating of 8.13 (out of 10), far higher than the 3.65 recorded in Section 2. This was due to both the higher resolution of the GoPro Hero 4[TM] *vs* the Sony CCD 700 TVL and the lack of turbidity encountered during the time the BRUVs very deployed. Therefore, while under calm conditions and when visibility is good the GoPro Hero 4[TM] should obtain higher quality footage, it remains to be seen whether this would still be the case on days were turbidity were higher.

Characteristics of the fish faunas facing towards and away from the artificial reef modules

Ecological groups

Although the artificial reef modules are in relatively close proximity to one another, there were some changes in the habitat recorded when the camera was facing towards or away from the modules. The benthos observed on footage facing away from the modules was predominantly sand, with occasionally the edges of beds of the seagrass *Posidonia sinuosa* (Oldham *et al.,* 2010), while those facing the modules recorded lower amounts of seagrass and, of course, the modules, which had established a relatively rich epibiotic community. In light of the habitat availability the Nakamura (1985) classification of ecological group for fish, which is based on their vertical distribution in the water column and their position relative to reef (Tessier *et al.,* 2005) , was modified to; Type A, *benthic* species were in direct contact with the seagrass and/or sand substrate, Type B, *epibenthic* species were in the immediate vicinity but not in direct contact of the other substrates and that Type C, *pelagic* species, were found mid-water above the different substrates.

Footage in which modules were observed recorded 37% more Type A and 50% more Type B species than footage where modules were not observed. Such a trend is not unexpected as, Type A and Type B fish are benthic and epibenthic, respectively, species, and thus would be more likely to be found in areas containing reef (artificial or natural) as the presence of reef increases habitat complexity and can provide shelter, food and induce different behavioural aspects of these species (Ody and Harmelin, 1994; Charbonnel *et al.,* 2002; Sherman *et al.,* 2002). Fewer numbers of species of Type C (pelagic) fish were recorded in both environments than type A or B species, a result which mirrors that of Tessier *et al* (2005) on natural/artificial reefs off Reunion Island (SW Indian Ocean). When comparing the two environments, slightly fewer numbers of Type C species recorded in footage facing modules (6) than away (8). As many of these species are pelagic and some highly mobile, one might not expect there to be a difference in the numbers of these type of species, particularly when the reefs are benthic, rather than pelagic in the case of a fish aggregation device. Nevertheless, as many of the pelagic species are higher order predators, such as *Seriola hippos* (Samson Fish) and *C. auratus* (Pink Snapper), their distribution may be related more

to the presence of potential prey species rather than their attraction to the reef or bare habitat.

Numbers of species and individuals

It is also noteworthy that, the number of species representing each of the ecological groups increased sequentially along with the numbers of modules observed in the footage. This may indicate that the presence of increasing modules, which, in turn, increases habitat complexity may be beneficial in increasing diversity.

A suite of studies, undertaken throughout the world, have demonstrated that the number of species and abundance was greater on artificial reefs than natural reefs/ surrounding habitats (*e.g.* Bombace *et al.,* 1994; Rilov and Benayahu, 2000; Sherman *et al.,* 2002; Charbonnel *et al.,* 2002; Folpp *et al.,* 2013). This was partially true in this study, where there was a significant increase in the number of species and a slight (but not significant) increase in the abundance of fish on footage facing towards rather than away from the reef.

The increased number of species recorded in the present study is likely due to the creation of complex habitat and shelter (Svane and Peterson, 2001; Sherman *et al.,* 2002; Hunter and Sayer, 2009), the source of food (Cresson *et al.,* 2014), vertical profile (Kellison and Sedberry, 1998), edge effects (Dorenbosch *et al.,* 2005) and potential upwelling effects (Yanagi and Nakajima, 1991) provided by the artificial reefs. As with the numbers of species in each ecological group increasing with the number of modules, the same was true for the total number of species. This increase could be explained by the additional modules increasing the surface area for colonisation of epifauna and associated organisms, thus fuelling further biomass production (Cresson *et al.,* 2014) creating more feeding opportunities. Another possible reason for the increase is that more modules provide a higher level of habitat complexity providing more shelter and differing environmental conditions (hydrological, temperature and light) (Svane and Peterson, 2001; Hunter and Sayer, 2009), which could propagate higher abundances of more different types of species.

While, there was a sequential increase in the mean number of individuals recorded with increasing numbers of modules, and a larger number of fish recorded on videos facing

towards rather than away from the reef, these differences were low and also subjected to relatively high levels of variability. Thus, in the case of the latter comparisons, no significant difference was detected. Such a trend is likely influenced by the variability in the numbers of *P. georgianus* (Sand Trevally) a highly schooling pelagic species, recorded in the individual samples. This species dominated the fish fauna to such an extent that it represented almost 60% of the total fish fauna and almost 70% on the videos facing away from the modules.

Of the 34 species identified, 12 were only recorded on footage facing modules, with 5 species being found only on footage not facing modules and 17 species being found on both suites of footage. Of the five species observed solely in areas without modules, three are mainly found in/on seagrass meadows and sand, and two of these species elasmobranchs, namely *D. brevicaudata* (Smooth Stingray) and *Trygonoptera personata* (Masked Stingaree). Both of these species are more commonly found over sand and seagrass (White, 2006; Duffy and Paul, 2003). All of the 12 species only recorded in videos facing the modules were fish typically associated with reef or rock habitats, including two species of wrasse (Labridae) and two species of boarfish (Pentacerotidae). Furthermore, all of these species were attributed to ecological group types A and B except for *Trachurus novaezelandiae* (Yellowtail Scad), which was only recorded in a single video.

Fifty percent of the species were recorded by cameras facing towards and away from the modules. Of these 17 taxa, 9 are associated with both rocky reef and seagrass/sand, while another 3, namely *P. georgianus*, *Parequula melbounensis* (Southern Silver Belly) and *Myliobatis australis* (Southern Eagle Ray) are predominantly found over purely sand or seagrass habitats (Froese and Pauly, 2015). The presence of these species around the modules could be attributed to several factors including the fact that modules are deployed on sand and are typically located in close proximity to seagrass meadows. The intermingling of these three 'substrate types' thus creates a mosaic of habitats, which the above species are able to exploit. It is also hypothesised that the modules and their associated epiphyte community may attract fish from nearby 'alternative' habitats. For example, *P. georgianus* and *P. melbournensis*, which were ranked first and fourth overall in terms of abundance, respectively, feed predominantly on copepods (Platell *et al.,* 1997), which are themselves

attracted to artificial reefs by other invertebrates feeding on the organic matter produced by the reef (Cresson *et al.,* 2014). Another suite of species (5), found to occur in both sets of footage, are predominantly associated with rocky habitats, namely *Glaucosoma hebraicum* (Western Australian Dhufish), *S. hippos* (Samson Fish), *Austrolabrus maculates* (Black Spotted Wrasse), *Pempheris klunzingeri* (Rough Bullseye) and *N. obliquus* (Footballer Sweep) (Froese and Pauly, 2015). The last three fish are small, schooling species that feed on invertebrates associated with the reef (May and Maxwell, 1986; Platell and Potter, 2001; Bray, 2011). Therefore, the shade, shelter and food production caused by the artificial reef modules may aggregate these species. The fact that these species were also recorded in footage where modules were not observed could be attributed to the fact that modules are very close by, and/or these species are moving between modules or that the species were attracted from the modules into the cameras field of view due to the bait plume or behaviour of other species.

Both *S. hippos* and are *G. hebraicum* are larger species, reaching 180 and 122 cm, respectively and are high trophic-level predators (Smallwood *et al.,* 2013). Their presence in both data sets could be a combination of i) attraction to the bait plume, ii) being in transit between territories or modules as they are both highly mobile, and/or iii) they were attracted to the area due to the aggregation of species. In the context of the last point, it is relevant that Rowland, (2009) identified 17 key prey items for *S. hippos,* of which 7 (representing 32.9% of their diet) of the prey item species were recorded in the footage on the reefs. Some of these species in the diet and seen in the footage included: *S. australis* (Southern Calamari), *T. novaezelandiae* (Yellowtail Scad) and various Labrids. It's also noteworthy that *G. hebraicum* feeds on fish species such as *C. auricularis*, which ranked second in terms of abundance, and others *e.g.* members of the Pempheridae (*i.e. P.klunzingeri*) and Ostraciidae (*i.e. A. amygdaloides*; Platell *et al.,* 2010).

Recreationally important species

One of the purposes of monitoring is to evaluate structures against proponents' objectives and one of these main objectives is the propagation of recreational target species (Department of Fisheries, 2012a). As the South West Artificial Reef Trial was partly funded and mainly advocated for by recreational fishers, one of the main objectives for the reef was the increase of recreationally important species such as *C. auratus* (Pink Snapper*), S. hippos* (Samson Fish) *and P. dentex* (Skipjack Trevally). Although this study identified *S. hippos* and *C. auratus* on the artificial reefs, it did not identify any *P. dentex*, although a very similar and targeted species, *P. georgianus* was the most abundant species identified in the study contributing to 57.51% of the total fish assemblage, this species also has the same edibility rating as *P. dentex* (Hutchins and Swainston, 2012)*.* Recreational target species can be defined as those species that are edible (Watson *et al.,* 2007) and thus to analyse the assemblages in relation to target species, the edibility scale from Hutchins and Swainston (2012) will be utilised. The scale ranges from 0-4 with 0 being a fish not generally eaten (usually due to its size or physical morphology) and 4 being the most prized table fish. Some of the species identified in this study are poisonous to ingest and are thus omitted from the data set. These include *Diodon nicthemerus* (Globe Fish), *Anoplocapros amygdaloides* (Western Smooth Boxfish), *Anoplocapros lenticularis* (White Barred Boxfish) and *Lagocephalus lunaris* (Rough Golden Toadfish). These four species are poisonous as they belong to the Order Tetradontiformes, all of the species in this order can produce tetrodotoxin, a lethal natural toxin that if ingested, can result in paralysis and even death for humans (Edgar, 1997 and Hutchins and Swainston, 2012). These four species were all observed in footage facing and not facing artificial reef modules, however they only contributed 4.52% to the overall fish assemblage.

Implications for citizen science

Given some of the problems with the methodology of the citizen science approach to monitoring the fish faunas of artificial reefs using recreational fishers employed in Section 2 and the use of a different technological approach here to collect video footage, there is the

opportunity to comment on the applicability of BRUVS for use in citizen science projects to monitor artificial reefs. This technology was first developed to count abundances of juvenile *Pristipomoides filamentosus* (Crimson Jobfish) in Hawaii in 1995 (Ellis and DeMartini, 1995) and, since then, their use has increased rapidly throughout the world and particularly in Australia. For example, Mallet and Pelletier (2014) identified 52 BRUV-based researcher papers globally, 32 or over 60% of which originated in Australia since 2003.

As mentioned in the materials and methods, the BRUVs employed in this study comprised a weighted frame constructed from PVC pipe filled with lead fishing weights, costing ~$75.00 per unit, on to which a GoPro Hero 4^{TM} was mounted (costing ~$475). Although I was not involved in the deployment of the BRUVS, from watching the video footage obtained there are two ways in which the units could be improved. Firstly, the addition of a second camera would increase the field of view and also, if facing a sufficiently different direction, would help overcome some of the above differences in direction of the camera on the fish fauna captured in the footage. Secondly, it was noted that some larger Batoids such as *D. brevicaudata* and *Trygonorrhina fasciata* (Southern Fiddler Ray) were observed rotating the BRUV and thus it might be worthwhile increasing the weight of the frames. The rotation of BRUVs were a negative factor in this study as the structures in the field of view dictated the grouping of that particular fish faunal data being filmed, whether facing modules or not facing modules. If the BRUVs were rotated from facing a module to facing no modules or from facing the surrounding area to facing an artificial reef module, the data were not included from that footage. Although, this only occurred once throughout the study.

As mentioned earlier, the quality of the footage obtained from the BRUVs (GoPro Hero 4^{TM}) was of a higher quality than that obtained from the Sony CCD 700 TVL camera (see above). This enabled a larger proportion of the fish to be identified and also increased the ease of identifying particular species thus resulting in more accurate results. The former type of camera are starting to be utilised more frequently in research projects due to their recent reductions in size and cost, and increases in quality of footage recorded and data storage capacity. For example, these types of camera have been used to monitor reef fish communities in marine protected areas (*e.g.* De Vos *et al.,* 2014), analysing fish interactions

with artificial structures (*e.g.* Hammar *et al.,* 2012) and seagrass assessments to help monitor dugong and sea turtle habitats using citizen scientists (*e.g.* McKenzie *et al.,* 2014). Furthermore, a study by Letessier *et al.,* (2015) compared low-cost small action cameras to traditional cameras. The purpose of the study was to 'assess the capacity of GoPro™ action cameras to provide accurate stereo-measurements of fish in comparison to the Sony handheld cameras that have traditionally been used for this purpose' (Letessier *et al.,* 2015). The results found that there was a strong correlation (R^2 = 0.94) between the cameras' length measurements of the same individual fish and that any 'difference in measurement accuracy becomes negligible for purposes of comparing population size structure' (Letessier *et al.,* 2015). The study concluded supporting the use of small action cameras such as GoPro™ cameras as they provide reductions in cost and increases in effective sampling efforts (as easier to use) when compared to traditional equipment for stereo-measurements such as the Sony handheld cameras.

The methodology employed in this study was conducted for pilot purposes and was purposely simplified to field-test a potential sampling regime able to be completed by citizen scientists. The method involved starting from the reef centre point (Fig. 27) and randomly deploying the BRUVs at intervals outwards on a spiral path. Although this method provided adequate data, for the purposes of this section of the project, there are several ways it could be improved if it is to be utilized in a citizen scientist monitoring program. Rather than using a spiral, participants could employ a grid system to guide their sampling efforts. In such a scheme, participants would be allocated a suite of grid squared (denoted by GPS co-ordinates), within which they could deploy the BRUV wherever they wish (Fig. 27). This would allow a higher level of randomisation, whilst still following a standardised approach. This would also decrease chance spatial biases due to participants not selected sites objectivity. However, it should be noted that once footage is collected, it would have to be screened to see whether artificial reef modules were in the field of view before analysis.

It is also recommended that each BRUV be deployed for a longer period of time than the 17-20 minutes employed here. A deployment time of 50 minutes will increase footage length,

but still allow three deployments of a BRUV before needing to recharge the battery. From a science perspective, this increase in video length allows more measurements to be collected on a larger number of species and individuals and thus increases the reliability of the results. It is therefore relevant that both Watson (2006) and Watson *et al.,* (2010) stated that at least 36 minutes of footage is required to accurately obtain measures on the majority of fish species and that, if possible, 60 minutes is advisable to obtain measures of numerous targeted species. From a citizen science perspective, increasing the soak time of the equipment would allow the participants to go fishing during the interim period, without having to stop every 10-15 minutes to deploy the camera/BRUV. It is suggested that this would increase the fishers' enjoyment and thus increase the fishers' involvement and motivation towards the project, which are vital aspects to successful citizen science projects (Rotman *et al.,* 2012). The soak duration of 50 rather than 60 minutes is to allow time to deploy and retrieve BRUVs with a one hour period.
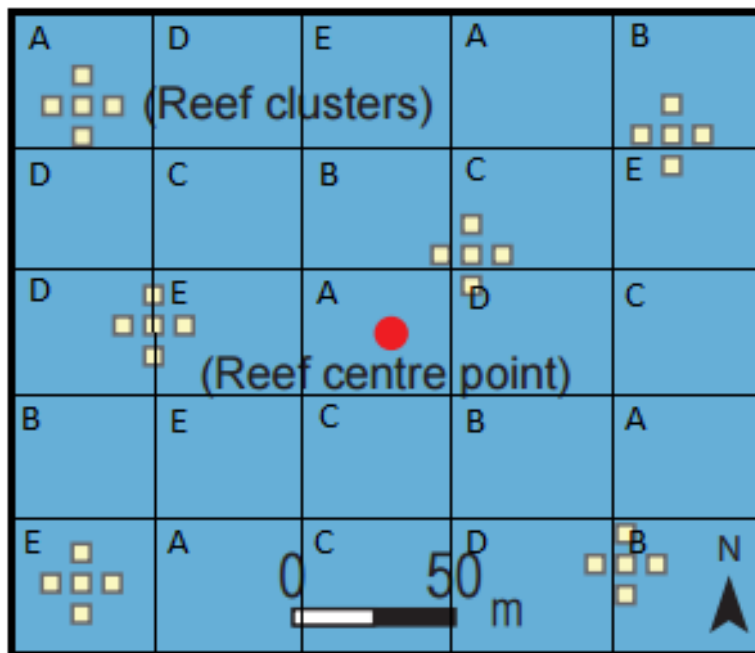
Fig. 27. Schematic of the proposed grid system, which could be utilised to randomise sampling and reduce spatial biases. In this example there are five citizen scientists (A-E) who would each be responsible for collecting data for a small suite of grid squares.

Section 5: Investigating the potential for observer bias in underwater video analysis

Overview

This section of the report is based on Chapter 4 in Thomas Bateman's honours thesis, which was completed in November 2015. The results were important in the context of the broader study because they demonstrate that using multiple observers to identify the fish species seen on video footage from Baited Remote Underwater Video systems can bias the resulting data. Thus, while univariate indices like the number of species and total MaxN remained relatively consistent among observers, there was a significant difference in the composition of the fish faunas. This reflects that fact that, while at the average video level, observers can distinguish between different species, misidentifications of the less common (well-known) species can occur. It is thus recommended that all observers undergo a comprehensive training program before watching and scoring video footage.

Introduction

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

Remote underwater video monitoring offers significant benefits over traditional diver visual census methods in that it reduces the need for skilled observers in the field and enables sampling of depths and for times not possible on SCUBA (Harding *et al*., 2000; Langlois *et al*., 2010; Lowry *et al*., 2012, Pelletier *et al*., 2012). The use of underwater video also has the additional benefit of providing a permanent data set, able to be retrieved at any time, allowing researchers access to a much wider suite of information (Cappo *et al*., 2003).

Whilst this method enables the collection of large amounts of information in a relatively short time frame, it does have the limitation of requiring post-field video analysis to extract the data (Harvey *et al.*. 2013). The processing, interpretation, image storage and retrieval of data can be a laborious task, which may result in a bottleneck of data analysis (Somerton and Glendhill, 2005; Harvey *et al.*, 2013).

As was explained in the Overall introduction, due to the high cost of artificial reefs, there is strong interest in establishing a cost-effective program for monitoring the fish faunas of the two artificial reefs recently deployed off Bunbury and Dunsborough in Geographe Bay in south-western Australia to determine whether citizen science monitoring could provide useful information on the fish fauna of these structures. Such a monitoring program would utilise recreational fishers, acting as citizen scientists, to deploy underwater cameras to collect footage that can be used to assess the characteristics of the fish faunas of these reefs. However, while the use of citizen science in this form would, if it was successful, provide a repository of video footage, data needs to be extracted from the footage collected by the fishers for data analysis. There is thus value in developing a cost-effective means for extracting data from the underwater video footage collected by the fishers. One possible solution that has been suggested is to get university students to extract information from video footage as part of their studies.

Whilst this method may counter the problems associated with data extraction, there is the potential for observer bias, as a number of different students will be involved in extracting data from the footage. Observer bias has the potential to render the data on fish faunas of the artificial reefs obtained via the footage collected by recreational fishers useless, as it could confound differences between observers with real spatial and temporal effects (Thompson and Mapstone, 1998). It is, therefore, important to provide some assessment of the potential for observer bias in extracting data on fish faunas from such footage. The first specific aim of this component of the study was to determine what level of observer bias, if any, is present among the observers when extracting the following information about fishes captured on remotely collected underwater footage; (*i*) the relative abundance (MaxN), (*ii*) species richness and (*iii*) species composition. Since observer bias was detected, the

second aim was to develop a series of recommendations that can be implemented to reduce observer effects in the context of using university students to extract data from underwater video footage collected by recreational fishers.

Materials and methods

Source of data

All underwater video footage employed in this study was collected from the Dunsborough artificial reef during two sampling trips on the 10th and 19th of March 2015 using a Baited Remote Underwater Video (BRUV) system. This is the same data collected by staff from Ecotone Consulting and used in Section 4.

Observers and video analysis

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

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

Previous work by Florisson (2015) and in Section 4 identified significant differences in the composition of fish species depending on whether the camera was collected facing or not facing reef modules. Thus, whilst not being the main focus of this study, this factor was considered and incorporated into the statistical testing.

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

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

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

Statistical analyses

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

Univariate analyses

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

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

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

Multivariate analysis

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

The MaxN for each species in each individual sample was subjected to a fourth root transformation to down weigh the contributions of highly abundant taxa and balance them

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

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

Finally, shade plots were employed to produce a visual display of the abundance matrix of variables (transformed and standardized species counts) against samples (groups of videos). As the PERMANOVA test demonstrated that the species composition differed among both observers and camera position, but that the interaction between these factors was not

significant, the fourth-root transformed MaxN data for each species in each sample was averaged and used to create two data matrices. In the first the transformed data was averaged across the four observers and in the second it was averaged across the two camera positions. The data in these two matrices were standardized and subjected to the Shade plot routine. This produced a visual display of the abundance matrix of variables (transformed and standardized species counts) against samples (either observers or camera positions), where the white represents the absence of taxa in a sample and the intensity of grey-scale shading is linearly proportional to 'abundance' (Clarke *et al.*, 2014a). The taxa (*y* axis of the shade plot) are ordered to optimise the seriation statistic (ρ) by non-parametrically correlating their resemblances to the distance structure of a linear sequence (Clarke *et al.*, 2014b). This seriation was constrained by the family of the taxa so that taxa within the same family, regardless of their similarity to one another, were kept together and separate from other families. The order of both the samples (displayed on the *x* axis) in the case of the shade plot showing observers were determined independently by the results of a group-average hierarchical agglomerative cluster analyses employing resemblance matrices defined using Whittaker's index of association (Whittaker 1952; Valesini *et al.*, 2014).

Results

The four observers identified a combined total of 46 taxa to species, three to genus and three to family (Table 7). The greatest number of taxa identified by a single observer was 36 (Observer 4), while the lowest number of taxa identified was 26 (Observer 3). Observer 4 recorded the highest total mean MaxN count, *i.e.* 34.1, while the mean MaxN counts for the other three observers ranged from 27 and 30 (Table 7).

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

Thirteen of the species detected by Observers 1, 2 and 4 were not identified by Observer 3, including species such as *T. novaezelandiae*, *Parequula melbournensis*, and *Austrolabrus maculatus*. However, Observer 3 identified eight species that were not detected by any other observer, including *Caesioscorpis theagenes* and *Labroides dimidiatus. Meuschenia freycineti* was only identified by Observer 1, and Observer 2 was the only observer to identify *Eubalichthys mosaicus, Cheilodactylus nigripe, Halichoeres brownfieldi* and *Lagocephalus lunaris.*

Table 7. Species table showing the mean MaxN (X) and standard error (SE) of each of the 52 fish and cephalopod taxa recorded by each of four observers who analysed the same five minute portion of the same 30 videos recorded using BRUV on the Dunsborough artificial reef. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5 % to abundance recorded by any observer are shaded in grey.

Species	Total X	SE	%	R	Obs1 X	SE	%	R	Obs2 X	SE	%	R	Obs3 X	SE	%	R	Obs4 X	SE	%	R
Pseudocaranx spp.	17.97	0.05	60.1	1	17.33	1.79	60.39	1	16.97	1.58	57.51	1	18.20	2.04	66.59	1	19.37	2.41	56.79	1
Coris auricularis	3.83	0.01	12.8	2	3.97	0.57	13.82	2	4.07	0.63	13.79	2	3.27	0.56	11.95	2	4.00	0.62	11.73	2
Neatypus obliquus	1.63	0.26	5.43	3	1.40	0.47	4.88	3	1.77	0.57	5.99	3	1.57	0.49	5.73	3	1.77	0.56	5.18	4
Trachurus novaezelandiae	1.02	0.98	3.40	4	0.63	0.47	2.21	6	0.43	0.43	1.47	8					3.00	2.67	8.80	3
Anoplocapros amygdaloides	0.83	0.08	2.79	5	0.87	0.16	3.02	4	0.80	0.15	2.71	4	0.87	0.17	3.17	4	0.80	0.18	2.35	6
Parequula melbournensis	0.66	0.14	2.20	6	0.77	0.23	2.67	5	0.80	0.29	2.71	4					1.07	0.38	3.13	5
Austrolabrus maculatus	0.43	0.07	1.42	7	0.53	0.16	1.86	7	0.53	0.16	1.81	6					0.63	0.17	1.86	7
Pempheris klunzingeri	0.38	0.11	1.28	8	0.33	0.22	1.16	9	0.53	0.28	1.81	6	0.33	0.14	1.22	7	0.33	0.22	0.98	8
Seriola hippos	0.28	0.69	0.92	9	0.37	0.11	1.28	8	0.37	0.11	1.24	10	0.03	0.03	0.12	21	0.33	0.11	0.98	8
Diodon nicthemerus	0.21	0.05	0.70	10	0.20	0.09	0.70	12	0.33	0.13	1.13	11					0.30	0.13	0.88	10
Trachinops noarlungae	0.21	0.11	0.70	10	0.27	0.27	0.93	11	0.27	0.27	0.90	12					0.30	0.27	0.88	10
Myliobatis australis	0.20	0.04	0.67	12	0.20	0.07	0.70	12	0.20	0.07	0.68	14	0.20	0.07	0.73	12	0.20	0.07	0.59	14
Upeneichthys vlamingii	0.19	0.06	0.64	13	0.30	0.18	1.05	10	0.23	0.12	0.79	13				27	0.23	0.12	0.68	12
Trygonorrhina fasciata	0.17	0.04	0.56	14	0.20	0.09	0.70	12	0.20	0.09	0.68	14	0.10	0.07	0.37	16	0.17	0.07	0.49	15
Glaucosoma hebraicum	0.14	0.04	0.47	15	0.13	0.06	0.46	16	0.17	0.08	0.56	18	0.13	0.08	0.49	14	0.13	0.06	0.39	16
Chromis klunzingeri	0.13	0.07	0.42	16				33	0.43	0.28	1.47	8					0.07	0.05	0.20	22
Sepioteuthis australis	0.12	0.04	0.39	17	0.13	0.08	0.46	16	0.20	0.10	0.68	14	0.03	0.03	0.12	21	0.10	0.07	0.29	19
Dasyatis brevicaudata	0.11	0.03	0.36	18	0.10	0.06	0.35	19	0.13	0.06	0.45	19	0.07	0.05	0.24	17	0.13	0.06	0.39	16
Parapercis haackei	0.11	0.02	0.36	18	0.17	0.10	0.58	15	0.03	0.03	0.11	28					0.23	0.10	0.68	12
Caesioscorpis theagenes	0.11	0.08	0.36	18									0.43	0.33	1.59	5				
Labroides dimidiatus	0.10	0.01	0.33	21									0.40	0.24	1.46	6				
Lagocephalus sceleratus	0.10	0.03	0.33	21	0.13	0.06	0.46	16					0.27	0.08	0.98	9				
Cheilodactylus vestitus	0.09	0.01	0.31	23	0.07	0.05	0.23	21					0.23	0.20	0.85	10	0.07	0.05	0.20	22
Chelmonops curiosus	0.08	0.03	0.28	24	0.10	0.07	0.35	19	0.10	0.07	0.34	20	0.03	0.03	0.12	21	0.10	0.06	0.29	19
Nemadactylus macropterus	0.08	0.05	0.28	24									0.33	0.14	1.22	7				37
Ophthalmolepis lineolatus	0.07	0.03	0.22	26	0.03	0.03	0.12	26	0.20	0.09	0.68	14					0.03	0.03	0.10	27
Pentapodus vitta	0.06	0.06	0.20	27									0.23	0.23	0.85	10				
Pentaceropsis recurvirostris	0.06	0.02	0.20	27	0.07	0.05	0.23	21	0.07	0.05	0.23	23	0.03	0.03	0.12	21	0.07	0.05	0.20	22
Chrysophrys auratus	0.05	0.020	0.17	29	0.07	0.05	0.23	21	0.07	0.05	0.23	23					0.07	0.05	0.20	22
Pictilabrus laticlavius	0.04	0.014	0.14	30									0.17	0.07	0.61	13				
Anoplocapros lenticularis	0.04	0.018	0.14	30					0.10	0.06	0.34	20	0.03	0.03	0.12	21	0.03	0.03	0.10	27
Parapercis ramsayi	0.04	0.037	0.14	30	0.07	0.05	0.23	21	0.03	0.03	0.11	28					0.07	0.05	0.20	22
Nemadactylus valenciennesi	0.03	0.037	0.11	33									0.13	0.06	0.49	14				
Parazanclistius hutchinsi	0.03	0.016	0.11	33	0.03	0.03	0.12	26	0.03	0.03	0.11	28	0.03	0.03	0.12	21	0.03	0.03	0.10	27
Lagocephalus spp.	0.03	0.016	0.11	33													0.13	0.06	0.39	16
Suezichthys cyanolaemus	0.03	0.296	0.08	36	0.03	0.03	0.12	26	0.03	0.03	0.11	28					0.03	0.03	0.10	27

Table 7 cont. Species table showing the mean MaxN (X) and standard error (SE) of each of the 52 fish and cephalopod taxa recorded by each of four observers who analysed the same five minute portion of the same 30 videos recorded using BRUV on the Dunsborough artificial reef. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5 % to abundance recorded by any observer are shaded in grey.

| Species	X	SE	%	R	X	SE	%	R	X	SE	%	R	X	SE	%	R	X	SE	%	R
Pseudolabrus biserialis	0.03	0.062	0.08	36	0.03	0.03	0.12	26	0.03	0.03	0.11	28					0.03	0.03	0.10	27
LABRIDAE spp.	0.03	0.018	0.08	36													0.10	0.06	0.29	19
Neosebastes pandus	0.03	0.014	0.08	36	0.03	0.03	0.12	26	0.03	0.03	0.11	28					0.03	0.03	0.10	27
Lagocephalus lunaris	0.03	0.014	0.08	36					0.10	0.06	0.34	20								
Trygonoptera mucosus	0.03	0.008	0.08	36	0.07	0.05	0.23	21									0.03	0.03	0.10	27
Cheilodactylus nigripes	0.02	0.016	0.06	42					0.07	0.05	0.23	23								
Halichoeres brownfieldi	0.02	0.017	0.06	42					0.07	0.05	0.23	23								
Achoerodus gouldii	0.02	0.014	0.06	42									0.07	0.07	0.24	17				
Eubalichthys mosaicus	0.02	0.012	0.06	42					0.07	0.05	0.23	23								
Platycephalus spp.	0.02	0.012	0.06	42	0.03	0.03	0.12	26									0.03	0.03	0.10	27
Scorpaenodes smithi	0.02	0.012	0.06	42									0.07	0.05	0.24	17				
Trygonoptera ovalis	0.02	0.014	0.06	42									0.07	0.05	0.24	17				
Meuschenia freycineti	0.01	0.008	0.03	49	0.03	0.03	0.12	26												
MONACANTHIDAE spp.	0.01	0.008	0.03	49													0.03	0.03	0.10	27
OSTRACIIDAE spp.	0.01	0.008	0.03	49													0.03	0.03	0.10	27
Trygonoptera personata	0.01	0.012	0.03	49					0.03	0.03	0.11	28								
Number of taxa	**52**				**32**				**34**				**26**				**36**			
Total mean MaxN	**29.9**				**28.70**				**29.50**				**27.33**				**34.10**			

Univariate analysis

Whilst there was slight variation, PERMANOVA showed no significant difference between either the mean number of species (Table 8a; Fig. 28a), or the relative abundance of species (Table 8b; Fig. 28b) identified per sample between observers. Significant differences were detected between the number of species on reef facing and not reef facing camera footage (Table 8a). Observers 2 and 4 identified the most species per sample, averaging just over 6 species, whilst the lowest mean number of species identified per sample was 5 (Observer 3). The highest mean abundance was recorded by Observer 4, with a mean of ~30, with the lowest recorded by Observer 3 with a mean of ~26.

Table 8. Mean squares (MS), Pseudo-*F* (p*F*) values and significance levels (*P*) for a two-way PERMANOVA test on (a) number of species, between observers and camera position and (b) abundance (total MaxN) counts between observers and camera position.

(a) Number of species	df	MS	*pF*	*P*
Observer	3	11	1.88	0.148
Position	1	64.53	11	0.003
Residual	112	656.9		

(b) Abundance	df	MS	*pF*	*P*
Observer	3	0.045	0.68	0.55
Position	1	0.089	1.35	0.23
Residual	112	7.41		
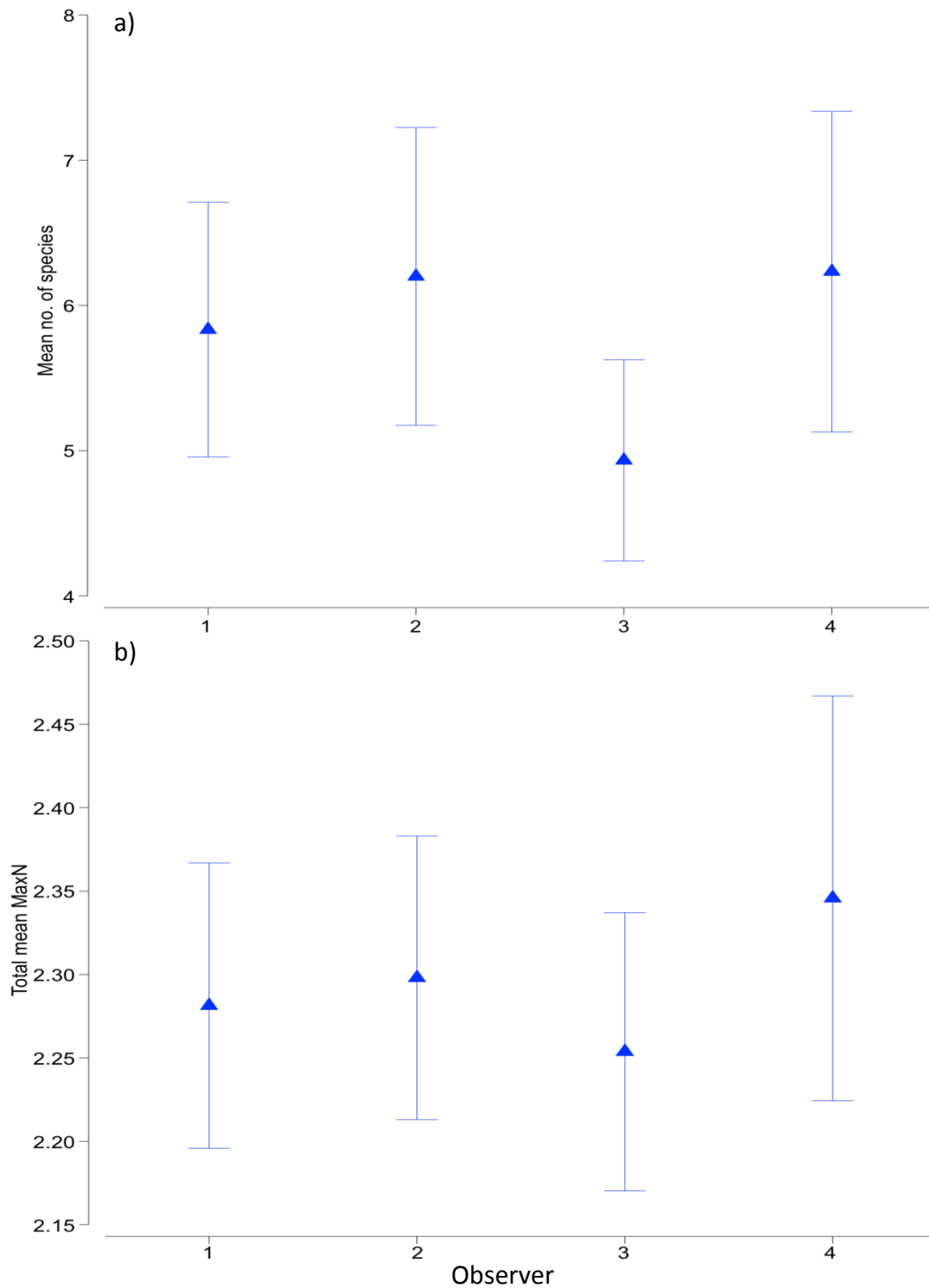
Fig. 28. (a) Mean number of species identified per sample by each observer and (b) the fourth root transformed, total mean MaxN identified per sample by each observer. Error bars represent 95% confidence intervals.

Multivariate analysis

PERMANOVA demonstrated that the composition of species identified by the four observers differed significantly (Table 9). ANOSIM found that the data collected by Observers 1, 2 and 4 were not significantly different, but were invariably significantly different to the data collected by Observer 3 (Table 10). These trends are highlighted in the 3-dimentional nMDS plot that shows a clear grouping of samples from Observer 3, whilst the remaining three observer samples show no clear pattern (Fig. 29). Significant differences in species composition were also detected by observers between footage from reef facing and not reef facing samples (Table 10). This is shown visually in the nMDS centroid plot that shows clear grouping of facing and not facing samples by all observers as well as a close grouping between Observers 1, 2 and 4 (Fig. 30).

Table 9. Mean squares (MS), Pseudo-*F* (p*F*) values and significance levels (*P*) for a two-way PERMANOVA test on the species composition between observers and camera position.

Species composition	df	MS	*pF*	*P*
Observer	3	3145	2.46	0.002
Position	1	11115	8.69	0.001
Observer x Position	3	1	0.66	0.83
Residual	112	143000		

Table 10. Pairwise *R* and significance levels (*P*) for ANOSIM analysis results of fish species composition among observers. Significant differences are highlighted in bold.

Observer	R	*P*
1 vs 2	-0.049	0.976
1 vs 3	**0.141**	**0.001**
1 vs 4	-0.055	0.986
2 vs 3	**0.174**	**0.001**
2 vs 4	-0.042	0.949
3 vs 4	**0.185**	**0.001**
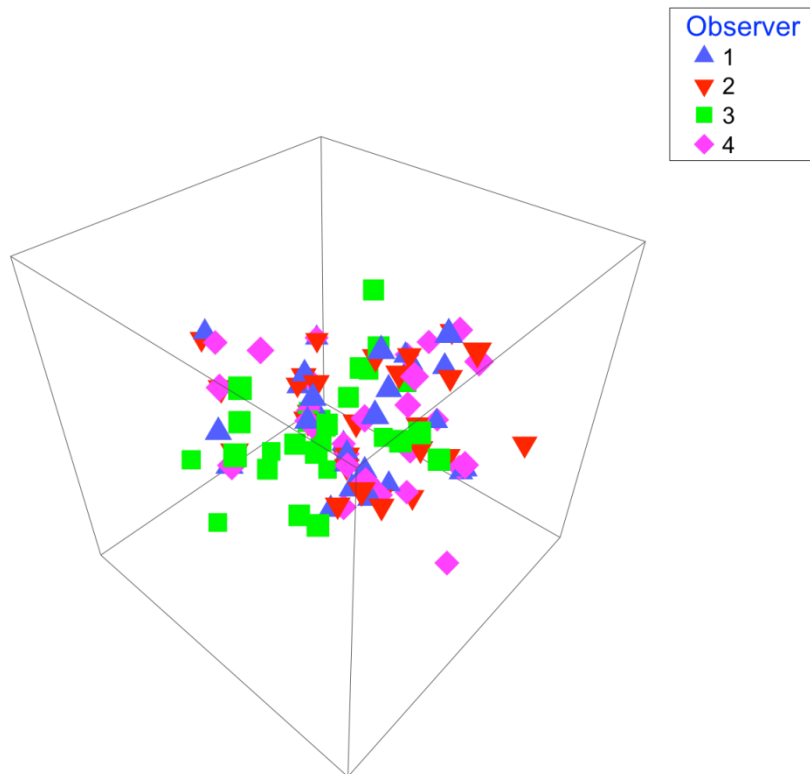
Fig. 29. 3D nMDS plot constructed using the Bay-Curtis Similarity matrix, using fourth root transformed data of the MaxN for each species in each sample coded by observer.
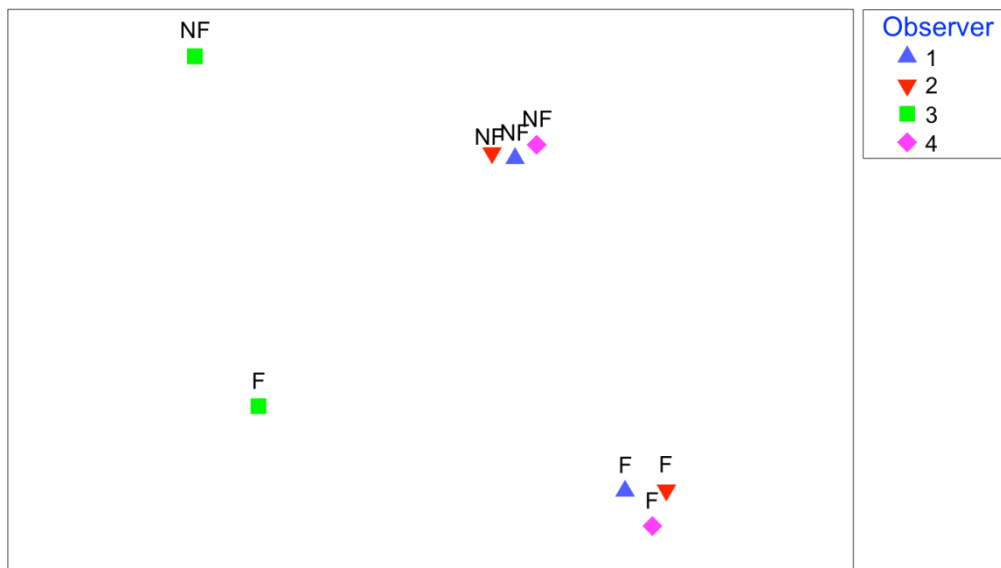

Fig. 30. A 2d centroid nMDS ordination plot, derived from distance among centroid matrices constructed from the Bay-Curtis Similarity matrix, created using fourth root transformed data of the MaxN for each species in each sample coded for observer.

A shade plot showing the mean MaxN of species identified highlights trends in species and families identified between the four observers (Fig. 31). *Pseudocaranx* spp., *Anoplocapros amygdaloides* and *Coris auricularis,* dominated the data set and were found in similarly high abundance by all observers. Other species found in similar abundance by all four observers were *Neatypus obliquus, Myliobatis australis*, and *Glaucosoma hebraicum*. A hierarchal conglomerative cluster analysis of the similarity between observers showed that the species composition of Observers 1 and 4 had the highest similarity (91%). This was followed by Observer 2, who showed a similarity of 89% to Observers 1 and 4, whilst Observer 3 showed the lowest similarity to the other observers with a species composition similarity of 70% (Fig. 31). Variation between Observer 3 and the other observers was found to be highest for taxa within the families Labridae, Cheilodactylidae and Monacanthidae (Fig. 31).

As with the shade plot comparing the species composition between observers, a shade plot showing the species composition between reef facing and not reef facing footage highlights that a small number of species dominated the data set and comprised the majority of individuals (Fig. 32). Overall the relative abundance and number of species was found to be higher on footage that was collected facing the reef modules. Whilst the most abundant species *Pseudocaranx* spp.*,* was found to be in similar densities on both facing and not facing footage, *Coris auricularis* and *Anoplocapros amygdaloides* were found in higher densities on facing footage (Fig. 32).
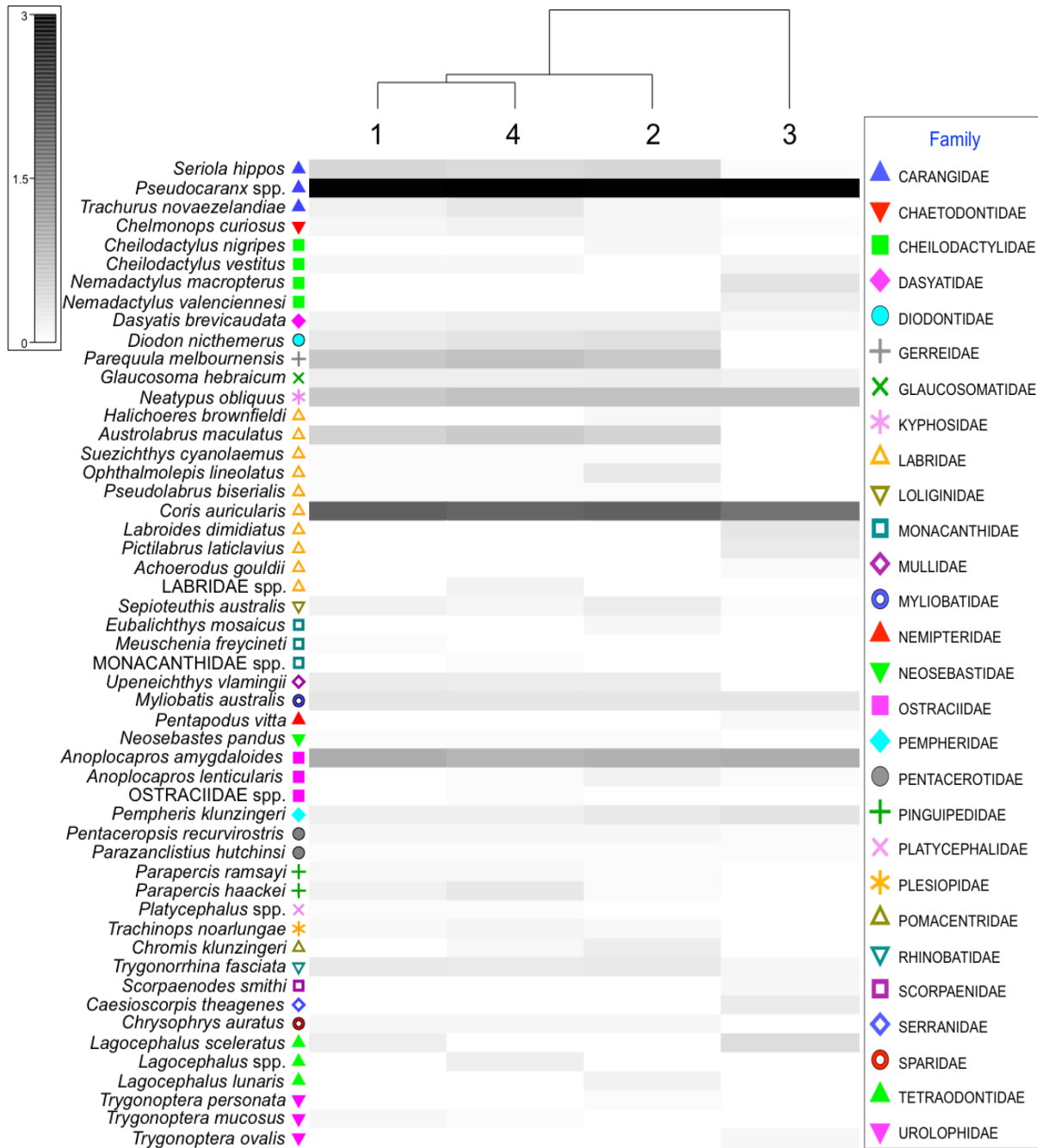
Fig. 31. Shade plot illustrating the fourth root transformed relative abundance (MaxN) of species with shading intensity being proportional to abundance. Relative abundance (MaxN) counts are categorized by observer, and species are ordered by their family.
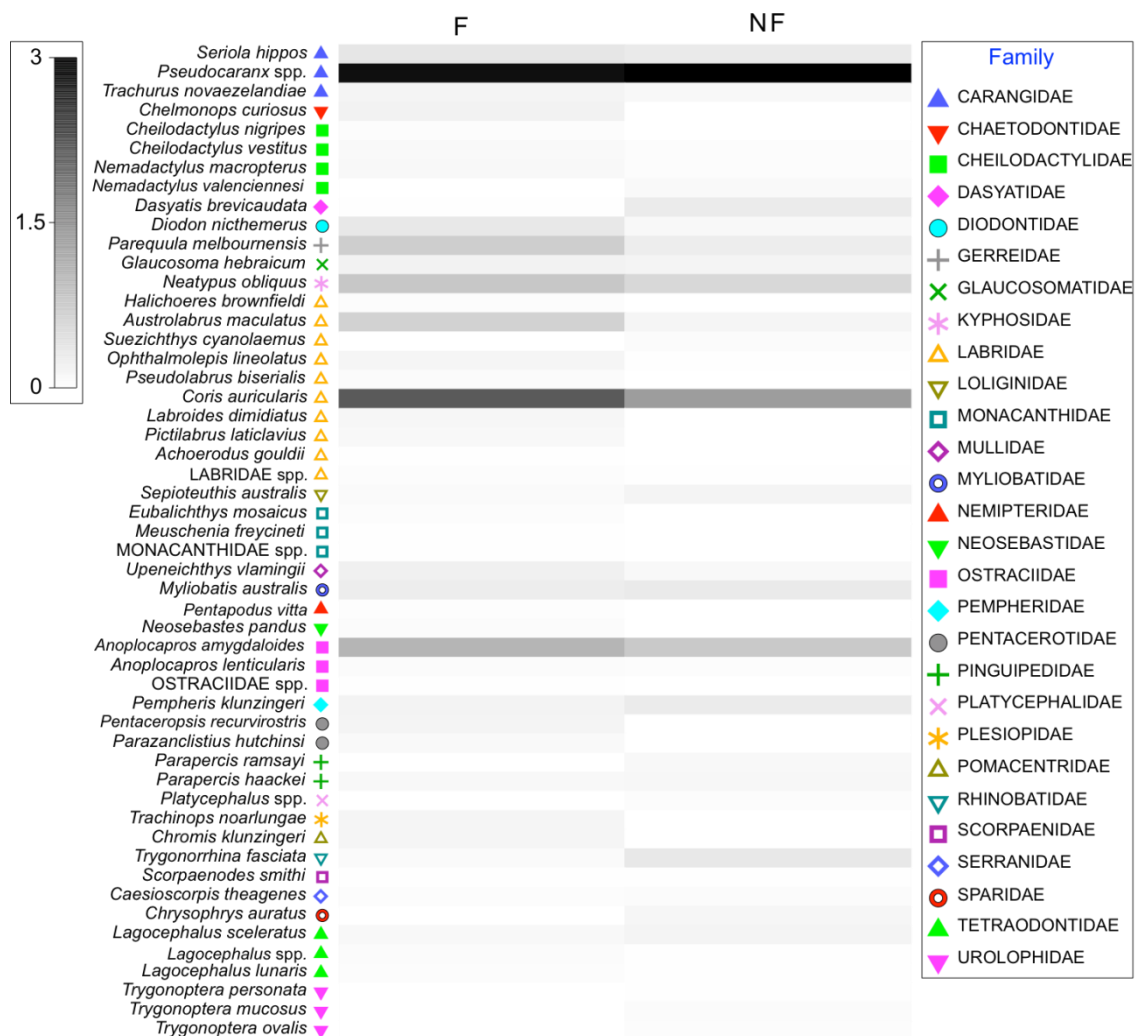
Fig. 32. Shade plot illustrating the fourth root transformed relative abundance (MaxN) of species with shading intensity being proportional to abundance. Relative abundance (MaxN) counts are categorized by facing (F) and not facing (NF) camera positions, and species are ordered by their family.

Discussion

The detection and management of observer bias is key to maintaining the quality of data collected in any monitoring study (Harding *et al.*, 2000; Pattengill-Semmens and Semmens 2003; Williams *et al.*, 2006). This study has provided a preliminary assessment of the extent of bias among four observers in extracting data on the abundance and composition of fish from underwater footage of an artificial reef deployed off Dunsborough. The study found that, whilst the fish fauna data extracted from the footage by three of the observers were similar, there was significant variation between the results obtained by these three

observers and those obtained by a fourth observer (Observer 3). Whilst the difference between the numbers of species or the number of individuals identified among the four observers were not statistically significant, there was a significant difference in the overall species composition. This indicates that individual fish on the footage were misidentified in some cases, particularly by Observer 3, rather than unsighted.

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

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

Although this study has focused primarily on the detection of observer bias, it has also been noted that similar to previous work by Florisson (2015), all observers identified significant differences between the species composition on facing and not facing footage. This is likely due to habitat preference between different species, as well as the increased availability of food and shelter provided by the artificial reefs. Previous studies have shown that species abundance was greater on artificial reefs than the surrounding area and it is possible that

the additional shelter and habitat created by the Geographe Bay artificial reefs promotes an increased abundance of fish species (Sherman *et al*. 2002, Folpp *et al*. 2011). However the limited data available means only assumptions can be made, and further investigation is required to determine the effects that camera apposition has on assessing the fish fauna of artificial reefs and if this should be taken into consideration in future monitoring.

Reducing observer bias in future studies

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

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

Providing observers with the opportunity to have species identifications reviewed by a more experienced observer/taxonomist would help to increase the quality of data. One of the key benefits of using underwater video is the ability to view the footage multiple times if ever there is confusion with the identification of a species. This can be easily achieved by having observers take snapshots from the footage of a species they were unclear on the

identification of and send it to a reviewer. These images could then be used to create a database over time that could be used as a reference in future monitoring of reefs in southwest Western Australia.

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

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

Section 6: Analysis of a cost-effective artificial reef monitoring method to detect fish faunal differences among reefs

Overview

This section of the report is based on Chapter 5 in Thomas Bateman's honours thesis, which was completed in November 2015. The results were important in the context of the broader study because they demonstrate that the video footage provided by the Baited Remote Underwater Video systems was of sufficient quality to detect differences in the fish assemblages utilising the artificial reefs in Bunbury and Dunsborough. Significant differences were detected in mean number of species, total MaxN and species composition, with in general a more abundant and diverse fauna on the Dunsborough reef. Although preliminary, the results will form the basis of more detailed comparisons of the fish faunas undertaken in during the *Reef Vision* project.

Introduction

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

A frequent stumbling block encountered in many monitoring programs is the collection of sufficient data over large temporal and spatial scales when resources are limited (Baird *et al.*, 2000). One solution to this is the use of volunteers to collect information. The use of volunteers, referred to as "citizen science", to collect biological data is well established in

both marine and terrestrial environments (Viswanathan *et al.*, 2004; Wiber *et al.*, 2004; Conrad and Daoust 2008; Conrad and Hilchey, 2011; Gollan *et al.*, 2012). The benefit of citizen science is that it allows a portion of monitoring costs to be borne by the volunteers, and has shown to increase stewardship of the resource (Pattengill-Semmens and Semmens, 2003). However, with all volunteer based projects, monitoring regimes need to be developed that are both simple and effective, to ensure reliable data collection (Harding *et al.*, 2000).

As detailed in the Overall introduction, there is a need to develop cost-effective monitoring regime for the fish faunas of artificial reefs in Western Australia. Initial trials involved the use of rotating remote underwater cameras, which provided a live feed of the video footage being collected to avoid collision with reef modules whilst monitoring (see Section 2). Analysis of the footage collected using these cameras, however, showed that this equipment was ineffective at monitoring the fish fauna of the artificial reefs due to the poor quality of the video captured. This led to trial the use of Baited Remote Underwater Video (BRUV) systems developed by staff from Recfishwest and Ecotone Consulting constructed from low cost materials. The provision on footage by Ecotone Consulting from the Bunbury artificial reef to complement that already captured from the Dunsborough reef (see Section 4) enabled the opportunity to investigate the types of information that can be extracted on the fish fauna of the Dunsborough and Bunbury artificial reefs by analyzing BRUV footage. This data was used to assess the ability of this method for monitoring the fish fauna on the reefs and determine whether the fish assemblages on the Dunsborough and Bunbury artificial reefs differed.

Materials and methods

Study site

This study was conducted on the Bunbury and Dunsborough artificial reefs located in Geographe Bay. Full details of the locations, composition and design of the artificial reefs and on Geographe Bay and its environmental characteristics are given in Section 1.

Source of data

BRUV footage of the Dunsborough and Bunbury artificial reefs was collected from three separate sampling trips. Data collection took place on the 10th and 19th of March 2015 at the Dunsborough reef and the 25th of May 2015 at the Bunbury reef. The BRUV design and video collection methodology were the same at both reefs and thus identical to those described in Section 4.

During the final stages of this thesis, a preliminary species list was provided by the Western Australian Department of Fisheries (DoF), who have been monitoring the artificial reefs using a combination of Diver Operated Video (DOV) and BRUV since the deployment of the reefs in 2013 (see Appendix 6.1). The species list provided by the DoF contains a preliminary list of species that have been identified from six separate monitoring surveys of both of the artificial reefs in Geographe Bay. Due to the short notice in which this information was obtained, it has not been included within the analysis of the results, however, it has been used as comparative data set to assess whether the trends observed in the footage collected by Recfishwest and Ecotone consulting, are mirrored by that of a broader data set.

Video analysis

Prior to analysis, the provided raw videos were coded according to their trip collection date (t), camera number (c), and video data number. For example a video collected on trip one, by camera one, with a video data number of 0001, would be coded (t1c1-0001). Two additional factors were given to each video that indicated the 'reef' that the footage was collected from and the camera 'position' as either facing reefs modules (F) or not facing reef modules (NF). The reason for including camera position as a factor in this study is due to previous work by Florisson (2015) and Section 4 and 5, which identified significant differences between the faunal compositions on footage collected from BRUVs facing towards reef modules and those facing away.

Thirty-three videos were analysed in total, with 24 from Dunsborough (12 facing reef modules, 12 not facing reef modules), and 9 from Bunbury (5 facing reef modules, 4 not

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

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

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

Statistical analyses

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

Univariate analyses

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

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

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

Multivariate analysis

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

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

If the interaction was not significant, or relatively small in relation to the main effects, the matrix was then subjected to a two-way ANOSIM test. ANOSIM was preferred at this stage of the analysis because, unlike PERMANOVA, this test is fully non-parametric and thus more robust, and because the ANOSIM *R*-statistic provides a universal measure of group separation to test for significant interactions between region and position (Lek *et al.*, 2011). The magnitude of the *R* statistic typically ranges between 1, when the compositions of the samples within each group are more similar to each other than to that of any of the samples from other groups, down to ~0, when within-group and between-group similarities do not differ (Clarke *et al.*, 2014b).

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

The order of both the variables and samples were determined independently (*i.e.* the order of variables is not influenced by the order of samples and *vice versa*) by the results of separate a group-average hierarchical agglomerative cluster analyses employing resemblance matrices defined using Whittaker's index of association (Whittaker 1952,

Valesini *et al*. 2014). Species exhibiting similar patterns of abundance across the samples were thus clustered together on the resultant dendrogram (*y* axis of the shade plot), while the samples (displayed on the *x* axis) were ordered by similarities in their 'species' composition. Note that, for clarity, only those taxa that occurred in two of more of the samples (*i.e.* 24 out of 35 taxa) were included in the shade plot.

Results

Mean density of species at artificial reef locations

A total of 35 taxa, from 22 families, including 34 fish and 1 cephalopod, were identified on BRUV footage, with the majority of taxa identified to species level (97%; Table 11). The only taxa that could not be identified to species from the footage were from the genus *Pseudocaranx*. The most specious families on the video footage were Labridae and Carangidae, which were represented by five and three taxa respectively.

Thirty-four of the 35 taxa identified were present on footage from the Dunsborough reef (Table 11). The most abundant taxa identified at the Dunsborough reef were *Pseudocaranx* spp., which represented ~48% of the total abundance. The following most abundant species were *Coris auricularis* and *Trachurus novaezelandiae,* which represented ~15% and ~8% respectively, of the total abundance. A total of 11 taxa were identified on footage from the Bunbury reef. The most abundant species found on this footage was *C. auricularis*, which accounted for ~39% of the total abundance, followed by *Parequula melbournensis* (~31%) and *Neatypus obliquus* (~14%). Neither *Pseudocaranx* spp*.* nor *T. novaezelandiae*, were identified on footage from the Bunbury reef, however both *P. melbournensis* and *C. auricularis* were seen in higher abundance on the Bunbury reef, with mean MaxNs of 3.89 and 4.89 respectively, compared to 1.88 and 4.88 at Dunsborough reef. Of the 35 identified taxa, 23 taxa were restricted to the footage from the Dunsborough reef, whilst only a single species, *Trygonoptera personata*, was restricted to the footage from the Bunbury reef (Table 11).

Table 11. Species table showing the mean MaxN (X) and standard error (SE) of each of the 35 fish and cephalopod taxa recorded using BRUVs on the Dunsborough and Bunbury artificial reefs. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5 % to abundance recorded by any observer are shaded in grey.

Species	Family	Total				Dunsborough				Bunbury			
		X	SE	%	R	X	SE	%	R	X	SE	%	R
Pseudocaranx spp.	CARANGIDAE	14.1	2	47.9	1	19.3	1.8	54.3	1				
Coris auricularis	LABRIDAE	4.48	0.8	15.3	2	4.33	0.7	12.2	2	4.9	2.6	38.6	1
Trachurus novaezelandiae	CARANGIDAE	2.24	1.4	7.65	3	3.08	1.8	8.67	3				
Parequula melbournensis	GERREIDAE	1.88	0.4	6.41	4	1.13	0.3	3.16	5	3.9	1.1	30.7	2
Neatypus obliquus	KYPHOSIDAE	1.67	0.5	5.68	5	1.63	0.6	4.57	4	1.8	0.9	14	3
Anoplocapros amygdaloides	OSTRACIIDAE	0.85	0.2	2.89	6	1.13	0.2	3.16	5	0.1	0.1	0.88	8
Seriola hippos	CARANGIDAE	0.61	0.1	2.07	7	0.58	0.1	1.64	8	0.7	0.2	5.26	4
Austrolabrus maculatus	LABRIDAE	0.48	0.2	1.65	8	0.63	0.2	1.76	7	0.1	0.1	0.88	8
Upeneichthys vlamingii	MULLIDAE	0.36	0.2	1.24	9	0.5	0.3	1.41	9				
Trygonorrhina fasciata	RHINOBATIDAE	0.3	0.1	1.03	10	0.25	0.1	0.7	13	0.4	0.2	3.51	5
Sepioteuthis australis	LOLIGINIDAE	0.27	0.2	0.93	11	0.38	0.3	1.05	10				
Pempheris klunzingeri	PEMPHERIDAE	0.24	0.2	0.83	12	0.33	0.3	0.94	11				
Diodon nicthemerus	DIODONTIDAE	0.24	0.1	0.83	12	0.33	0.1	0.94	11				
Chelmolops curiosus	CHAETODONTIDAE	0.24	0.1	0.83	12	0.21	0.1	0.59	14	0.3	0.2	2.63	6
Myliobatis australis	MYLIOBATIDAE	0.21	0.1	0.72	15	0.21	0.1	0.59	14	0.2	0.2	1.75	7
Parapercis haackei	PINGUIPEDIDAE	0.15	0.1	0.52	16	0.21	0.1	0.59	14				
Dasyatis brevicaudata	DASYATIDAE	0.15	0.1	0.52	16	0.21	0.1	0.59	14				
Chyrosophyrs auratus	SPARIDAE	0.12	0.1	0.41	18	0.17	0.1	0.47	18				
Glaucosoma hebraicum	GLAUCOSOMATIDAE	0.09	0.1	0.31	19	0.13	0.1	0.35	19				

Table 11 cont. Species table showing the mean MaxN (X) and standard error (SE) of each of the 35 fish and cephalopod taxa recorded using BRUVs on the Dunsborough and Bunbury artificial reefs. For each taxon, a percentage contribution (%) and ranking by mean MaxN (R) was calculated. Abundant species *i.e.* those that contributed ≥ 5 % to abundance recorded by any observer are shaded in grey.

| Species	Family	Total X	Total SE	Total %	Total R	Dunsborough X	Dunsborough SE	Dunsborough %	Dunsborough R	Bunbury X	Bunbury SE	Bunbury %	Bunbury R
Cheilodactylus gibbosus	CHEILODACTYLIDAE	0.09	0.1	0.31	19	0.13	0.1	0.35	19				
Pentaceropsis recurvirostris	PENTACEROTIDAE	0.06	0	0.21	21	0.08	0.1	0.23	21				
Parapercis ramsayi	PINGUIPEDIDAE	0.06	0	0.21	21	0.08	0.1	0.23	21				
Meuschenia freycineti	MONACANTHIDAE	0.06	0	0.21	21	0.04	0	0.12	24	0.1	0.1	0.88	8
Aptychotrema vincentiana	RHINOBATIDAE	0.06	0	0.21	21	0.08	0.1	0.23	21				
Choerodon rubescens	LABRIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Chromis klunzingeri	POMACENTRIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Parazanclistius hutchinsi	PENTACEROTIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Aracana aurita	OSTRACIIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Eubalichthys mosaicus	MONACANTHIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Tilodon sexfasciatus	KYPHOSIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Lagocephalus sceleratus	TETRAODONTIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Trygonoptera mucosa	UROLOPHIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Trygonoptera personata	UROLOPHIDAE	0.03	0	0.1	25					0.1	0.1	0.88	8
Suezichthys cyanolaemus	LABRIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Pseudolabrus biserialis	LABRIDAE	0.03	0	0.1	25	0.04	0	0.12	24				
Species		**35**				**34**				**11**			
Mean MaxN		**29**				**36**				**13**			
# Samples		**33**				**24**				**9**			

Number of species

PERMANOVA demonstrated that number of species differed significantly between the footage from the two reefs (Table 12a; Fig. 33a), but not between footage from different camera positions (Table 12a; Fig. 33b), with no significant interaction between reef and position. The mean number of species identified on the Bunbury and Dunsborough reef footage was roughly three and seven. As for camera position the mean number of species identified on reef facing and not reef facing footage was roughly six and five, respectively.

Table 12. Mean squares (MS), Pseudo-*F* (p*F*) values and significance levels (*P*) for a two-way PERMANOVA test on (a) number of species between reef and camera position and (b) abundance (total MaxN) between reef and camera position.

(a) Number of species	df	MS	*pF*	*P*
Reef	1	4.150	18.62	0.001
Position	1	0.163	0.73	0.396
Reef x Position	1	0.013	0.06	0.805
Residual	29	0.223		

| (b) Abundance	df	MS	*pF*	*P*
Reef	1	8.21	37.16	0.001
Position	1	1.49	6.74	0.016
Reef x Position	1	0.92	4.16	0.051
Residual	29	0.221		
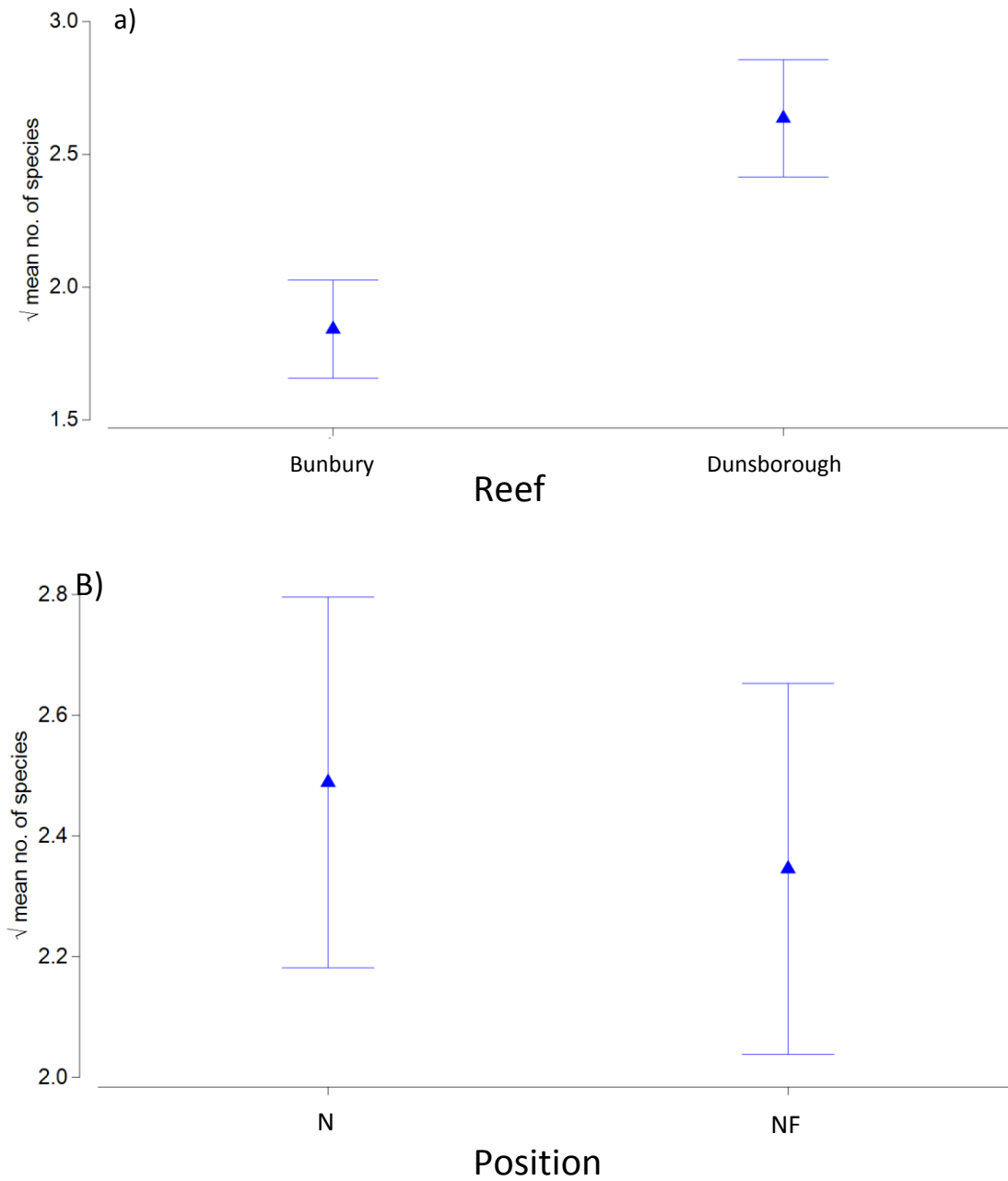
Fig. 33. Mean number of species, square root transformed, recorded at (A) the Bunbury and Dunsborough artificial reefs, and (B) by video footage facing reef modules (F) and not facing reef modules (NF). Error bars represent 95% confidence intervals.

Overall abundance

As for overall density, PERMANOVA identified significant differences between footage from the two reefs (Table 12b; Fig. 34a), and camera position (Table 12b; Fig. 34b). However, it should be noted the error values for relative abundance by position were large. As with the mean number of species, there was no significant interaction between reef and position in regards to abundance of species (Table 12b).
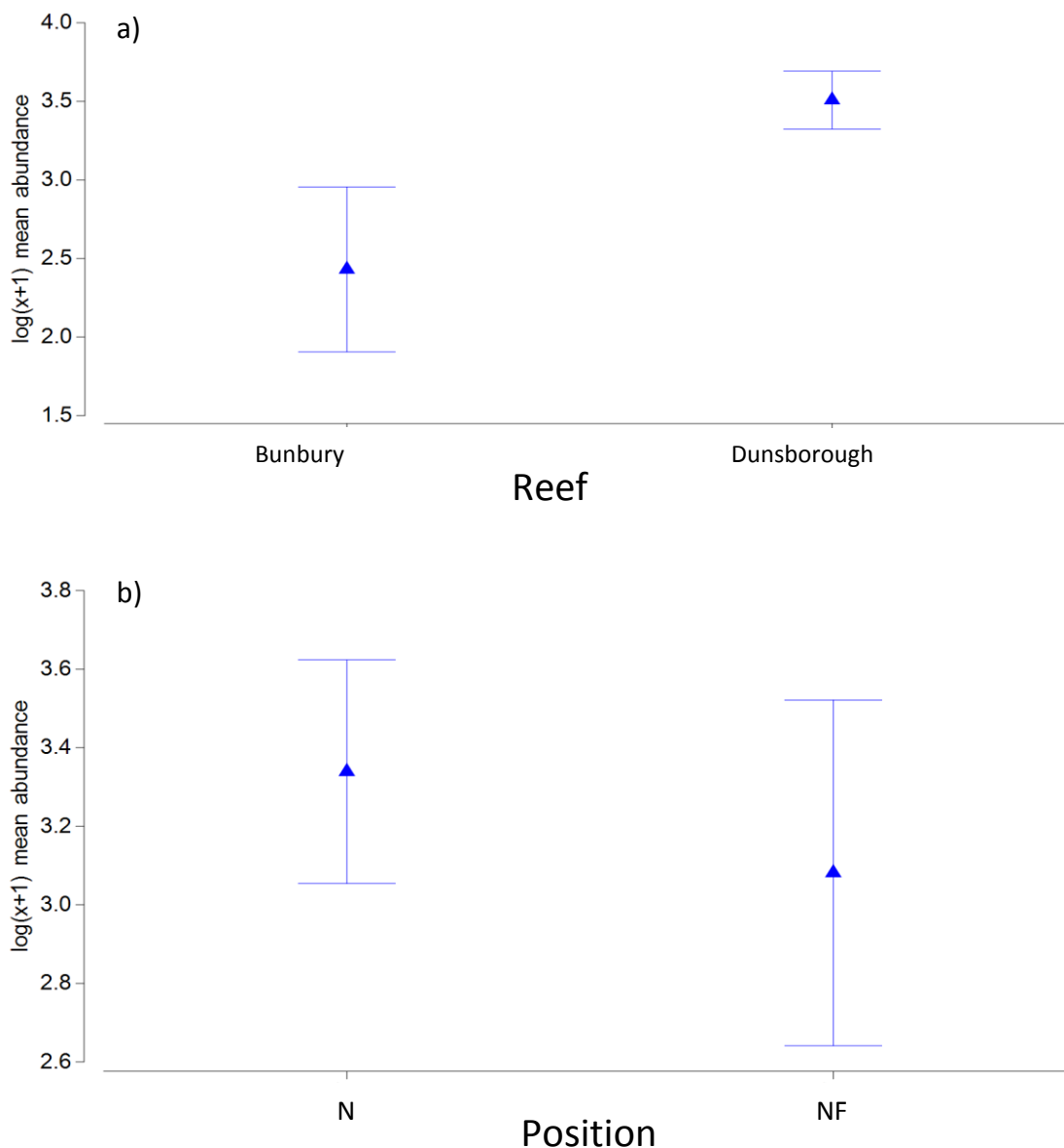
Fig. 34. Mean abundance (MaxN), log(x+1) transformed, of individuals recorded at (a) Bunbury and Dunsborough artificial reefs, and (b) by video footage facing reef modules (F) and not facing reef modules (NF). Error bars represent 95% confidence intervals.

Multivariate analysis

ANOSIM showed that the composition of species differed significantly between footage from the two reefs (Global R = 0.867, P = 0.001), but not for camera position (Global R = 0.071, P = 0.114), with PERMANOVA showing no significant interaction between reef and position (P = 0.817). The nMDS ordination plot, derived from the log(x+1) transformation of densities from all species, show clearly identifiable differences between regions (Fig. 35a), whilst the differences between positions are less clearly observable (Fig. 35b).
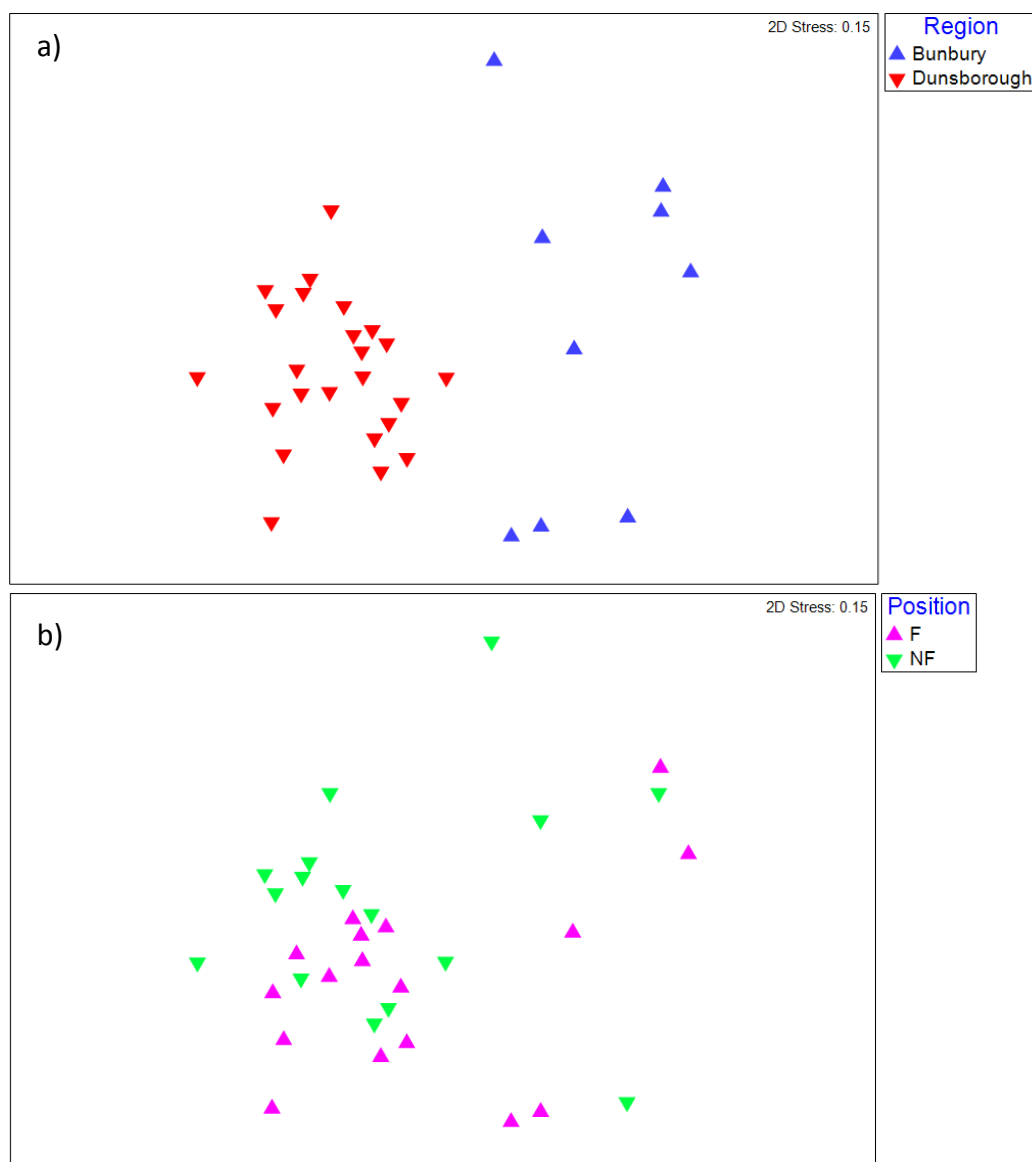

Fig. 35. An nMDS constructed using the Bay-Curtis Similarity matrix, using log(x+1) transformed data of the MaxN for each species in each sample. (a) Plot has been coded for the reef on which the footage was collected, *i.e.* Bunbury or Dunsborough and (b) for the position of the camera, *i.e.* facing towards (F) or away (NF) from the reef.

A shade plot showing the percentage contribution to overall abundance of species that occurred in two or more samples only, highlights trends in individual species between both reef and camera position (Fig. 36). *Parequula melbournensis*, *S. hippos* and *C. auricularis* were found to occur frequently in samples from both reefs and camera positions; however *S. hippos* was found in lower numbers.

Species such as *Anoplocapros amygdaloides* and *Pseudocaranx* spp. were found in high numbers of video samples from the Dunsborough reef, but relatively few at the Bunbury reef. *Trachurus novaezelandiae,* which was the third most abundant species at the Dunsborough site occurred only in three samples, however in very high numbers. The shade plot also shows that species such as *Pentaceropsis recurvirostris* were found only to occur in footage that was collected facing reef modules whilst others such as *Dasyatis brevicaudata* and *Trygonorrhina fasciata*, were far more abundant in footage not facing reef modules.

In regards to recreationally important fish species, whilst *S. hippos* was found in similar abundance regardless of the reef or camera position, *Glaucosoma hebraicum, Chrysophrys auratus* and *Pseudocaranx* spp. were only identified on footage collected from the Dunsborough artificial reef. *Chyrosophyrs auratus* was also only identified on footage that was collected facing away from reef modules (Fig. 36).
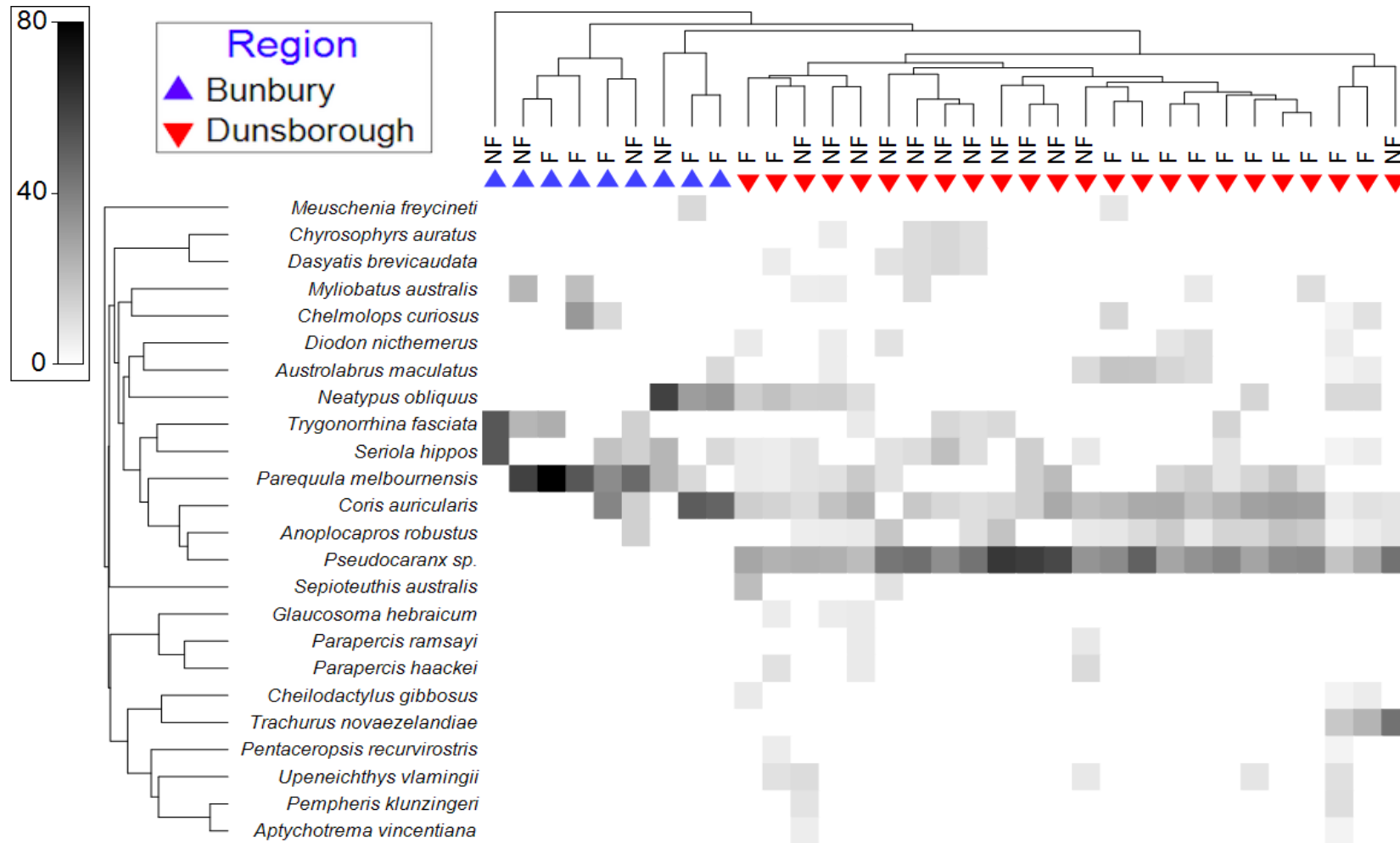
Fig. 36. Shade plot illustrating species that were identified in two or more samples. Data has been log(x+1) transformed and converted to percentage contribution for each sample. Cluster analysis has grouped species and individual video samples by their similarity. Darker shading represents a greater percentage contribution.

Discussion

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

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

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

Trends in the data between reefs

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

numbers, giving it a high MaxN count despite only occurring in a small number of samples (Hutchins and Swainston 1986, Froese and Pauly 2015).

Pseudocaranx spp. on the other hand was found in every video sample at the Dunsborough reef and would likely have been captured had it been present on the Bunbury reef in similar abundance at the time of collecting the footage. As this species has been detected at both regions by previous monitoring (Appendix 6.1), the lack of *Pseudocaranx* spp. on the BRUV footage from the Bunbury reef is likely not due to an absence of the species but rather a lower abundance, and possibly may have been detected with additional sampling. This may also be the case for other recreational target species such as *G. hebraicum* and *C. auratus*, which were only detected at the Dunsborough reef in this study, but have been shown to occur at both reefs (Appendix 6.1).

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

Another significant difference observed between the two reefs was the overall diversity of species. Thirty-five species from 22 families were identified overall, with 34 of these species found at the Dunsborough reef and 11 found at the Bunbury reef. Monitoring by the DoF identified a total of 57 taxa from six monitoring surveys, 25 of which were not recorded on the footage collected by Recfishwest and Ecotone consulting (Appendix 6.1). Of the total number of species identified by the DoF, 44 and 38 were detected at the Dunsborough and Bunbury reefs, respectively, using a combination of both BRUVs and DOV, with 31 taxa identified at both reefs using only BRUVs (Appendix 6.1). This indicates that whilst sampling

was fairly effective at the Dunsborough reef, the lack of footage collected from the Bunbury reef may not have provided an accurate representation of the species composition on the reef.

Trends in data between camera direction

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

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

Recommendations for future study

One of the major factors likely to influence estimates of fish abundance and diversity is the length of time that the BRUV is positioned on the seafloor to record footage, known as the soak time (Gladstone *et al*., 2012; Harasti *et al*., 2015). Previous studies using BRUVs have generally employed soak times between 30-60 minutes with longer times recommended to

attract more 'delayed reaction' species (Stobart *et al.*, 2007; Gladstone *et al.*, 2012; Harvey *et al.*, 2013). Increasing the soak time of BRUVs does, however, add extra costs, as this increases the time need to collected samples and analyze footage.

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

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

Although monitoring by the DoF has not looked at the differences between facing and not facing footage, they have detected significant differences in species composition and abundance on different clusters of reef modules (Paul Lewis; Department of Fisheries WA pers.com. 2015). Variation between the clusters may be caused by a range of differences in ocean currents and sedimentation levels between exposed and protected reef modules (Pais *et al.*, 2007). Haphazard dropping of BRUVs has been successfully used in the past to monitor fish assemblages, but it limits the amount of spatial analysis that can be done

(Cappo and Brown, 1996; Westera *et al*., 2003). By modifying the deployment method to ensure each cluster of modules is sampled separately and assigning each sample with a cluster code depending on its location (*i.e.* North cluster, South-West cluster etc.), analysis of the variation between clusters can be done in much the same way this study has compared the fish assemblages of the two artificial reefs.

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

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

Appendix 6.1. Artificial reef fish species list

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

Species	Dunsborough	Bunbury
Anoplocapros amygdaloides	BRUV / DOV	BRUV / DOV
Anoplocapros lenticularus	BRUV	DOV
Apogon victoriae	DOV	BRUV / DOV
Aptychotrema vincentiana	BRUV	
Arcana aurita	BRUV / DOV	BRUV / DOV
Achoerodus gouldii	BRUV	
Aulohalaelurus labiosus		BRUV
Austrolabrus maculatus	BRUV / DOV	BRUV / DOV
Caesioscorpis theagenes	BRUV / DOV	DOV
Cheilodactylus gibbosus	DOV	BRUV / DOV
Chelmolops curiosus	BRUV / DOV	BRUV / DOV
Choerodon rubescens		BRUV / DOV
Chromis klunzingeri		DOV
Chrysophrys auratus		BRUV
Coris auricularis	BRUV / DOV	BRUV / DOV
Dactylophora nigricans	BRUV	
Dasyatis brevicaudata	BRUV	BRUV
Diodon nicthemerus	BRUV / DOV	
Eubalichthys mosaicus	BRUV	
Eupetrichthys angustipes	DOV	BRUV / DOV
Glaucosoma hebraicum	DOV	
Halichoeres brownfieldii		DOV
Helcogramma decurrens		DOV
Heniochus acuminatus	DOV	
Hypoplectrodes nigroruber		DOV
Meuschenia freycineti	BRUV	
Mustelus antarcticus	BRUV	
Myliobatus australis	BRUV	BRUV
Neatypus obliquus	BRUV / DOV	BRUV / DOV
Neosebastes pandus	BRUV	
Notolabrus parilus		BRUV / DOV
Ophthalmolepis lineolatus		BRUV
Parapercis haackei	DOV	DOV

Appendix 6.1 cont. Artificial reef fish species list

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

Species	Dunsborough	Bunbury
Paraplotosus albilabris	BRUV	
Parapriacanthus elongatus	DOV	
Parequula melbournensis	BRUV	BRUV / DOV
Paristiopterus gallipavo	BRUV / DOV	BRUV
Parma mccullochi	DOV	
Parupeneus crysopleuron	BRUV	
Pentapodus vittae		BRUV
Pempheris klunzingeri	BRUV / DOV	BRUV / DOV
Platycephelus sp.	BRUV	BRUV
Platycephelus speculator	BRUV	BRUV
Platycephelus longispinis	BRUV	
Pseudocaranx sp.	BRUV / DOV	BRUV
Pseudocaranx dentex		BRUV
Pseudolabrus biserialis	DOV	
Pseudorhombus jenynsii		BRUV
Seriola hippos	BRUV / DOV	BRUV
Siganus sp.		BRUV / DOV
Tilodon sexfasciatus	BRUV	BRUV
Trachinops noarlungae	DOV	
Trachurus novaezelandiae	BRUV / DOV	
Trygonoptera personata	DOV	BRUV
Trygonorrhina fasciata	BRUV	BRUV
Upeneichthys vlamingii	DOV	BRUV / DOV
Urolophus sp.	BRUV	
Total no. of species	**44**	**38**
Total no of species detected by BRUV	**31**	**31**

Section 7: Conclusions and recommendations for future citizen science projects monitoring artificial reefs

The overriding goal of this project was to assess the feasibility of using recreational fishers as citizen scientists to assist with the monitoring of the fish assemblages on two purpose-built artificial reefs deployed off Dunsborough and Bunbury in Geographe Bay in 2013.

The overall results of this project suggest that it should be possible to use recreational fishers to assist with monitoring of the fish assemblages of the Dunsborough and Bunbury reefs. Using fishers to assist with the monitoring of these reefs will have many significant advantages, including the provision of important, but cost-effective, data on the fish assemblages of these reefs and the promotion of community stewardship of the reefs.

The overall amount of footage of the artificial reefs collected by recreational fishers in this project was very limited. This was many due to problems with the quality of the footage obtained from the live action cameras provided to fishers and fisher engagement during the first phase of the project. The second phase of the project addressed these problems by testing the suitability of video footage obtained from different camera and developing a more suitable method for managing the project and volunteers, which is now being successfully implemented in *Reef Vision*.

The results of this project have demonstrated that high quality video footage of the fish assemblages of the Dunsborough and Bunbury artificial reefs can be obtained using a small Baited Remote Underwater Video (BRUV) system, equipped with a GoPro Hero 4^{TM} camera, designed by Ecotone Consulting. Furthermore, preliminary analyses, based on a limited amount of BRUV footage that was supplied by Ecotone Consulting, showed evidence of the presence of a broad range of fish species, including some key recreational species, on the artificial reefs, although not as many as revealed by more intensive sampling using a range of methodologies undertaken by the Department of Fisheries WA. Preliminary statistical analyses also indicated the presence differences in the fish assemblages of the Bunbury versus the Dunsborough reefs, based on the BRUV footage. A number of the BRUVs have

now been given to recreational fishers for the reef monitoring, as part of the Reef Vision project.

This project has developed a number of key recommendations regarding the management of recreational fishers (citizen scientists) in an artificial reef monitoring project, as follows.

- Both traditional and social media should be use to recruit volunteers as in combination these are likely to reach a wider audience.

- Social media should be used to facilitate regular communication between project managers and recreational fishers and among fishers. Social media also provides a platform where more experienced volunteers (local champions) can assist new volunteers.

- A workshop, where project managers and recreational fishers can meet in person should be held at the start of the project. This will facilitate a two-way exchange of information between project managers and fishers, and also among fishers, and engender a sense of belonging to the project.

- The project should screen potential volunteers for their suitability to reduce volunteer attrition rates.

- The project should recruit more volunteers than are initially needed to accomodate volunteer attrition.

- Monitoring protocols should be clear,concise and simple to decrease volunteer attrition rates. This should also help to reduce spatial and temporal biases in the data extracted from the footage.

- Updates on the results of the monintoring should be provided to the recreational fishers regularly to reinforce the value of the project and the important role that the fishers are playing in the project.

The project has also developed key recommendations regarding the methodology and technology that would be suitable for a citizen science approach to monitoring the artificial reefs in Geographe Bay, as follows

- Cameras should be adequately trialled for ease of use, safety issues, performance, data storage capabilities and the quality of the footage. Preliminary trials should be conducted by the Project Managers, but recreational fishers should be used in the final stages of the trial to simulate real monitoring conditions.

- Clear written instructions on how to use the cameras must be provided to fishers.

- Small action cameras, such as the GoPro^{TM}, are recommended because they provide a cost-effective means of obtained high quality .

- Simplified BRUVs are recommended over drifting rotational cameras attached to the vessel, because they are easier to operate and also less likely to get entangled in the modules.

- The sampling regime should consist of randomised and standardised squares on a grid, of which fishers are allocated specific squares to monitor with given boundary co-ordinates. This will reduce spatial and temporal bias as well as bias related to sampler objectivity, while increasing the ease of sampling for the fishers.

- If possible, the project need to run for a period of several years or more to reduce the impacts of unforeseen circumstances, such as unseasonal weather and delays in permits and delivery of equipment. This is also essential to properly document any temporal trends in the fish assemblages on the reef.

While there is much to be gaining by using a citizen science approach to monitor the fish assemblages of artificial reefs in Geographe Bay and similar, it is also important to highlight some of the limitations of using this approach, as is discussed below.

For a citizen science approach to be effective, data on the footage collected by the recreational fishers must be recorded, analysed and the results documented in a way that is suitable for dissemination to the participating fishers and also a broader audience.   This adds another layer of management to the project and, in part, requires specialist skills. Specifically, recording of the fish data from the footage obtained from recreational fishers will involve many hours of identifying and counting fish and therefore must rely on volunteers, *i.e.* it would be cost prohibitive to pay a professional scientist to do this. The results of this project showed the potential for errors to occur when data were recorded by University-level students with training in marine science. In order to address potential observer bias, it is necessary to ensure that volunteers are properly trained, that instructions are clear and ideally that footage is scored independently be two different people so potential errors can be identified. Once the data have been accurately recorded, a high level of skill is required to statistically analyse these data and, to a lesser extent, document the results in a suitable form for dissemination. This work must be done by someone with specialist training and the project needs to take into account costs of employing this specialist.

There are several attributes that may help future projects determine the approach to utilising citizen science. These attributes can be seen in Fig. 37; a conceptual diagram stating the attributes that are more closely associated with either citizen science or professional scientific projects.
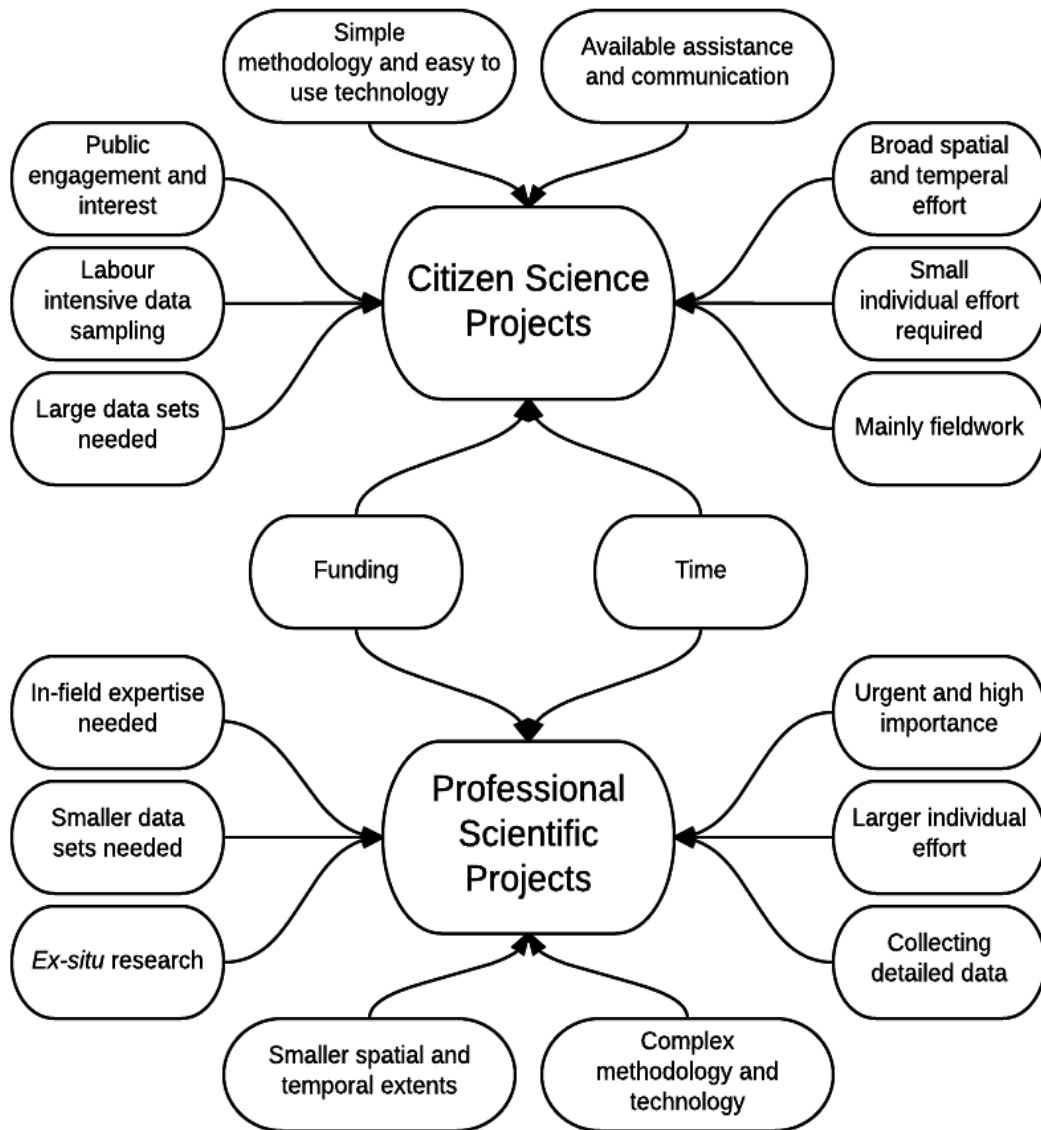
Fig. 37. A conceptual model showing the different attributes generally associated with citizen science projects and non-citizen science projects (attributes were adapted from Hill and Wilkinson, 2004; Gommerman and Monroe, 2012 and during the development of Section 3). Taken from Florisson (2015).

Section 8: Reef Vision

The lessons learned during this project have been actively applied to Reef Vision, an expanded and modified version of the current project funded by the Fisheries Research and Development Corporation (FRDC 2014/005) and led by Recfishwest. Reef Vision has the same goals as the current study, *i.e.* citizen science (recreational fisher) monitoring of the artificial reefs in Bunbury and Dunsborough using underwater cameras (Fig. 38). Only this time, the Sony drop cameras with the live feed have been replaced by the custom-designed BRUVs with a GoPro camera, as used in the later stages of this project. Among the 11 recreational fishers who are involved in the video monitoring component of Reef Vision, seven were involved during the first or second round of recruitment for the current project.

As of February 2016, 54 separate deployments of the BRUVs have occurred providing over 4,000 minutes of footage comprising over 500 Gigabytes. The success, so far, of the Reef Vision project is likely due to two major improvements. Firstly, while the Sony camera with the live feed provided a good picture for fishers out on the water, the quality of the recorded footage was poor (see examples above in Fig. 6). Thus, fishers were unable to watch the footage back and see clearly what they had 'captured' on film. We consider that this contributed to them becoming disengaged with the project. Secondly, the communication between the fishers and the scientists was been dramatically improved following the recommendations outlined above. Each fisher was interviewed by phone before signing up to the project and made aware of their commitments and they were also sent a letter outlining the responsibilities of the scientists and stating what their role was. Recruitment occurred once the BRUVs were built, negating the problem that occurred in the current study with the delay between recruitment and the delivery of the cameras. Two onsite training evenings were held to provide the fishers the chance to meet the project team and each other and receive their BRUVs and instructions.

A private Facebook group was established to allow participants to communicate with each other (and the scientists) and post photos and videos from their BRUV deployments (Fig. 39). The availability of high quality footage and some significant 'captures' such as Saw Sharks (*Pristiophorus cirratus*), Spinner Sharks (*Carcharhinus brevipinna*) and the Spotted

Wobbegong (*Orectolobus maculatus*) and large numbers of recreationally important species such as Mulloway (*Argyrosomus japonicas*), Samson Fish (*Seriola hippos*) also increase engagement. Furthermore, the use of email rather than phone calls enabled more regular contact to occur between the participants and the scientists and at times suitable for both parties. Regular emails containing highlights from footage collected during the previous month has helped maintain interest in the project and subtlety act as a reminder to deploy the BRUV and/or provide the footage to the scientists.

There is no question that the lessons learned from the pilot study (which was the first of its kind in Western Australia, and possibly the world) have greatly assisted the development of Reef Vision and helped it be a success. It was concluded the end of the pilot study that, while recreational fishers did not provide a cost-effective means of monitoring the fish faunas of artificial reefs in that project, they could do so in the future. The success of Reef Vision to date supports that latter part of that statement.

Fig. 38. The flyer attracting recreational fishers to participate in Reef Vision.
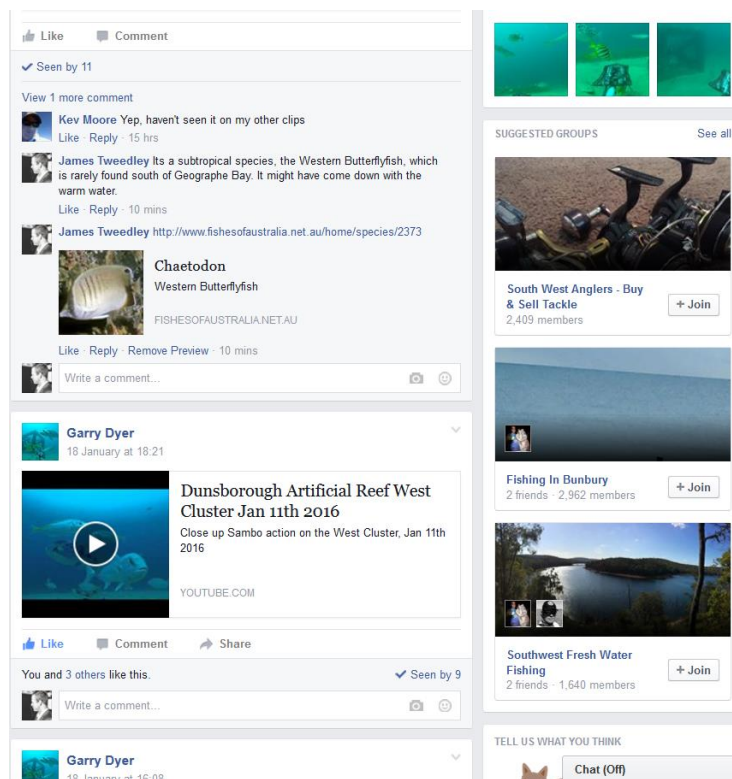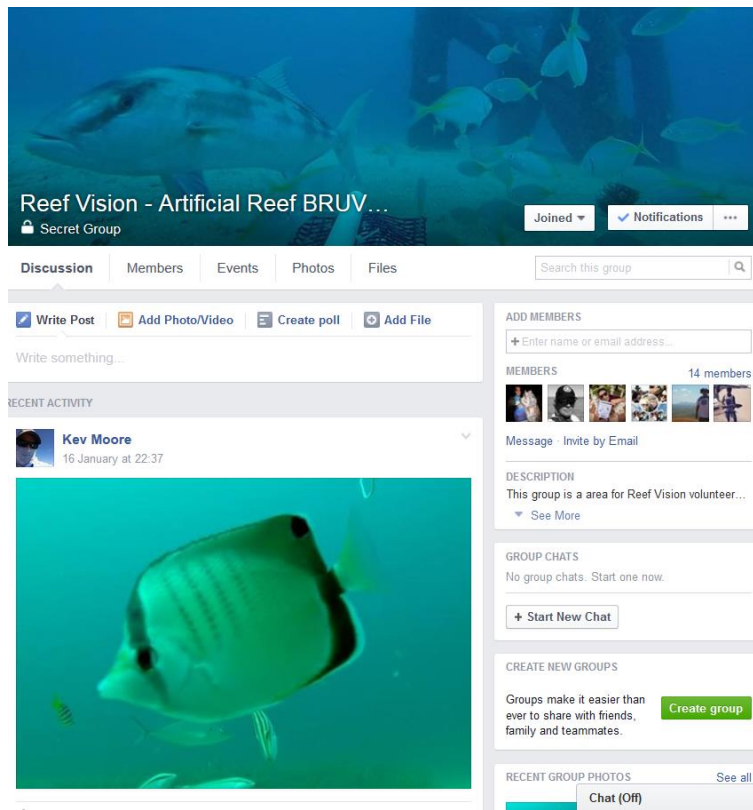
Fig. 39. Recent extracts from the Reef Vision Facebook page, which was set up to facilitate contact among the recreational fishers, and also between the fishers and the scientists, and thereby promote engagement in the project.

References

Anderson, M. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, **26:** 32-46.

Anderson, M., Gorley, R. and Clarke, K. (2008). PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER-E, Plymouth, UK.

Arena, P., Jordan, L. and Spieler, R. (2007). Fish assemblages on sunken vessels and natural reefs in southeast Florida, USA. *Hydrobiologia*, **580:** 157-171.

Armstrong, J., Bagley, P. and Priede, I. (1992). Photographic and acoustic tracking observations of the behaviour of the grenadier *Coryphaenoides (Nematonurus) armatus*, the eel *Synaphobranchus bathybius*, and other abyssal demersal fish in the North Atlantic Ocean. *Marine Biology*, **112:** 535-544.

Bailey-Brock, J. (1989). Fouling community development on an artificial reef in Hawaiian waters. *Bulletin of Marine Science*, **44:** 580-591.

Baine, M. (2001). Artificial reefs: a review of their design, application, management and performance. *Ocean and Coastal Management*, **44:** 241-259.

Baltais, S. (2013). Citizen Science, the benefits and challenges: How do you make citizen science both fun and useful? 4th Queensland Coastal Conference, Townsville October 2013.

Bassett, D. and Montgomery, J. (2011). Investigating nocturnal fish populations in situ using baited underwater video: with special reference to the olfactory capabilities. *Journal of Experimental Marine Biology and Ecology*, **409:** 194-199.

Bellchambers, L., K. Smith and D. Harris (2006). An assessment of the blue swimmer crab fishery in Geographe Bay. Fisheries Research Report No. 158. Perth, WA: Department of Fisheries Government of Western Australia.

Belmaker, J., N. Shashar and Y. Ziv (2005). Effects of small-scale isolation and predation on fish diversity on experimental reefs. *Marine Ecology Progress Series*, **289:** 273-283.

Bohnsack, J. A., D. E. Harper, D. B. McClellan and M. Hulsbeck (1994). Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, USA. *Bulletin of Marine Science*, **55:** 796-823.

Bombace, G., Fabi, G., Fiorentini, L. and Speranza, S. (1994). Analysis of the efficacy of artificial reefs located in five different areas of the Adriatic Sea. *Bulletin of Marine Science*, **55:** 559-580.

Bonney, R., Cooper, C., Dickinson, J., Kelling, S., Phillips, T., Rosenberg, K. and Shirk, J. (2009). Citizen science: a developing tool for expanding science knowledge and scientific literacy. *Bioscience*, **59:** 977-984.

Bonter, D. and Cooper, C. (2012). Data validation in citizen science: a case study from Project FeederWatch. *Frontiers in Ecology and the Environment*, **10:** 305-307.

Booth, D. and Fowler, A. (2013). Seasonal dynamics of fish assemblages on breakwaters and natural rocky reefs in a temperate estuary: consistent assemblage differences driven by sub-adults. *PLoS ONE,* **8:** e75790.

Bortone, S. (2007). Coupling fisheries with ecology through marine artificial reef deployments. *American Fisheries Society Symposium*, **49:** 587-594.

Bortone, S. A., F. P. Brandini, G. Fabi and S. Otake. (2011). Artificial reefs in fisheries management. Florida, USA: Taylor & Francis.

Branden, K. L., D. A. Pollard and H. A. Reimers (1994). A review of recent artificial reef developments in Australia. *Bulletin of Marine Science*, **55:** 982-994.

Bray, D. (2011). Footballer Sweep, *Neatypus obliquus*, in Fishes of Australia. Accessed 17 May 2015 from http://fishesofaustralia.net.au/home/species/2445.

Brock, R. (1994). Beyond Fisheries Enhancement: Artificial Reefs and Ecotourism. *Bulletin of Marine Science*, **55:** 1181-1188.

Burchmore, J., Pollard, D., Bell, J., Middleton, M., Pease, B. and Matthews, J. (1985). An ecological comparison of artificial and natural rocky reef fish communities in Botany Bay, New South Wales, Australia. *Bulletin of Marine Science*, **37:** 70-85.

Cappo, M. M. and I. I. Brown (1996) Evaluation of sampling methods for reef fish populations of commercial, recreational interest. CRC Reef Research Technical report, 6, 1-72.

Cappo, M., Harvey, E., Malcolm, H. and Speare, P. (2003). Potential of video techniques to monitor diversity, abundance and size of fish in studies of Marine Protected Areas. In: Beumer, J., Grant, A. and Smith, D. (eds.) Aquatic protected areas – What works best and how do we know? Proceedings of the World Congress on Aquatic Protected Areas. Australian Society for Fish Biology, North Beach, Western Australia, p. 455-464.

Cappo, M., Speare, P. and De'ath, G. (2004). Comparison of baited remote underwater video stations (BRUVs) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. *Journal of Experimental Marine Biology and Ecology*, **302:** 123-152.

Cappo, M., Harvey, E. and Shortis, M. (2007). Counting and measuring fish with baited video techniques- an overview. *Proceedings of the Australian Society for Fish Biology Workshop*, Hobart, August 2007, p. 101-114.

Carr, M. H. and M. A. Hixon (1997). Artificial reefs: the importance of comparisons with natural reefs. *Fisheries*, **22:** 28-33.

Charbonnel, E., Serre, C., Ruitton, S., Harmelin, J. and Jensen, A. (2002). Effects of increased habitat complexity on fish assemblages associated with large artificial reef units (French Mediterranean coast). *ICES Journal of Marine Science*, **59:** 208-213.

Chen, I., Hill, J., Ohlemuller, R., Roy, D. and Thomas, C. (2011). Rapid range shifts of species associated with high levels of climate warming. *Science*, **333:** 1024-1026.

Clark, S. and A. J. Edwards (1994). Use of artificial reef structures to rehabilitate reef flats degraded by coral mining in the Maldives. *Bulletin of Marine Science*, **55:** 724-744.

Clarke, K. and R. Green (1988). Statistical design and analysis for a "biological effects" study. *Marine Ecology Progress Series*, **46:** 213-226.

Clarke, K. (1993). Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology, 18: 117-143.

Clarke, K. R., J. R. Tweedley and F. J. Valesini (2014a) Simple shade plots aid better long-term choices of data pre-treatment in multivariate assemblage studies. *Journal of the Marine Biological Association of the United Kingdom*, **94:** 1-16.

Clarke, K. R., R. N. Gorley, P. J. Somerfield and R. M. Warwick (2014b). Change in Marine Communities: An Approach to Statistical Analysis and Interpretation 3rd Edition. Plymouth, UK: PRIMER-E Ltd.

Clarke, K. R. and R. N. Gorley (2015). PRIMER v7 User Manual/Tutorial. Plymouth, UK: PRIMER-E Ltd.

Claudet, J. and D. Pelletier (2004). Marine protected areas and artificial reefs: A review of the interactions between management and scientific studies. *Aquatic Living Resources*, **17:** 129-138.

Conrad, C. and Daoust, T. (2008). Community-based monitoring frameworks: Increasing the effectiveness of environmental stewardship. *Environmental management*, **41:** 356-358.

Conrad, C. C. and K. G. Hilchey (2011). A review of citizen science and community-based environmental monitoring: issues and opportunities. *Environmental Monitoring and Assessment*, **176:** 273-291.

Cornell University. (2012). Pioneering eBird citizen science program tops 100 million observations. Accessed 2 February 2015 from

http://www.newswise.com/articles/pioneering-ebird-citizen-science-program-tops-100-million-observations.

Crall, A., Newman, G., Jarnevich, C., Stohlgren, T., Waller, D. and Graham, J. (2010). Improving and integrating data on invasive species collected by citizen scientists. *Biological Invasions*, **12:** 3419-3428.

Cresson, P., Ruitton, S. and Harmelin-Vivien, M. (2014). Artificial reefs do increase secondary biomass production: mechanisms evidenced by stable isotopes. *Marine Ecology Progress Series*, **509:** 15-26.

De Vos, L., A. Götz, H. Winker and C. Attwood (2014). Optimal BRUVs (baited remote underwater video system) survey design for reef fish monitoring in the Stilbaai Marine Protected Area. *African Journal of Marine Science*, **36:** 1-10.

Delaney, D., Sperling, C., Adams, C. and Leung, B. (2008). Marine invasive species: validation of citizen science and implications for national monitoring networks. *Biological Invasions*, **10:** 117-128.

Department of Fisheries. 2012a. Policy on habitat enhancement structures in Western Australia. Fisheries Management Paper Number 256. Government of Western Australia.

Department of Fisheries. 2012b. Aquatic invaders identification guide. Department of Fisheries, Western Australia, 8 pp.

Department of Fisheries. (2012c). Go fishing for science! Department of Fisheries, Western Australia, 6 pp.

Department of Fisheries. (2013). South West artificial reefs project. Government of Western Australia. Accessed 10 February 2015 from http://www.fish.wa.gov.au/Documents/artificial_reefs/south_west_artificial_reefs_flyer.pdf.

Department of Fisheries. 2015. Artificial Reefs. Department of Fisheries, Western Australia. Accessed 2 January 2015, from http://www.fish.wa.gov.au/Fishing-and-Aquaculture/Recreational-Fishing/Pages/Artificial-Reefs.aspx.

Dickinson, J., Zuckerberg, B. and Bonter, D. (2010). Citizen science as an ecological research tool: challenges and benefits. *Annual review of ecology, evolution and systematics*, **41:** 149-172.

Diplock, J. (2010). Artificial Reefs - Design and Monitoring Standards Workshops. Fisheries Research and Development Corporation Project No. 2010/400. Summer Hill, NSW: Fisheries Research and Development Corporation. 52 pp.

Dorenbosch, M., Grol, M., Nagelkerken, I. and Velde, G. (2005). Distribution of coral reef fishes along a coral reef-seagrass gradient: edge effects and habitat segregation. *Marine Ecology Progress Series*, **299:** 277-288.

Dorman, S., Harvey, E. and Newman, S. (2012). Bait effects in sampling coral reef fish assemblages with stereo-BRUVs. *PLoS ONE* **7:** e41538.

Duffy, C. and Paul, L. (2003). *Dasyatis brevicaudata*. The IUCN Red List of Threatened Species. Version 2014.3. Accessed 16 May, 2015 from www.iucnredlist.org/details/41796/0.

Ebner, B., Clear, R., Godschalx, S. and Beitzel, M. (2009). In-stream behaviour of threatened fishes and their food organisms based on remote video monitoring. *Aquatic Ecology*, **43:** 569-576.

Edgar, G. (1997). Australian Marine Life, the plants and animals of temperate waters. Reed Books, Kew, Australia, 544 pp.

Ellis, D. and DeMartini, E. (1995). Evaluation of a video camera technique for indexing abundances of juvenile pink snapper, *Pristipomoides filamentosus*, and other Hawaiian insular shelf fishes. *Fishery Bulletin*, **93:** 67-77.

European Commission. (2014). Green paper on citizen science. Accessed 3 May 2015 from http://ec.europa.eu/digital-agenda/en/news/green-paper-citizen-science-europe-towards-society-empowered-ctisens-and-enhanced-research-0.

Fabi, G. and L. Fiorentini (1996). Molluscan aquaculture on reefs: European Artificial Reef Research Network Conference (EARRN). Anoca, IT.

Fairclough, D., Brown, J., Carlish, B., Crisafulli, B. and Keay, I. (2014). Breathing life into fisheries stock assessments with citizen science. *Nature Scientific Reports*, **4:** 7249.

Felix-Hackradt, F., Hackradt, C., Trevino-Oton, J., Perez-Ruzafa, A. and Garcia-Charton, J. (2013). Temporal patterns of settlement, recruitment and post-settlement losses in a rocky reef fish assemblage in the South-Western Mediterranean Sea. *Marine Biology*, **160:** 2337-2352.

Fernandez-Gimenez, M., Ballard, H. and Sturtevant. (2008). Adaptive management and social learning in collaborative and community based monitoring: a study of five community based forestry organisations in the western USA. *Ecology and Society*, **13:** 4.

Fischer, P., Weber, A., Heine, G. and Weber, H. (2007). Habitat structure and fish: assessing the role of habitat complexity for fish using a small, semiportable, 3-D underwater observatory. *Limnology and Oceanography: Methods,* **5:** 250-262.

Florisson, J. H. (2015). Can recreational fishers provide and effective means of monitioring artificial reefs. Honours Thesis, Murdoch University. Western Australia.

Folpp, H., M. Lowry, M. Gregson and I. M. Suthers (2011. Colonization and community development of fish assemblages associated with estuarine artificial reefs. *Brazilian Journal of Oceanography,* **59:** 55-67.

Folpp, H., Lowry, M., Gregson, M. and Suthers, I. (2013). Fish assemblages on estuarine artificial reefs: natural rocky-reef mimics or discrete assemblages? *PLoS One*, **8:** e63505.

Fowler, A. and Booth, D. (2012). How well do sunken vessels approximate fish assemblages on coral reefs? Conservation implications of vessel-reef deployments. *Marine Biology*, **159:** 2787-2796.

Froese, R. and D. Pauly. (2015). "FishBase." Assess 5 September, 2015 from http://www.fishbase.org/.

Gersbach, G.H., Pattiaratchi, C.B., Ivey, G.N., Cresswell, G.R. (1999). Upwelling on the south-west coast of Australia - source of the Capes Current? *Continental Shelf Research* **19:** 363-400.

Gladstone, W., S. Lindfield, M. Coleman and B. Kelaher (2012). Optimisation of baited remote underwater video sampling designs for estuarine fish assemblages. *Journal of Experimental Marine Biology and Ecology*, **429:** 28-35.

Gledhill, C., Lyczkowski-Schultz, J., Rademacher, K., Kargard, E., Crist, G. and Grace, M. (1996). Evaluation of video and acoustic index methods for assessing reef-fish populations. *ICES Journal of Marine Science*, **53:** 483-485.

Goetze, J., Langlois, T., Egli, D. and Harvey, E. (2011). Evidence of artisanal fishing impacts and depth refuge in assemblages of Fijian reef fish. *Coral Reefs*, **30:** 1-11.

Goldstein, E., Lawton, C., Sheehy, E. and Butler, F. (2014). Locating species range frontiers: a cost and efficiency comparison of citizen science and hair-tube survey methods for use in tracking an invasive squirrel. *Wildlife Research*, **41:** 61.

Gollan, J., L. L. de Bruyn, N. Reid and L. Wilkie (2012). Can volunteers collect data that are comparable to professional scientists? A study of variables used in monitoring the outcomes of ecosystem rehabilitation. *Environmental Management*, **50:** 969-978.

Gollan. J. (2013). Citizen science can produce reliable data. School of the Environment, University of Technology, Sydney. Accessed 28 January 2015 from http://world.edu/citizen-science-can-produce-reliable-data/.

Gomelyuk, V. (2012). Assessment of Darwin Harbour fish communities using baited remote underwater video station (BRUVS). Report to Department of Land Resource Management, Darwin, 53 pp.

Gommerman, L. and Monroe, M. (2012). Lessons learned from evaluations of citizen science programs. Florida Cooperative Extension Service, Institute of Food and Agricultural Services, University of Florida, USA, 5 pp.

Granneman, J. and Steele, M. (2015). Effects of reef attributes on fish assemblage similarity between artificial and natural reefs. *ICES Journal of Marine Science*. 72, doi: 10.1093/icesjms/fsv094.

Great Barrier Reef Marine Park Authority (GBRMPA). (2013). Citizen science in the Great Barrier Reef: a scoping study. Great Barrier Reef Marine Park Authority. Accessed 1 February 2015 from http://www.researchgate.net/publication/269225105_Citizen_Science_in_the_Great_Barrier_Reef_A_scoping_study.

Grove, R. S., M. Nakamura, H. Kakimoto and C. J. Sonu (1994). Aquatic habitat technology innovation in Japan. *Bulletin of Marine Science*, **55:** 276-294.

Hackradt, C. W., Felix-Hackradt, F. C., and Garcia-Charton, J. A. (2011). Influence of habitat structure on fish assemblage of an artificial reef in southern Brazil. *Marine Environmental Research*, **72:** 235–247.

Haejoo. (2011). QLD Moreton Bay artificial reef program. Accessed 5 February 2015, from www.haejoo.com.au/qld-moreton-bay-artificial-reef-program/.

Haejoo. (2013). WA South West artificial reef project. Accessed 5 February from www.haejoo.com.au/the-south-west-artificial-reef-project-western-australia/.

HaeJoo. (2015). "WA South West Artificial Reef Project." Accessed 13 March 2015, from http://haejoo.com/the-south-west-artificial-reef-project-western-australia/.

Hammar, L., Ehnberg, J., Eggertsen, L., Andersson, S. and Molander, S. (2012). Stereo-video methodology for quantitative analysis of fish-turbine interactions. 1st Asian Wave and Tidal Conference Series, Jeju, Korea. 8pp.

Harasti, D., H. Malcolm, C. Gallen, M. A. Coleman, A. Jordan and N. A. Knott (2015). Appropriate set times to represent patterns of rocky reef fishes using baited video. *Journal of Experimental Marine Biology and Ecology*, **463:** 173-180.

Harding, S., C. Lowery and S. Oakley (2000). Comparison between complex and simple reef survey techniques using volunteers: is the effort justified: International Coral Reef Society: 9th International Symposium Conference Proceedings. Bali, Indonesia.

Harman, N., Harvey, E. and Kendrick, G. (2003). Differences in fish assemblages from different reef habitats at Hamelin Bay, south-western Australia. *Marine and Freshwater Research*, **54:** 177-184.

Harris, L. (1988). Edge effects and conservation of biotic diversity. *Conservation Biology*, **2:** 330-332.

Harvey, E., D. McLean, S. Frusher, M. Haywood, S. Newman and A. Williams (2013). The use of BRUVs as a tool for assessing marine fisheries and ecosystems: a review of the hurdles and potential. University of Western Australia Report. Perth, WA: Fisheries Research and Development Corporation.

Heagney, E., Lynch, T., Babcock, R. and Suthers, I. (2007). Pelagic fish assemblages assessed using mid-water baited video: standardising fish counts using bait plume size. *Marine Ecology Progress Series*, **350:** 255-266.

Henriques, S., Pais, M., Costa, M. and Cabral, H. (2013). Seasonal variability of rocky reef fish assemblages: detecting functional and structural changes due to fishing effects. *Journal of Sea Research*, **79:** 50-59.

Hill, J. and Wilkinson, C. (2004). Methods for Ecological Monitoring of Coral Reefs. Australian Institute of Marine Science, Townsville, Australia, 117 pp.

Hochachka, W., Fink, D., Hutchinson, R., Sheldon, D., Wong, W. and Kelling, S. (2012). Data-intensive science applied to broad-scale citizen science. *Trends in Ecology and Evolution*, **27:** 130-137.

Holbrook, S., Kingsford, M., Schmitt, R. and Stephens Jr, J. (1994). Spatial and temporal patterns in assemblages of temperate reef fish. *American Zoology,* **34:** 463-475.

Holmes, T. H., S. K. Wilson, M. J. Travers, T. J. Langlois, R. D. Evans, G. I. Moore, R. A. Douglas, G. Shedrawi, E. S. Harvey and K. Hickey (2013). A comparison of visual-and stereo-video based fish community assessment methods in tropical and temperate marine waters of Western Australia. *Limnology and Oceanography Methods*, **11:** 337-350.

Hunter, W. and Sayer, M. (2009). The comparative effects of habitat complexity on faunal assemblages of northern temperate artificial and natural reefs. *ICES Journal of Marine Science*, **66:** 691-698.

Hutchins, B. and R. Swainston (1986). Sea fishes of southern Australia: complete field guide for anglers and divers. Perth, WA: Swainston Publishing.

Hutchins, B. and Swainston, R. (2012). Sea Fishes of Southern Australia. Swainston Publishing, Perth, Australia, 180 pp.

Jenkins, G., French, D., Potter, I., de Lestang, S., Hall, N., Partridge, G., Hesp, A. and Sarre, A. (2006). Restocking the Blackwood River Estuary with Black Bream *Acanthopagrus butcheri*. Fisheries Research and Development Corporation Report 2000/180, Challenger TAFE, Fremantle, 84 pp.

Jensen, A. (2002). Artificial reefs of Europe: perspective and future. *ICES Journal of Marine Science*, **59:** 3-13.

Jiguet, F. (2009). Method learning caused a first-time observer effect in a newly started breeding bird survey. *Bird Study*, **56:** 253-258.

Jordan, R., Gray, S., Howe, D., Brooks, W. and Ehrenfeld, J. (2011). Knowledge gain and behavioural change in citizen science programs. *Conservational Biology*, **25:** 1148-1154.

Kellison, T. and Sedberry, G. (1998). The effects of artificial reef vertical profile and hole diameter on fishes off South Carolina. *Bulletin of Marine Science*, **62:** 763-780.

Koeck, B., Tessier, A., Brind'Amour, A., Pastor, J., Bijaoui, B., Dalias, N., Astruch, P., Saragoni, G. and Lenfant, P. (2014). Functional differences between fish communities on artificial and natural reefs: a case study along the French Catalan coast. *Aquatic Biology*, **20:** 219-234.

Lambert, J. 2014. Citizen Science for Flora and Fauna Conservation: Ensuring Success. IEST5004 Research Internship Final Report. University of New South Wales, Sydney, Australia, 31 pp.

Langlois, T., E. Harvey, B. Fitzpatrick, J. Meeuwig, G. Shedrawi and D. Watson (2010). Cost-efficient sampling of fish assemblages: comparison of baited video stations and diver video transects. *Aquatic Biology*, **9:** 155-168.

Lawler, J. and O'Connor, R. (2004). How well does consistently monitored breeding bird survey routes represent the environments of the conterminous United States? *Condor*, **106:** 801-814.

Lek, E., D. Fairclough, M. Platell, K. Clarke, J. Tweedley and I. Potter (2011) To what extent are the dietary compositions of three abundant, co-occurring labrid species different and related to latitude, habitat, body size and season? *Journal of Fish Biology*, **78:** 1913-1943.

Letessier, T., Juhel, J., Vigliola, L. and Meeuwig, J. (2015). Low-cost action cameras in stereo generates accurate underwater measurements of fish. *Journal of Experimental Marine Biology and Ecology*, **466:** 120-126.

Link, W. and Sauer, J. (1999). Controlling for varying effort in count surveys-an analysis of Christmas Bird Count data. *Agriculture, Biology, Environment and Statistics*, **4:** 116-125.

Lopez-Perez, R., Calderon-Aguilera, L., Zepeta-Vilchis, R., Lopez-Perez Maldonado, I. and Lopez Ortiz, A. (2013). Species composition, habitat configuration and seasonal changes of coral reef fish assemblages in western Mexico. *Journal of Applied Ichthyology*, **29:** 437-448.

Lowry, M., H. Folpp, M. Gregson and R. McKenzie (2010). Assessment of artificial reefs in Lake Macquarie NSW. Port Stephens Fisheries Institute Final Report Series No. 125. Port Stephens, NSW: Department of Primary Industries.

Lowry, M., H. Folpp, M. Gregson and I. Suthers (2012). Comparison of baited remote underwater video (BRUV) and underwater visual census (UVC) for assessment of artificial reefs in estuaries. *Journal of Experimental Marine Biology and Ecology*, **416:** 243-253.

Malcolm, H. A., W. Gladstone, S. Lindfield, J. Wraith and T. P. Lynch (2007). Spatial and temporal variation in reef fish assemblages of marine parks in New South Wales, Australia-baited video observations. *Marine Ecology Progress Series*, **350:** 277-290.

Mallet, D. and D. Pelletier (2014). Underwater video techniques for observing coastal marine biodiversity: A review of sixty years of publications (1952–2012). *Fisheries Research*, **154:** 44-62.

May, J. and Maxwell, J. (1986). Field guide to trawl fish from temperate waters of Australia. Hobart. CSIRO Division of Fisheries Research, 492 pp.

McKenzie, L., Yoshida, R., Smith, N. and Langlois, L. (2014). Indigenous community capacity building to assess dugong and sea turtle seagrass habitats for sea country management. A report to the Threatened Species Division, Queensland Department of Environment & Heritage Protection (EHP). Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) report 14/58. James Cook University, Cairns, Australia, 53 pp.

McLean, D., Harvey, E., Fairclough, D. and Newman, S. (2010). Large decline in the abundance of a targeted tropical lethrinid in areas open and closed to fishing. *Marine Ecology Progress Series*, **418:** 189-199.

McMahon, K., Young, E., Montgomery, S., Cosgrove, J., Wilshaw, J. and Walker, D. (1997). Status of a shallow seagrass system, Geographe Bay, south-western Australia. *Journal of the Royal Society of Western Australia*, **80:** 255-262.

Milne, R., Rosolen, S., Whitelaw, G. and Bennett, L. (2006). Multi-party monitoring in Ontario: Challenges and emerging solutions. *Environments*, **34:** 11-23.

Nakamae, A. (1991). Artificial reef projects in Japan. Coastal Fisheries Development Division Report No. 1991/11. Toyko, JP: Fisheries Agency of Japan.

Nakamura, M. (1985). Evolution of artificial fishing reef concepts in Japan. *Bulletin of Marine Science*, **37:** 271-278.

Newman, G., Zimmerman, D., Crall, A., Laituri, M., Graham, J. and Stapel, L. (2010). User-friendly web mapping: lessons from a citizen science website. *International Journal of Geographical Information Science*, **24:** 1851-1869.

Ng, K., Phillips, M., Borges, P., Thomas, T., August, P., Calado, H. and Veloso-Gomes, F. (2014). Maintaining a way of life for Sao Miguel Island (the Azores archipelago, Portugal): An assessment of coastal processes and protection. *Science of the Total Environment,* **481:** 142-156.

Niemuth, N., Dahl, A., Estey, M. and Loesch, C. (2007). Representation of land cover along breeding bird survey routes in the northern plains. *Journal of Wildlife Management*, **71:** 2258-2265.

Oberhauser, K. and Prysby, M. (2008). Citizen science: creating a research army for conservation. *American Entomologist*, **54:** 97-99.

Ody, D. and Harmelin, J. (1994). Effects of the design and location of artificial reefs on Mediterranean fish assemblages. *Cybium: international journal of ichthyology*, **18:** 57-70.

Oldham, C., Lavery, P., McMahon, K., Pattiaratchi, C. and Chiffings, T. (2010). Seagrass wrack dynamics in Geographe Bay, Western Australia. Synopsis prepared for Department of Transport and Shire of Busselton by Edith Cowan University, University of Western Australia and DHI, 23 pp.

Open Scientist. (2011). Finalizing a Definition of "Citizen Science" and "Citizen Scientists". Accessed 29 January 2015 from http://www.openscientist.org/2011/09/finalizing-definition-of-citizen.html.

Pais, A., E. Azzurro and P. Guidetti (2007). Spatial variability of fish fauna in sheltered and exposed shallow rocky reefs from a recently established Mediterranean Marine Protected Area. *Italian Journal of Zoology*, **74:** 277-287.

Pattengill-Semmens, C. V. and B. X. Semmens (2003). Conservation and management applications of the reef volunteer fish monitoring program. *Environmental Monitioirng and Assesment*, **81:** 43-50.

Pearce, A. and Pattiaratchi, C. (1999). The Capes Current: A summer countercurrent flowing past Cape Leeuwin and Cape Naturaliste, Western Australia. *Continental Shelf Research,* **19:** 401-420. 48

Pecl, G., Barry, Y., Brown, R., Frusher, S., Gartner, E., Pender, A., Robinson, L., Walsh, P. and Stuart-Smith, J. (2014). REDMAP: Ecological monitoring and community engagement through citizen science. *The Tasmanian Naturalist*, **136:** 158-164.

Pelletier, D., K. Leleu, D. Mallet, G. Mou-Tham, G. Herve, M. Boureau and N. Guilpart (2012). Remote high-definition rotating video enables fast spatial survey of marine underwater macrofauna and habitats. *PLoS One* **7**: 30-36.

Pepperell Research and Consulting. (2010). Monitoring and research on landed fish at game fishing tournaments in NSW 2008/2009. Report prepared for the NSW Recreational Fishing Trust Expenditure Committee.

Pickering, H., D. Whitmarsh and A. Jensen (1999). Artificial Reefs as a Tool to Aid Rehabilitation of Coastal Ecosystems: Investigating the Potential. *Marine Pollution Bulletin*, **37**: 505-514.

Platell, M., Sarre, G. and Potter, I. (1997). The diets of two co-occuring marine teleosts, *Parequula melbournensis* and *Pseudocaranx wrighti*, and their relationships to body size and mouth morphology, and the season and location of capture. *Environmental Biology of Fishes*, **49:** 361-376.

Platell, M. and Potter, I. (2001). Partitioning of food resources amongst 18 abundant benthic carnivorous fish species in marine waters on the lower west coast of Australia. *Journal of Experimental Marine Biology and Ecology*, **261:** 31-54.

Platell, M., Hesp, S., Cossington, S., Lek, E., Moore, S. and Potter, I. (2010). Influence of selected factors on the dietary compositions of three targeted and co-occuring temperate species of reef fishes: implications for food partitioning. *Journal of Fish Biology*, **76:** 1255-1276.

Pollard, D. (1989). Artificial habitats for fisheries enhancement in the Australia region. *Marine Fisheries Review*, **51:** 11–26.

Pradel, R. (1996). Utilization of capture-mark-recapture for the study of recruitment and population growth rate. *Biometrics*, **52:** 703-709.

Priede, I., Bagley, P., Smith, A., Creasey, S. and Merrett, N. (1994). Scavenging deep demersal fishes of the Porcupine Seabight, north-east Atlantic: observations by baited camera, trap and trawl. *Journal of the Marine Biological Association of the UK*, **74:** 481-498.

Priede, I. and Merrett, N. (1996). Estimation of abundance of abyssal demersal fishes; a comparison of data from trawls and baited cameras. *Journal of Fish Biology*, **49:** 207-216.

Ramos-Esplá, A. A., J. E. Guillén, J. T. Bayle and P. Sánchez-Jérez (2000). Artificial anti-trawling reefs off Alicante, South-Eastern Iberian Peninsula: Evolution of reef block and set designs. Artificial reefs in European seas. A. C Jenson, K. J. Collins and A. P. Lockwood (Eds.). Southampton, UK: Springer, 195-218.

Ries, L. and Sisk, T. (2004). A predictive model of edge effects. *Journal of Ecology*, **85:** 2917-2926.

Rilov, G. and Benayahu, Y. (2000). Fish assemblage on natural versus vertical artificial reefs: the rehabilitation perspective. *Marine Biology*, **136:** 931-942.

Root, T. and Alpert, P. (1994). Volunteers and the NBS. *Science*, **263:** 1205.

Rotman, D., Preece, J., Hammock, J., Procita, K., Hansen, D., Parr, C., Lewis, D. and Jacobs, D. (2012). Dynamic changes in motivation in collaborative citizen science projects. Proceedings of the ACM 2012 conference on Computer Supported Cooperative Work 2012. ACM, 217-226.

Rowland, A. (2009). The biology of Samson Fish *Seriola hippos* with emphasis on the sport fishery in Western Australia. PhD thesis. Murdoch University.

Sale, P. (1980). Assemblage of fish on patch reefs – predictable or unpredictable? *Environmental Biology of Fishes*, **5:** 243-249.

Samoilys, M. A. and G. Carlos (2000). Determining methods of underwater visual census for estimating the abundance of coral reef fishes. *Environmental Biology of Fishes*, **57:** 289-304.

Santos, M. and Monteiro, C. (1997). The Olhao artificial reef system (south Portugal): fish assemblages and fishing yield. *Fisheries Research*, **30:** 33-41.

Sbrocchi, C. (2013). Marine citizen science in Australia. School of the Environment, University of Technology, Sydney. Accessed 21 January 2015 from http://www.redmap.org.au/article/marine-citizen-science-in-australia/.

Schobernd, Z., Bacheler, N. and Conn, P. (2013). Examining the utility of alternative video monitoring metrics for indexing reef fish abundance. *Canadian Journal of Fisheries and Aquatic Sciences*, 71: 464-471.

Seaman, W. and K. Tsukamoto (2008). Coastal artificial habitats for fishery and environmental management and scientific advancement. Fisheries for global welfare and environment. Tokyo, JP: World Fisheries Congress.

Seaman, W. J. and L. M. Spraque. (1991). Artificial habitats for marine and freshwater fisheries. London, UK: Academic Press.

Sherman, R., Gilliam, D. and Spieler, R. (2002). Artificial reef design: void space, complexity, and attractants. *ICES Journal of Marine Science,* **59:** 196-200.

Shmeller, D., Henry, P., Julliard, R., Gruber, B. and Clobert, J. (2009). Advantages of volunteer-based biodiversity modelling in Europe. *Journal of Conservational Biology*, **23:** 307-316.

Shulman, M. (1984). Resource limitation and recruitment patterns in coral reef fish assemblage. *Journal of Experimental Marine Biology and Ecology*, **74:** 85-109.

Silvertown, J. (2009). A new dawn for citizen science. *Trends in Ecology and Evolution*, **24:** 467-471.

Smale, D. A., T. J. Langlois, G. A. Kendrick, J. J. Meeuwig and E. S. Harvey (2011). From fronds to fish: the use of indicators for ecological monitoring in marine benthic ecosystems, with case studies from temperate Western Australia. *Reviews in Fish Biology and Fisheries,* **21:** 311-337.

Smallwood, C., Hesp, S. and Beckley, L. (2013). Biology, stock status and management summaries for selected fish species in south-western Australia. Fisheries Research Report No. 242. Department of Fisheries, Western Australia, 180 pp.

Smith, M. P. L. (1989) Improving multispecies rocky reef fish censuses by counting different groups of species using different procedures. *Environmental Biology of Fishes*, **26:** 29-37.

Smith-Vaniz, W. F. and H. L. Jelks (2006) Australian trevallies of the genus *Pseudocaranx* (Teleostei: Carangidae), with description of a new species from Western Australia. *Memoirs of Museum Victoria*, **63:** 97-106.

Somerton, D. and C. Glendhill (2005). Report of the National Marine Fisheries Service Workshop on underwater video analysis. NOAA Technical Memorandum. Seattle, WA: Department of Commerce.

Spanier, E. and G. Almog-Shtayer (1992). Shelter preferences in the Mediterranean slipper lobster: effects of physical properties. *Journal of Experimental Marine Biology and Ecology*, **164:** 103-116.

Stobart, B., J. A. García-Charton, C. Espejo, E. Rochel, R. Goñi, O. Reñones, A. Herrero, R. Crec'hriou, S. Polti and C. Marcos (2007). A baited underwater video technique to assess shallow-water Mediterranean fish assemblages: Methodological evaluation. *Journal of Experimental Marine Biology and Ecology*, **345:** 158-174.

Sullivan, B., Aycrigg, J., Barry, J., Bonney, R., Bruns, N., Cooper, C., Damoulas, T., Dhondt, A., Dietterich, T. and Farnsworth, A. (2014). The eBird enterprise: An integrated approach to development and application of citizen science. *Biological Conservation*, **169:** 31-40.

Svane, I. and Petersen, J. (2001). On the problems of epibiosis, fouling and artificial reefs, a review. *Marine Ecology,* **22:** 169-188.

Tessier, E., Chabanet, P., Pothin, K. and Lasserre, G. (2005). Visual censuses of tropical fish aggregations on artificial reefs: slate versus video recording techniques. *Journal of Experimental Marine Biology and Ecology*, **315:** 17-30.

Thiel, M., Penna-Diaz, M. A., Luna-Jorquera, G., Salas, S., Sellanes, J. and Stotz, W. (2014). Citizen scientists and marine research: volunteer participants, their contributions, and projection for the future. *Oceanography and Marine Biology: An Annual Review*, **52:** 257-314.

Thompson, A. and B. Mapstone (1998). Observer effects and training in underwater visual surveys of reef fishes. *Oceanographic Literature Review*, **2:** 291.

Thresher, R. and J. Gunn (1986). Comparative analysis of visual census techniques for highly mobile, reef-associated piscivores (Carangidae). *Environmental Biology of Fishes*, **17:** 93-116.

Tissot, B. N. (2008). Video analysis, experimental design, and database management of submersible-based habitat studies. Marine Habitat Mapping Technology for Alaska. Fairbanks. J. R. Reynolds and H. J. Greene (Eds.). Fairbanks, AK: Alaska Sea Grant College Program.

Trayler, K., Tweedley, J. and Jenkins, G. (2015). Prawn Watch: Engaging fishers in science, restocking and sustainable management of an iconic species that has declined in abundance. 2015 Australian Citizen Science Conference. Canberra, Australia.

Tulloch, A., Possingham, H., Joseph, L., Szabo, J. and Martin, T. (2013). Realising the full potential of citizen science monitoring programs. *Biological Conservation* **165:** 128-138.

Tweedley. J., Loneragan, N., Jenkins, G., Trayler, K. and Chaplin, J. (2014). Assessing the decline in western school prawn (Metapenaeus dalli) abundance in a temperate Australian estuary: evaluating the effectiveness of prawn releases and the impacts of environmental change. 2014 Annual Symposium of the Fisheries Society of the British Isles. Hull, United Kingdom.

Tweedley, J. Warwick, R. and Potter, I. (2015a). Can biotic indicators distinguish between natural and anthropogenic environmental stress in estuaries? *Journal of Sea Research*, **102:** 10-21.

Tweedley, J.R., Poh, B., Broadley, A.D., Crisp, J.A. & Loneragan, N.R. (2015b). Understanding the ecology of the Western School Prawn to maximize restocking success. 5th International Symposium on Stock Enhancement and Sea Ranching. Sydney, Australia.

Tyre, A. J., Tenhumberg, B., Field, S. A., Niejalke, D., Parris, K., and Possingham, H. P. (2003). Improving precision and reducing bias in biological surveys: estimating false-negative error rates. *Ecological Applications*, **13:** 1790-1801.

Valesini, F., J. Tweedley, K. Clarke and I. Potter (2014). The importance of regional, system-wide and local spatial scales in structuring temperate estuarine fish communities. *Estuaries and Coasts*, **37:** 525-547.

Viswanathan, M., A. Ammerman, E. Eng, G. Garlehner, K. N. Lohr, D. Griffith, S. Rhodes, C. Samuel-Hodge, S. Maty and L. Lux (2004). Community-based participatory research: Assessing the evidence: Summary. Technology Assessment No. 99. Rockville, MD: Department of Health and Human Resources.

Walker, D., R. Lukatelich and A. McComb (1987). Impacts of proposed developments on the benthic marine community of Geographe Bay. Environmental Protection Authority Technical Series, 11, 20.

Walsh, W. J. (1985). Reef fish community dynamics on small artificial reefs: the influence of isolation, habitat structure, and biogeography. *Bulletin of Marine Science,* **36:** 357-376.

Wartenberg, R. and Booth, A. (2014). Video transects are the most appropriate underwater visual census method for surveying high-latitude coral reef fishes in the southwest Indian Ocean. Marine Biodiversity. 1-14.

Watson, D., Harvey, E., Anderson, M. and Kendrick, G. (2005). A comparison of temperate reef fish assemblages recorded by three underwater stereo-video techniques. *Journal of Marine Biology*, **148:** 415-425.

Watson, D. (2006). Use of underwater stereo-video to measure fish assemblage structure, spatial distribution of fishes and change in assemblages with protection from fishing. PhD Thesis. University of Western Australia.

Watson, D., Harvey, E., Kendrick, G., Nardi, K. and Anderson, M. (2007). Protection from fishing alters the species composition of fish assemblages in a temperate-tropical transition zone. *Marine Biology*, **152:** 1197-1206.

Watson, D., Harvey, E., Fitzpatrick, B., Langlois, T. and Shedrawi, G. (2010). Assessing reef fish assemblage structure: how do different stereo-video techniques compare? *Marine Biology*, **157:** 1237–1250.

Wernberg, T., Russel, B., Thomsen, M., Gurgel, C., Bradshaw, C., Poloczanska, E. and Connell, S. (2011). Seaweed communities in retreat from ocean warming. *Current Biology*, **21:** 1828-1832.

Westera, M., P. Lavery and G. Hyndes (2003). Differences in recreationally targeted fishes between protected and fished areas of a coral reef marine park. *Journal of Experimental Marine Biology and Ecology*, **294:** 145-168.

White, W. (2006). *Trygonoptera personata*. The IUCN Red List of Threatened Species. Version 2014.3. Accessed 16 May 2015 from www.iucnredlist.org/details/60084/0.

Whitelaw, G., Vaughan, H., Craig, B. and Atkinson, D. (2003). Establishing the Canadian Community Monitoring Network. *Environmental Monitoring and Assessment*, **88:** 409-418.

Whittaker, R. H. (1952) A study of summer foliage insect communities in the Great Smoky Mountains. *Ecological Monographs,* **22:** 1-44.

Wiber, M., F. Berkes, A. Charles and J. Kearney (2004). Participatory research supporting community-based fishery management. *Marine Policy*, **28:** 459-468.

Wiersma, Y. (2010). Birding 2.0: citizen science and effective monitoring in the Web 2.0 world. *Avian Conservation and Ecology*, **5:** 13.

Wiggins, A. and Crowston, K. (2011). From conservation to crowdsourcing: A typology of citizen science. System Sciences (HICSS) 2011. 44th Hawaii International Conference, 10 pp.

Williams, I., W. Walsh, B. Tissot and L. Hallacher (2006). Impact of observers' experience level on counts of fishes in underwater visual surveys. *Marine Ecology Progress Series*, **310:** 185-191.

Willis, T. J. and R. C. Babcock (2000) A baited underwater video system for the determination of relative density of carnivorous reef fish. *Marine and Freshwater Research*, **51:** 755-763.

Wilson, C. and Godinho, L. (2013). The benefits of engaging volunteers in urban bat research. *Victorian Naturalist*, **130:** 182-187.

Yanagi, T. and Nakajima, M. (1991). Change of oceanic condition by the man-made structure for upwelling. *Marine Pollution Bulletin*, **23:** 131-135.

Yoklavich, M. M. and V. O'Connell (2008). Twenty years of research on demersal communities using the Delta submersible in the Northeast Pacific. Marine habitat mapping technology for Alaska. J. R. Reynolds and H. G. Greene (Eds.). Fairbanks, AK, 143-155.

Can recreational fishers provide an effective means of monitoring artificial reefs?

Artificial reefs have been constructed and deployed globally to enhance the productivity of aquatic habitats. In April 2013, two artificial reefs were deployed in Geographe Bay, Western Australia for the purpose of enhancing recreational fishing opportunities. These reefs are designed to create varied complex spaces and habitats, as well as to create shallow water upwelling to drive nutrients up into the water column. The deployment of artificial reefs in Australia has recently become the subject of specific focus of policy makers and regulators. Monitoring costs to meet legislative requirements can be prohibitive, however, a potential method to reduce these costs is to utilise volunteers from the general public to collect data (*i.e.* citizen science). Thus, the overall objective of this project was to determine whether recreational fishers could potentially provide an effective means for monitoring artificial reefs.

A small number of recreational fishers were provided with underwater video cameras and asked to record footage of artificial reefs and nearby natural reefs. Unfortunately, only limited amounts of data were received due to the lack of participation, unseasonal weather and the short timeframe of the project. However, enough videos were received to undertake a preliminary analysis of the differences in the characteristics of the fish faunas of the two types of reef. The results demonstrated that artificial reefs had much higher levels of mean and maximum abundance, number of species and ecological group affinities. However, multivariate statistical analyses did not detect any differences between the fish faunal compositions between artificial and natural reefs. This was due to the dominance of the labrid *Coris auricularis* and the large amount of variability between replicates.

Given the limited data provided by the above citizen science program, a literature review on other similar projects to evaluate the effectiveness of the citizen science components of the pilot project was completed and provided a set of key recommendations. These included

enhancing the methods of contacting and recruiting volunteers, providing simplified and consistent instructions and consistent communication and engagement with volunteers.

Finally, Baited Remote Underwater Video (BRUV) systems, constructed from readily available materials, were deployed randomly around the Busselton artificial reef to test the applicability of this method for future use as a citizen science artificial reef monitoring tool. The video footage was analysed to determine whether there was a difference in fish assemblages between artificial reef modules and the surrounding area, *i.e.* videos observing areas in which artificial reef modules were, and were not, observed in the camera's field of view. The results demonstrated that mean number of species and the number of benthic and epibenthic species were greater on footage recorded when the camera faced the modules. There was also a difference in the faunal composition. The footage observing artificial reef modules also exhibited 52.63% more recreational target species than surrounding areas. It was concluded that the BRUV technology employed here could be used, by citizen scientists, to monitor the fish faunas of artificial reefs. However, as this study has also demonstrated that there were significant differences in the characteristics of the fish faunas recorded depending on the direction the camera was facing, consideration is needed to design an unbiased and robust quantitative monitoring regime.

It is concluded that recreational fishers did not provide an effective means for monitoring artificial reefs during this project. This result, however, is a consequence of a lack of data stemming from an absence of volunteer engagement in a limited pilot project with a short time frame and unseasonal weather. This does not exclude the potential for using citizen scientists to monitor artificial reefs, following some changes in the methodology, technology and management of citizen science protocols, and thus it is possible to utilise recreational fishers as an effective means for monitoring artificial reefs. This project was subjected to restrictive and limiting factors but more importantly, discovered ways to overcome these issues by provided key recommendations on technology, methodologies and community engagement that should be followed to increase the effectiveness of using recreational fishers to provide sound scientific information in the future.

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

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

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