School of Molecular and Life Sciences

A Comparison of Stereo-video Techniques for Assessing Fish Communities On and Off Spatiality Limited Habitats

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Declaration

To the best of my knowledge and belief this thesis contains no material previously published by any other person except where due acknowledgment has been made. This thesis contains no material which has been accepted for the award of any other degree or diploma in any university.

The research presented and reported in this thesis was conducted in compliance with the National Health and Medical Research Council Australian code for the care and use of animals for scientific purposes 8th edition (2013). The proposed research study received animal ethics approval from the Curtin University Animal Ethics Committee, Approval Number #AEC_2014_09; AEC_2014_42; ARE_2018_20.

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Date: 9/11/2020

Abstract

The use of an appropriate sampling technique is an important consideration when sampling fish communities on spatially limited habitats such as patch reefs or artificial structures. The intrinsic biases and limitations of different techniques can influence the composition of fish sampled, potentially resulting in different biological interpretations. This can cause implications for new or revised management strategies that may rely on such data. Ongoing advancements in technology have allowed the development of new sampling approaches, some of which may serve as appropriate tools for fine-scale applications. It is important, however, to evaluate the performance of new sampling approaches to see how they compare and complement existing techniques, particularly across habitats where sampling biases are known to vary.

The oil and gas industry could benefit from the development of a standardised tool for assessing fish communities on their spatially limited infrastructure. With decommissioning activities expected to increase, information on the ecological value of oil and gas infrastructures (platforms, wells and subsea pipelines) as marine habitat will assist decisions on decommissioning outcomes. These outcomes could range from complete removal to leave *in-situ* to supplement as artificial reefs. There are a limited number of environmental studies that assess fish communities on subsea pipelines, despite their extensive footprint on the seafloor. The studies that do exist have often used different sampling approaches making it challenging to draw general conclusions on the ecological role of subsea pipelines as fish habitat.

In this thesis I investigated the sampling abilities of four different stereo-video techniques, two of which are commonly used in fisheries and conservation management (diver operated stereo-video (stereo-DOV) and baited remote underwater stereo-video systems (stereo-BRUVs)) and two which are newly developed (slow towed stereo-video (stereo-STV) and stereo-video remotely operated vehicle (stereo-ROV)), for surveying fish communities on and off spatially limited habitats.

To compare the four techniques fish data was collected on a narrow limestone ridge within Geographe Bay, Western Australia (chapter two). Stereo-BRUVs out performed transect sampling techniques (stereo- DOV, STV, ROV) by sampling more species and fish with greater statistical power. However, combining the data from stereo-BRUVs with any one transect technique increased the observed number of species by ~10%, providing a more holistic interpretation.

Using background and technical developments outlined in chapter two, fish assemblage data sampled by stereo-BRUVs and stereo-ROV techniques were compared on exposed subsea pipelines and surrounding natural habitats (reef and soft sediment), around Thevenard Island, Western Australia (chapter three). Larger disparities were observed in soft sediment habitats between techniques, while more similar patterns were encountered on the pipeline and in reef habitat. These differences likely reflected the use of bait in stereo-BRUVs surveys where fish were potentially attracted from a large and unknown area. It is also possible that fish avoided the stereo-ROV, an effect likely heightened in open soft sediment habitat. Given the narrow footprint of pipelines, I conclude that stereo-ROV is a more useful tool for surveying fish on these structures. I also warn potential users that caution is needed when sampling open soft sediment habitats with a stereo-ROV due to potential fish behavioural biases.

Using stereo-ROVs, the ecological value of subsea pipelines was investigated and compared to the surrounding natural habitat (chapter four). Pipelines contained a distinct composition of fish, characterised by a high abundance and biomass of fish from higher trophic levels. Ease of access to prey and positioning of pipelines over favourable foraging habitat likely drive these associations. While these structures do not mimic reef habitats, pipelines do have significant ecological and socioeconomic values.

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Author contribution: KS, MM, CJ, and EH conceived the study. KS, MM and EH collected the data. KS analysed the video imagery with assistance. KS, BS, TE and EH conducted the statistical analyses. KS wrote the manuscript. All co-authors (MM, TE, CJ, BS, SN, EH) were involved in writing and editing the manuscript.

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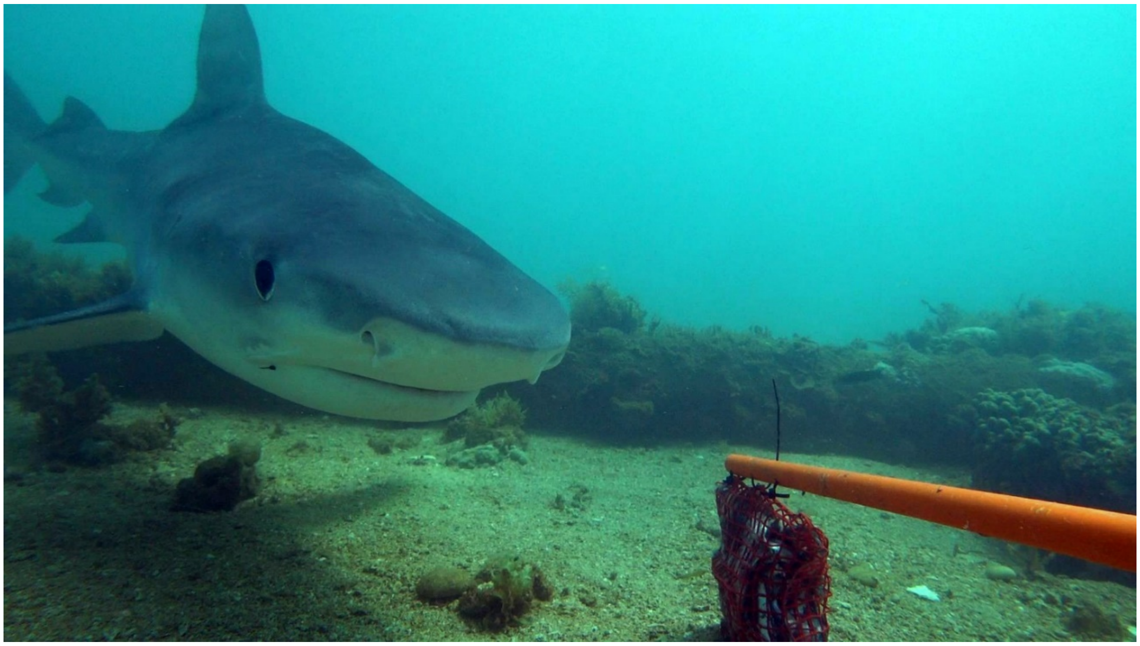
Chapter 1: General Introduction



1.1. Background and rationale

Ecological studies that aim to describe the diversity, biomass and abundance of fish rely on appropriate sampling techniques capable of collecting accurate and reliable data (Andrew and Mapstone, 1987). When collecting ecological data, it is important to consider the potential sampling biases of each possible technique as this data is often used to quantify natural variation and identify the effects of anthropogenic activities (e.g. fishing, habitat loss, climate change) on fish populations. Population and diversity estimates are known to vary depending on the technique employed and may result in different biological insights and interpretations (Harvey et al., 2007; Logan et al., 2017; Watson et al., 2010). Furthermore, the sampling biases of a technique may also change over a spatial scale. For example, Lindfield et al. (2014) found that fish behavioural biases were heightened outside of protected areas in response to bubbles produced by SCUBA, but were mitigated using silent closed-circuit rebreather, resulting in different conclusions between fished and protected areas. This can cause implications for both fishery and conservation management where data is used to create or revise management strategies (McCormick and Choat, 1987). Consideration is also needed in regards to the repeatability and cost-effectiveness of a sampling approach to ensure it can be used for long-term monitoring if necessary (Langlois et al., 2010). As fisheries and conservation managers are often concerned with losses in diversity and the biomass of fish in an ecosystem, non-extractive techniques such as video-based surveys represent a more suitable approach than extractive techniques that rely on hook and line, trawls or traps to describe populations (Murphy and Jenkins, 2010; Pelletier et al., 2011). The outcomes of extractive techniques can also be bias towards the type of species caught and the class size (Wells et al., 2008; Patterson et al., 2012; Langlois et al., 2015). To evaluate the performance and suitability of sampling techniques, it is essential that comparisons are made between techniques across different habitats. This is particularly crucial in the validation of newly developed techniques that may overcome some of the biases and limitations of more traditional approaches (Fig. 1.4).

An important concept in fish ecology is understanding fish-habitat associations. Research on habitat usage can provide insights into the mechanisms that drive fish distributions and the community structure of an assemblage (Anderson et al., 2009; Moore et al., 2010; Valavanis et al., 2004; Ward et al., 1999). However, sampling habitats that are spatially limited requires the careful choice of technique to ensure that appropriate spatial scales of measurement are used, whereby fish associations are representative of the targeted habitat (Chittaro, 2004; Hale et al., 2019; Hewitt et al., 1998; Sale, 1998). The spatial scale used to assess fish-habitat associations has shown to be an important predictor of fish occupancy (e.g. fine vs. broad scale sampling) (Galaiduk et al., 2017). Patch reefs, limestone ridges, artificial reefs and oil and gas infrastructure are all considered spatially limited habitats and require appropriate sampling approaches that have the ability to capture representative fish associations in order to determine the ecological role of such habitats (Fig. 1.4).

Investigations into the ecological role of oil and gas infrastructure as habitat has increased in recent years as decommissioning activities are expected to increase within the coming decades (Macreadie et al., 2011; Parente et al., 2006). This has led to an increase in studies that investigate fish communities on and off oil and gas platforms to understand possible impacts to the marine environment (e.g. Ajemian et al., 2015a, 2015b; Claisse et al., 2014; Friedlander et al., 2014; Love and York, 2005; Streich et al., 2017; Torquato et al., 2017). There is increasing evidence that oil and gas platforms have the potential to serve as artificial reefs (Claisse et al., 2014; Fowler and Booth, 2012; Friedlander et al., 2014; Gallaway et al., 2009; Love et al., 2006; McLean et al., 2020a). Knowledge about their role in the marine environment will be of value to stakeholders and a critical component in facilitating appropriate decommissioning options (Fig. 1.4). This could range from regulators opting for complete removal (the default for most nations), or the consideration of proposed alternatives such as leaving infrastructure in place, toppling onto the seafloor, partial removal, or relocating to form artificial reefs (‘rig-to-reef’) (Fig. 1.1) (Bull and Love, 2019; Fowler et al., 2014; Schroeder and Love, 2004). Subsea pipelines are an integral component of oil and gas operations and form extensive networks which can span across multiple habitats ranging from coral reefs to soft sediment habitats interspersed with sessile invertebrates. Despite their prevalence on the seafloor there has been limited environmental studies which assess their ecological role as fish habitat (although see Bond et al., 2018a, 2018b; 2018c; Love and York, 2005; McLean et al., 2017, 2020b). Of the studies that do exist, they have used a range of sampling techniques (e.g. submersible, industry ROV, and stereo-BRUVs) making it challenging to draw general conclusions about the broad ecological patterns of fish assemblages associated with these structures. Therefore, it is necessary that technique comparisons are undertaken on spatially limited habitats like subsea pipelines and other natural structures with equally limited footprints to determine best practice (Fig. 1.4).

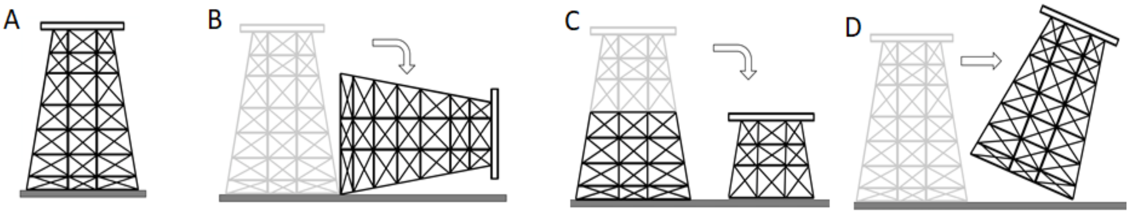


Fig. 1.1 Proposed in-situ alternative decommissioning strategies (A) leaving infrastructure in place, (B) toppling onto the seafloor, (C) partial removal, or (D) relocating to form artificial reefs (‘rig-to-reef’). Figure based on Macreadie et al. (2011).

Diver based surveys such as underwater visual census (UVC) and diver operated stereo-video systems (Stereo-DOVs) are two common non-extractive approaches for assessing reef fish communities. UVC has advantages over other video-based techniques for assessing cryptic species (Colton and Swearer, 2010; Lowry et al., 2012), but is dependent on skilled scientific divers who can promptly identify and measure fish *in-situ* making it undesirable for less experienced researchers or in organisations with high turnover of staff (Harvey et al., 2001c; Holmes et al., 2013) (Fig. 1.4). The development of a stereo-video system used in stereo-DOV surveys overcomes some of the limitations and biases of UVC by enabling improved accuracy and precision for determining fish length estimates while delineating the sampling area (Goetze et al., 2019; Harvey et al., 2001a, 2001b; Harvey and Shortis, 1998)(Fig. 1.4). This is necessary for assessing spatially limited habitats where the unit area can be constrained. Permanent recording of surveys also enables fish to be identified and counted post data collection with validation (Goetze et al., 2015; Holmes et al., 2013). However, the use of divers in both sampling approaches means fish surveys are often restricted to shallow water, limiting the ability to investigate habitats which typically extend into deep-water (e.g. oil and gas structures) (Cappo et al., 2003) (Fig. 1.4). The presence of a diver is also known to introduce behavioural biases where 'diver-shy' fish may be under-represented or over-represented for species that are attracted to diver activity (Cappo et al., 2003; Lindfield et al., 2014). Furthermore, due to occupational health and safety constraints there is increasing pressure to remove divers from the water. Consequently, researchers are increasingly adopting remote sampling techniques (Fig. 1.4).

The most common of these remote sampling methods is baited remote underwater stereo-video systems (stereo-BRUVs) (Harvey et al., 2018). This sampling method has become a standard approach for recording the relative abundance and diversity of fish across a range of habitats that encompass and exceed diver limits (Langlois et al., 2020). The use of bait reduces zero counts of fish and permits observations of large predatory and targeted species which may be considered 'diver-shy' (Cappo et al., 2006; Goetze et al., 2015; Gray et al., 2016) (Fig. 1.4). Like all techniques there are limitations in using stereo-BRUVs, for example the dispersal range of the bait plume is difficult to quantify, limiting fish counts to be expressed as a relative abundance (Cappo et al., 2001; Harvey et al., 2007). There is also the possibility of attracting fish from nearby habitats, limiting fish surveys aimed at fine-scale assessments (Fig. 1.4). However, despite these limitations, stereo-BRUVs are a useful tool for investigating fish-habitat associations and status at a broad scale (e.g. Langlois et al., 2012; MacNeil et al., 2020).

Technological advancements in remote transect sampling techniques have provided new platforms for researchers to survey fish communities at greater depths, whilst permitting unit per area measurements. Towed video is an example that has been used for deep water surveys (Logan et al., 2017; Sherlock et al., 2016), however controlled navigation has been identified as a limitation of this technique (Warnock et al., 2016) (Fig. 1.4). In recent years, remotely operated vehicles (ROVs) have emerged as an alternative approach for remote transect sampling (McLean et al., 2020c). Advancements in ROV design have allowed reductions in cost and size, making operations more accessible in the field (Consoli et al., 2016; Sward et al., 2019) (Fig. 1.4). Controlled maneuverability of the system also enables fine scale sampling where habitats are spatially limited or have vertical relief (Smith and Lindholm, 2016). This however, requires communications through a tether that has the potential to become entangled, a limiting factor in ROV operations (Ajemian et al., 2015b). Fish behavioural biases have also been documented while using ROVs (e.g. Laidig et al., 2013; Lorance and Trenkel, 2006; McLean et al., 2017; Stoner et al., 2008; Trenkel et al., 2004b), but have been mostly associated with larger work-class size vehicles (Fig. 1.4). Using a smaller ROV with a mounted stereo-video system may overcome some of the existing biases of ROV operations, whilst permitting accurate length measurements of fish within a defined sampling area (Schramm et al., 2020a). The mobile nature of ROVs also means transects or stationary point counts can be recorded, either of which may be a suitable approach for sampling fish-habitat relationships on spatially restricted structures (Fig. 1.4).

1.2. Study area

The research presented in this thesis was collected in two main locations along the coast of Western Australia (WA) (Fig. 1.2). Data for chapter two was collected in a temperate setting at Geographe Bay in the southwest of WA, along a five-kilometer limestone ridge. Data for chapters three and four were collected around Thevenard Island, north west Australia, on a network of exposed subsea pipeline, and surrounding tropical reef and soft sediment habitats. The study area of each location is further described in each chapter.

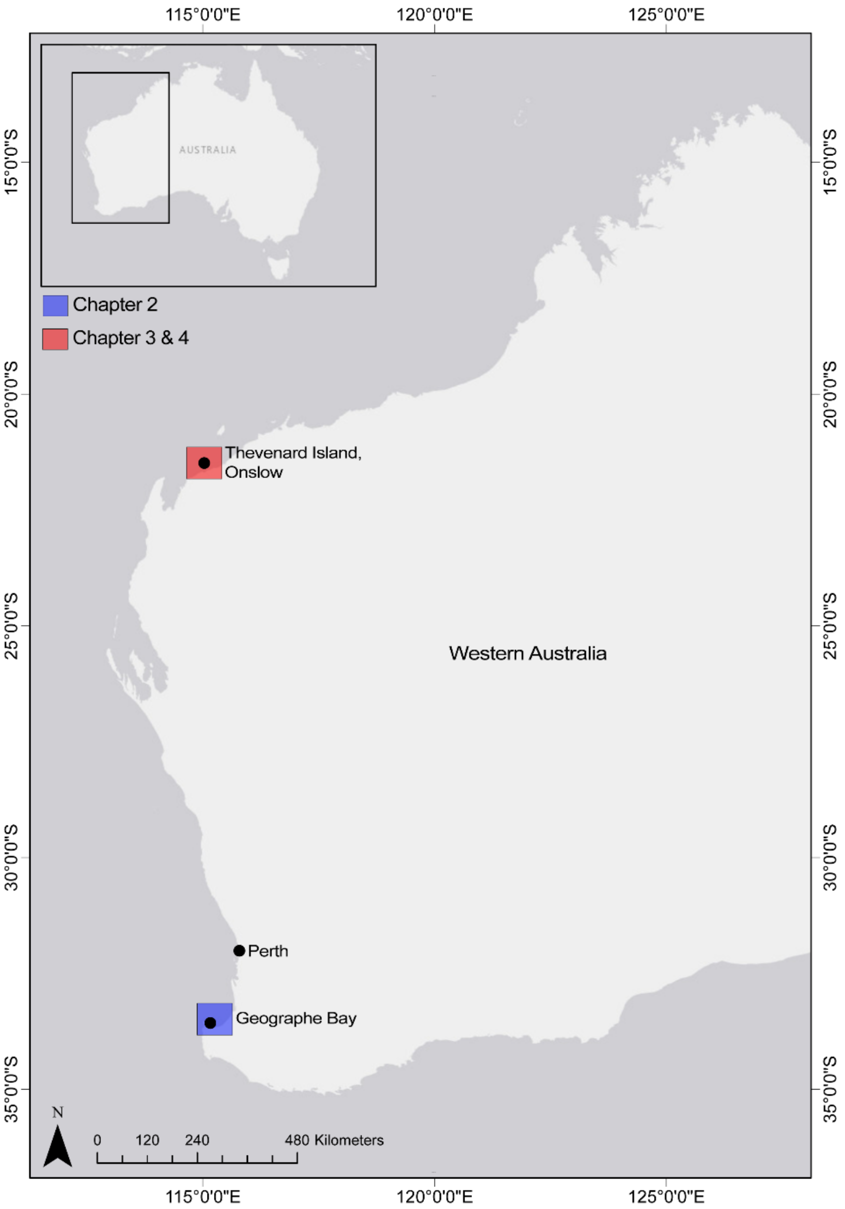


Fig. 1.2 Map of study areas, Western Australia.

1.3. Aims and objectives

The purpose of this thesis is to investigate the sampling abilities of four different stereo-video techniques, two of which are commonly employed for surveying marine fishes (stereo-DOVs and stereo-BRUVs; Fig. 1.3A and B), and two of which are being proposed (stereo-STV and stereo-ROV; Fig. 1.3C and D) for assessing fish communities on spatially limited structures (Fig. 1.4). I aim to address three questions in this thesis about technique choice and the ecological role of subsea pipelines:

1) How do common and novel marine fish sampling techniques compare in terms of observed fish assemblage and sampling effort?

Based on the findings from question one I developed an additional question comparing two techniques.

2) How do stereo-BRUVs and stereo-ROVs techniques compare in sampling fishes on and off subsea pipelines?

My third question used the most suitable technique, stereo-ROVs, to look at the fish assemblages on and off a shallow water oil and gas pipeline.

3) How do fish communities on subsea pipelines compare to those in the surrounding natural habitats, and what does this mean in terms of their ecological role in the marine environment?

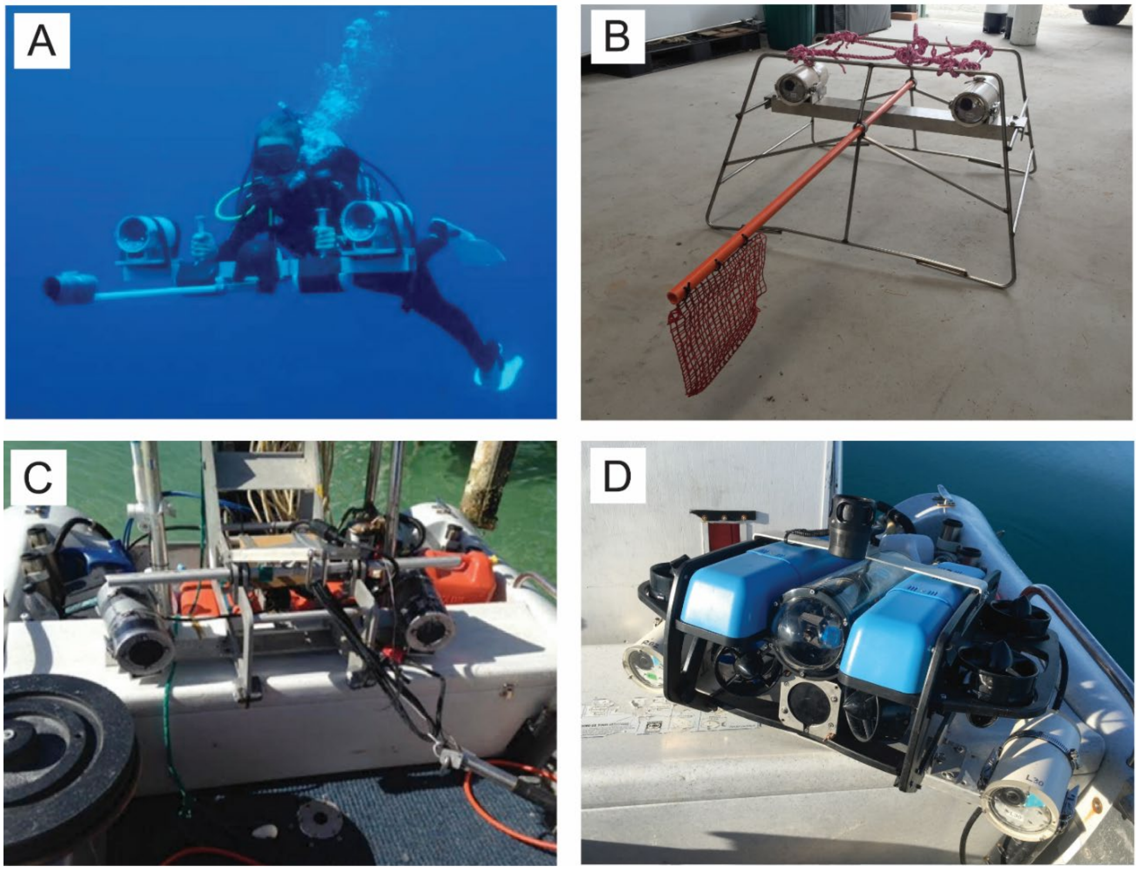


Fig. 1.3 Representative pictures of (A) diver-operated stereo-video (stereo-DOV), (B) baited remote underwater stereo-video system (stereo-BRUVs), (C) slow towed stereo-video (stereo-STV), and (D) remotely operated vehicle stereo-video (stereo-ROV). Photo credit: Laura Fullwood

1.4. A comparison of stereo-BRUV, diver operated and remote stereo-video transects for assessing reef fish assemblages (chapter 2)

Chapter two investigated the sampling abilities of four stereo-video techniques using a single narrow limestone ridge as a case study (Fig. 1.4). Across the five-kilometer ridge, 36 belt transects (25 m x 5 m, 125 m^2) were completed for each transect based technique (stereo-DOV, stereo-STV, and stereo-ROV), while 18 deployments were carried out for stereo-BRUVs. Comparisons were made based on the observed diversity and abundance of fish, statistical power, and cost effectiveness of each sampling technique. The abundance of focal species that represented targeted and non-targeted species of the region were also investigated between techniques.

1.5. A comparison of stereo-BRUVs and stereo-ROV techniques for sampling shallow water fish communities on and off pipelines (chapter 3)

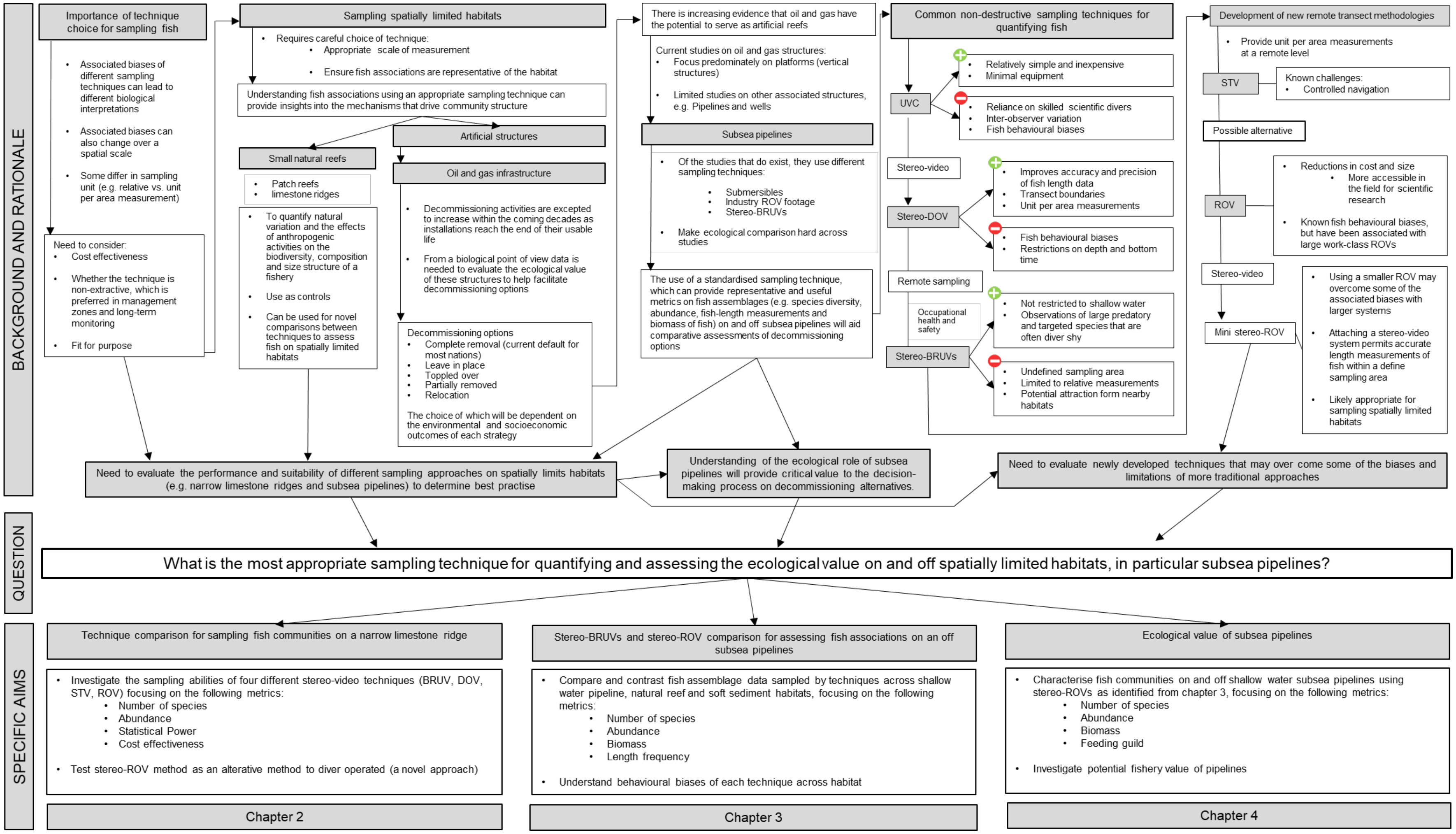
Applying the outcomes of chapter two, I investigated stereo-BRUVs and stereo-ROV techniques as a means of sampling shallow water subsea pipelines, and surrounding natural habitats (reef and soft sediment) (Fig. 1.4). Stereo-BRUVs surveys along the pipeline assessed seven sites with four deployments per site, while reef and soft sediment habitats involved 30 and 28 replicate deployments respectively. Stereo-ROV surveys involved 14 sites for each habitat type, where ~6 transects (50 m x 5 m, 250 m^2) were completed within each site. Similar to chapter two, I focused on observed diversity and abundance of fish, in addition to length frequency and biomass, all of which are considered relevant metrics for communicating with stakeholders and facilitating decisions on decommissioning outcomes. The abundance of three fish species that are commonly targeted by fishers in the Pilbara, *Plectropomus* spp. (Coral trout) *Lethrinus laticaudis* (Grass emperor)*,* and *Choerodon schoenleinii* (Blackspot tuskfish), were also compared between techniques.

1.6. Fish associations with shallow water subsea pipelines compared to surrounding reef and soft sediment habitats (chapter 4)

Chapter three identified that stereo-ROV was the most suitable technique for sampling pipeline habitats. Therefore, in chapter four I applied the stereo-ROV technique to investigate the ecological value of pipelines, with comparisons made to the surrounding reef and soft sediment habitat to provide ecological context (Fig. 1.4). Eleven segments of pipeline ranging from 0.3 - 1.7 km were included in the survey, involving 88 transects (50 m x 5 m, 250 m^2). For reef habitat 150 transects were sampled, while 145 transects were used for soft sediment. Comparisons were made in the context of observed diversity, abundance and biomass of fishes and focal species, in addition to the feeding guild composition across habitat. The fishery value of targeted fish was also explored.

1.7. Thesis structure

This thesis has been structured into five chapters, general introduction, three data chapters, and general discussion. Data chapter one has been published in the Journal of Experimental Marine Biology and Ecology (Schramm et al., 2020a), data chapter two has been published in the journal of Marine Environmental Research (Schramm et al., 2020b), and data chapter three has been accepted for publication in Scientific Reports. As stand-alone chapters, there is some repetition throughout this thesis.



Importance of technique choice for sampling fish

- Associated biases of different sampling techniques can lead to different biological interpretations
- Associated biases can also change over a spatial scale
- Some differ in sampling unit (e.g. relative vs. unit per area measurement)

Need to consider:
- Cost effectiveness
- Whether the technique is non-extractive, which is preferred in management zones and long-term monitoring
- Fit for purpose

Sampling spatially limited habitats

- Requires careful choice of technique:
    - Appropriate scale of measurement
    - Ensure fish associations are representative of the habitat

Understanding fish associations using an appropriate sampling technique can provide insights into the mechanisms that drive community structure

Small natural reefs

- Patch reefs
- limestone ridges

- To quantify natural variation and the effects of anthropogenic activities on the biodiversity, composition and size structure of a fishery
- Use as controls
- Can be used for novel comparisons between techniques to assess fish on spatially limited habitats

Artificial structures

Oil and gas infrastructure

- Decommissioning activities are excepted to increase within the coming decades as installations reach the end of their usable life
- From a biological point of view data is needed to evaluate the ecological value of these structures to help facilitate decommissioning options

Decommissioning options
- Complete removal (current default for most nations)
- Leave in place
- Toppled over
- Partially removed
- Relocation

The choice of which will be dependent on the environmental and socioeconomic outcomes of each strategy

There is increasing evidence that oil and gas have the potential to serve as artificial reefs

Current studies on oil and gas structures:
- Focus predominately on platforms (vertical structures)
- Limited studies on other associated structures, e.g. Pipelines and wells

Subsea pipelines

- Of the studies that do exist, they use different sampling techniques:
    - Submersibles
    - Industry ROV footage
    - Stereo-BRUVs
- Make ecological comparison hard across studies

The use of a standardised sampling technique, which can provide representative and useful metrics on fish assemblages (e.g. species diversity, abundance, fish-length measurements and biomass of fish) on and off subsea pipelines will aid comparative assessments of decommissioning options

Common non-destructive sampling techniques for quantifying fish

UVC
- ➕ Relatively simple and inexpensive
- ➕ Minimal equipment
- ➖ Reliance on skilled scientific divers
- ➖ Inter-observer variation
- ➖ Fish behavioural biases

Stereo-video

Stereo-DOV
- ➕ Improves accuracy and precision of fish length data
- ➕ Transect boundaries
- ➕ Unit per area measurements
- ➖ Fish behavioural biases
- ➖ Restrictions on depth and bottom time

Remote sampling

Occupational health and safety

Stereo-BRUVs
- ➕ Not restricted to shallow water
- ➕ Observations of large predatory and targeted species that are often diver shy
- ➖ Undefined sampling area
- ➖ Limited to relative measurements
- ➖ Potential attraction form nearby habitats

Development of new remote transect methodologies

- Provide unit per area measurements at a remote level

STV

Known challenges:
- Controlled navigation

Possible alternative

ROV
- Reductions in cost and size
    - More accessible in the field for scientific research
- Known fish behavioural biases, but have been associated with large work-class ROVs

Stereo-video

Mini stereo-ROV
- Using a smaller ROV may overcome some of the associated biases with larger systems
- Attaching a stereo-video system permits accurate length measurements of fish within a define sampling area
- Likely appropriate for sampling spatially limited habitats

Need to evaluate the performance and suitability of different sampling approaches on spatially limits habitats (e.g. narrow limestone ridges and subsea pipelines) to determine best practise

Understanding of the ecological role of subsea pipelines will provide critical value to the decision-making process on decommissioning alternatives.

Need to evaluate newly developed techniques that may over come some of the biases and limitations of more traditional approaches

What is the most appropriate sampling technique for quantifying and assessing the ecological value on and off spatially limited habitats, in particular subsea pipelines?

Technique comparison for sampling fish communities on a narrow limestone ridge

- Investigate the sampling abilities of four different stereo-video techniques (BRUV, DOV, STV, ROV) focusing on the following metrics:
    - Number of species
    - Abundance
    - Statistical Power
    - Cost effectiveness
- Test stereo-ROV method as an alterative method to diver operated (a novel approach)

Chapter 2

Stereo-BRUVs and stereo-ROV comparison for assessing fish associations on an off subsea pipelines

- Compare and contrast fish assemblage data sampled by techniques across shallow water pipeline, natural reef and soft sediment habitats, focusing on the following metrics:
    - Number of species
    - Abundance
    - Biomass
    - Length frequency
- Understand behavioural biases of each technique across habitat

Chapter 3

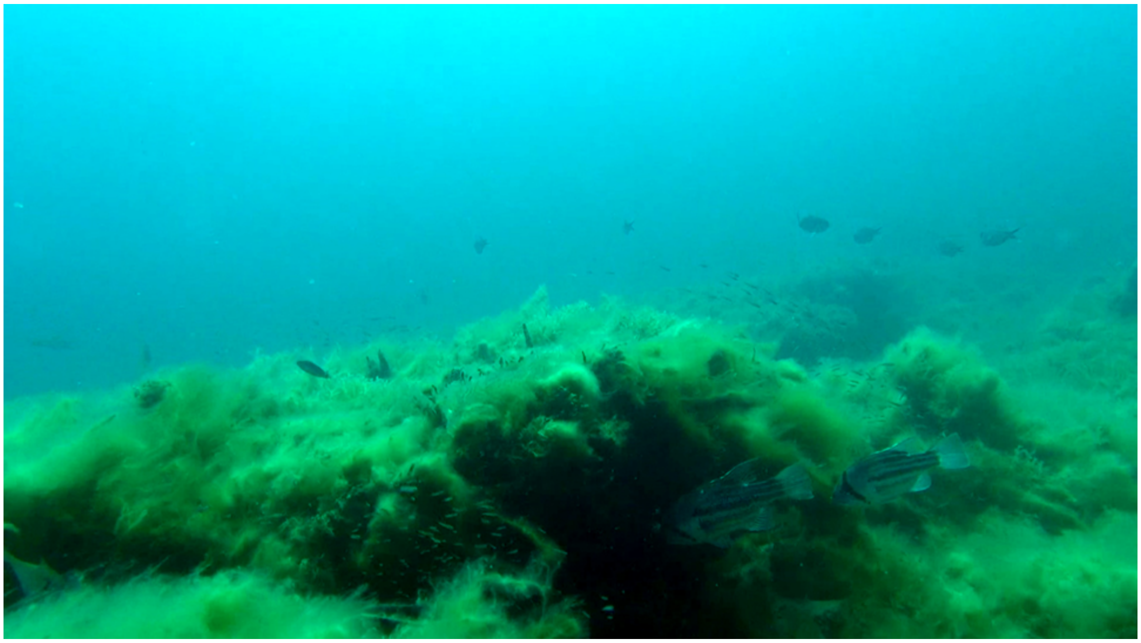
Ecological value of subsea pipelines

- Characterise fish communities on and off shallow water subsea pipelines using stereo-ROVs as identified from chapter 3, focusing on the following metrics:
    - Number of species
    - Abundance
    - Biomass
    - Feeding guild
- Investigate potential fishery value of pipelines

Chapter 4

Fig. 1.4 Thesis flow diagram outlining the background and rationale, main research question, and specific aims for each data chapter.

Chapter 2: A comparison of stereo-BRUV, diver operated and remote stereo-video transects for assessing reef fish assemblages



Preface: This chapter has been published in the Journal of Experimental Biology and Ecology (https://doi.org/10.1016/j.jembe.2019.151273) and has been formatted to the journal's guidelines.

2.1. Abstract

Monitoring programs that aim to measure the diversity, abundance and biomass of fishes depend on accurate and reliable biological data to guide the development of robust management plans. It is important to implement an appropriate sampling technique that provides a comprehensive assessment, while reducing associated biases in the data collection process. We compared the sampling abilities of four different underwater stereo-video techniques; baited remote underwater stereo-video systems (BRUV), diver operated stereo-video (DOV), and two relatively new methodologies; slow towed stereo-video (STV) and remotely operated vehicle (ROV) fitted with a stereo-video. Comparisons were conducted in a temperate reef system along a single limestone ridge within Geographe Bay, Western Australia. More fish species and individuals were sampled by BRUVs, while transect based methods (DOV, STV, and ROV) were comparable. Combining BRUVs with a transect method resulted in a ~10% increase in observed diversity. BRUVs had a higher statistical power to detect change in comparison to the transect methods. This was due to the higher variability observed in transect methods that may sample greater habitat heterogeneity within a sample unit compared to BRUVs. Although differences did exist, data collected with the ROV was generally comparable to the other transect methods. We believe further research into ROV use is warranted as they provide a safer alternative to diver-based techniques and have the ability to extend into deeper-waters. We recommend that for fish diversity and abundance assessments a sampling strategy that combines BRUVs and a transect method should be adopted.

2.2. Introduction

Effective fisheries and conservation management relies upon quality information to describe the diversity, abundance, size structure and biomass of fish populations, to quantify natural variation, and identify the effects of anthropogenic activities (Cinner et al., 2016; Jennings and Kaiser, 1998; Jennings et al., 1999; Jennings et al., 2014; McClanahan et al., 2011). When collecting ecological data, it is crucial that the experimental design and the sampling technique chosen is capable of detecting change over an appropriate spatial and temporal scale (Bach et al., 2019; Harasti et al., 2015; Holmes et al., 2013). Researchers also need to understand and evaluate the biases of a potential technique (e.g. bait attraction and system avoidance), as data from different techniques can result in different insights and biological interpretations (Logan et al., 2017; Watson et al., 2010). As fisheries and conservation managers are often concerned with losses in diversity and changes in the biomass of an ecosystem, non-extractive techniques often present a more desirable sampling approach (Murphy and Jenkins, 2010; Pelletier et al., 2011). The use of non-extractive techniques also enables sampling in management areas, such as no-take marine reserves, where extractive activities are prohibited (Cappo et al., 2003, 2006).

One of the most common non-extractive survey techniques is underwater visual census (UVC). UVC is a simple and comparatively inexpensive method for identifying and estimating fish abundances (Brock, 1954; English et al., 1997). However, the limitations and biases of UVC surveys include; the requirement of skilled observers who can identify fish *in-situ* at the species level (Holmes et al., 2013), inter-observer variations between divers (Harvey et al., 2001a; Watson et al., 1995), fish avoidance or attraction to divers (Gray et al., 2016; Lindfield et al., 2014; Watson and Harvey, 2007) and SCUBA restrictions with depth and bottom time (Harvey et al., 2001b). The use of underwater video can help to overcome some of these observer biases when sampling fishes (Goetze et al., 2015; Harvey et al., 2001b, 2002a, 2004; Harvey and Shortis, 1995) and have been used increasingly since 2000 (see reviews in Cappo et al. (2001, 2003, 2006), Harvey and Mladenov (2001c), Murphy and Jenkins (2010)). In particular, the use of stereo-video systems improves the accuracy and precision of fish length estimates, allowing for a more robust assessment of fishes in regards to size structure and biomass (Harvey et al., 2001b, 2002b; Harvey and Shortis, 1998). Underwater stereo-video has been adapted to a range of observational based techniques, including baited remote underwater stereo-video systems (Stereo-BRUVs; herein referred to as BRUVs), diver operated stereo-video (Stereo-DOV; herein referred to as DOV), and other transect based stereo-video methods.

For transect based methods, stereo-video allows three-dimensional definition of the transect boundaries and therefore accurate density estimates (Harvey et al., 2004; Harvey and Shortis, 1995). Underwater video can also provide permanent recordings *in situ* removing the need for skilled observers who can identify fish in the field (Goetze et al., 2015; Holmes et al., 2013; Langlois et al., 2010). However, like UVC the use of divers can influence fish behaviour, and therefore underestimate diver shy and cryptic species, or overestimate those that are attracted to diver activity (Cappo et al., 2003; Lindfield et al., 2014). In addition, the reliance on SCUBA based techniques often restricts surveys to shallow waters no deeper than 30m (Cappo et al., 2003; Harvey et al., 2001b). Closed circuit and semi-closed circuit rebreather systems can address these limitations by reducing the behavioural impact of SCUBA diver bubbles whilst increasing the operating depth (Gray et al., 2016; Lindfield et al., 2014). However, the costs of training, equipment, and the health and safety considerations of the maximum operating depths (normally ~100m) are critical to consider (Norro, 2016; Sieber and Pyle, 2010). Because of these inherent operational limitations with diving techniques, researchers are increasingly adopting remote sampling techniques to survey fishes (Logan et al., 2017; Seiler et al., 2012; Warnock et al., 2016).

BRUV systems have become a widely used method for surveying fish assemblages across a range of habitats and depths (Cappo et al., 2003, 2006; Harvey et al., 2007, 2012). In comparison to diver-based techniques the use of BRUVs overcomes biases associated with fish avoidance of divers, resulting in greater statistical power and higher counts for larger-bodied predatory species. The use of bait increases the proportion of predatory species, which are often targeted by fisheries, without compromising counts of herbivorous or omnivorous fishes (Harvey et al., 2007; Langlois et al., 2010; Watson et al., 2010). But, BRUVs have acknowledged inherent biases (Langlois et al., 2010; Watson et al., 2010; Willis et al., 2000) including that the area covered by the bait plume and the distance from which fish are attracted to the camera system is unknown (Cappo et al., 2003, 2006; Harvey et al., 2007). As a result, fish estimates are recorded in measures of relative abundance, MaxN (i.e. the maximum number of individuals observed in one frame for each species), as opposed to density (per unit area) (Ellis and DeMartini, 1995; Logan et al., 2017). Additionally, it is recommended that a minimum distance of 250-500 m is used between simultaneously deployed BRUVs to avoid the potential for double counts of the same fish or shark which may swim between camera systems (Cappo et al., 2001, 2003, 2006; Harvey et al., 2007, 2018), therefore limiting the use of BRUVs for the assessment of small or patchy habitat features.

While most studies use one type of observational technique for the assessment of fishes, it is preferable to combine transect and BRUVs sampling techniques to sample a larger diversity of fishes (Langlois et al., 2010; Logan et al., 2017; Watson et al., 2010, 2005; Willis and Babcock, 2000). This allows observations of different species which would normally be biased towards one type of sampling technique (Murphy and Jenkins, 2010). In the context of ecosystems based management, it is not only important to sample those species that are targeted by fisheries, but also non-target species, which can be indicators of ecosystem health and indirect effects of anthropogenic pressures (Langlois et al., 2017; McLaren et al., 2015; Metcalf et al., 2011). Ongoing advancement in technology provides new platforms that can be used to conduct stereo-video transects. It is important, however, to evaluate the limitations and biases of new sampling techniques to see how they compare and complement existing techniques.

The development of new remote transect sampling methods presents an alternative to diver-based techniques. These may overcome some of the limitations of SCUBA based methods, enhancing sampling capabilities for a more complete assessment of fish assemblages. For example, slow towed stereo-video (herein referred to as STV) has been explored as an alternative to diver-based techniques, showing comparable measures to DOV (Warnock et al., 2016). However, a challenge of this method is the controlled navigation of the system which can lead to limitations in the data collected. An alternative approach is the use of a remotely operated vehicle (ROV) which increases the manoeuvrability of the camera system along the water column and over narrow strips of reef. While evidence suggests that the use of an ROV for sampling fish may influence fish behaviour, these conclusions are based on surveys using work-class ROVs, which are large, create a lot of sound underwater and generally use artificial lights (Stoner et al., 2008). The use of a micro stereo-video-ROV (herein referred to as ROV) which does not rely on artificial lights or produce high noise outputs may overcome some of these limitations (Trenkel et al., 2004a).

We aimed to compare and contrast the fish assemblages sampled by four stereo-video based methods: DOV, STV, ROV, and BRUVs in a temperate system along a limestone ridge within Geographe Bay, Western Australia. Here, comparisons were made in the context of (1) fish assemblage composition, total number of species and individuals, and the density of focal species, (2) the statistical power of each technique to detect change, and (3) cost effectiveness.

2.3. Methods

2.3.1. Study area

Sampling was conducted during November 2014, in Geographe Bay (33°35′S, 115°12′E), southwestern Australia, along a ~5 km limestone ridge, ranging in water depths of 15-20 m (Fig. 2.1). The formation is predominantly covered in brown macroalgae (*Ecklonia radiata*), and is surrounded by patches of seagrass (*Posidonia* spp. and *Amphibolis* spp.) that cover sandy substrate (Galaiduk et al., 2018; McMahon et al., 1997).

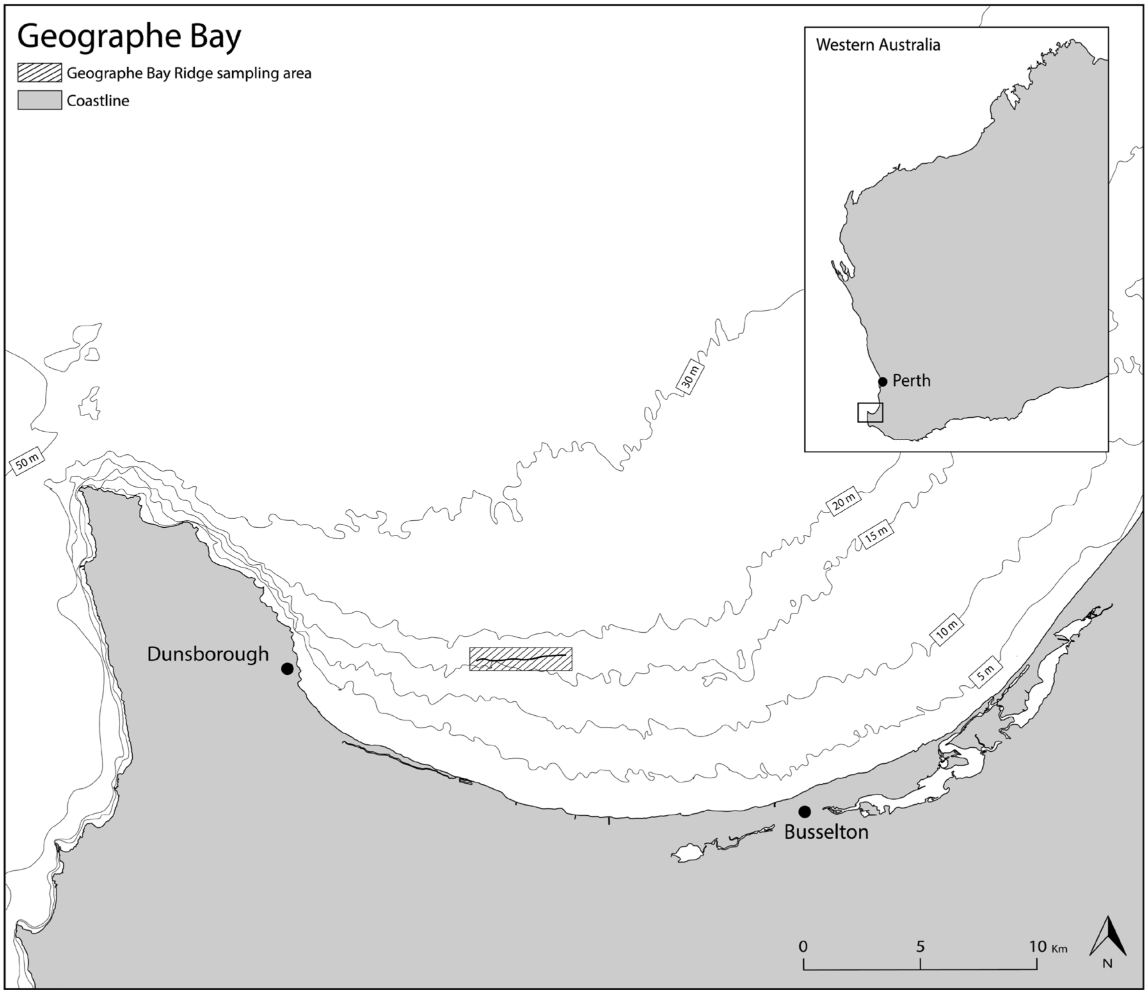


Fig. 2.1 Location of the limestone ridge and sampling area in Geographe Bay, Western Australia.

2.3.2. Experimental design

For this study, 36 belt transects (25x5m) were completed for each of the transect methods (DOV, STV, and ROV) with a separation of >10 m between each transect (Saunders et al., 2014). Eighteen replicates were completed for BRUV surveys with a separation of at least 250 m to minimise the possibility of fish recorded in one camera system swimming into the field of view of another (Cappo et al., 2001, 2003, 2006; Harvey et al., 2007, 2018). With each technique the samples (transects or BRUV deployments) were positioned haphazardly within the target location along the limestone ridge. A lower replication was used for BRUV when compared to transects, due to a combination of the limited length of the reef and required minimum distance between BRUV systems. BRUV sampling was completed the day before transect surveys to avoid any potential influence of bait on transect data. Sampling was conducted between 09:00 and 16:00 h to minimise diurnal variations in fish assemblage (Hammerschlag et al., 2017; Myers et al., 2016; Willis et al., 2006).

2.3.3. Stereo-video systems

System design and configuration for each sampling technique followed those described in Harvey and Shortis (1995, 1998) and Watson et al. (2010). All video cameras recorded at a resolution of 1920 ×1080.

2.3.3.1. Diver operated stereo-video (DOV)

DOV sampling was completed by two SCUBA divers, following procedures as described in Goetze et al. (2019). Transects were measured using a Chainman cotton counter and biodegradable cotton. Each transect took an average of 1 min and 28 s (± 4 s SE) to complete. The DOV used SONY® HDR CX700 cameras and were set at 1080p recording 50 frames per second (fps) with medium field of view (FOV) to minimise motion parallax associated with the stereo-video system and the fish moving simultaneously.

2.3.3.2. Slow towed stereo-video (STV)

STV operations were conducted using a frame, fitted with two Sony® CX12 cameras (set at 1080p, 25 fps, medium FOV), connected via a coaxial umbilical to provide a live feed for height adjustments in real time, following procedures in Warnock et al. (2016). The system was deployed off the front of a small vessel (4.6 m) with the motor used to keep the camera system along the reef while towing. We estimated from GPS tracks that a slow towing speed of ~0.3 ms^{-1} could be maintained by thrusting the motor when required. Each transect took an average of 1 min and 41 s (± 11s SE) to complete.

2.3.3.3. Micro stereo-video remotely operated vehicle (ROV)

ROV surveys were carried out using a SeaBotix vLBV300 equipped with a stereo-video system using two GoPro® Hero 3 Silver camera (set at 1080p, 30 fps, medium FOV) mounted to vehicle. The ROV was 390 mm high, 625 mm long and 390 mm wide. An Ultra Short Baseline Positioning system was used to determine transect length (25 m). During ROV sampling the survey vessel was anchored. The operating range of the ROV was limited to ~ 100m either side of the vessel by the length of the tether. This range allowed ~6 × 25m transects, with at least a 10m separation between transects to be conducted from each anchored position. The vessel was relocated on six occasions to record the 36 transects. On average transects took 1 min and 20 s (± 3 s SE) to complete.

2.3.3.4. Baited remote underwater stereo-video (BRUV)

Sampling was carried out using eight BRUV systems, each equipped with two Sony® HDR CX12 cameras (set at 1080i, 25 fps, medium FOV). Each system was baited with ~1 kg of pilchards (*Sardinops sagax*) and deployed from a vessel with a line up to a surface marker buoy. BRUVs remained on the seafloor for a sampling period of 60 min. Prior to each deployment, bait was crushed to promote fish oil dispersal. Simultaneous deployments were conducted in the field to ensure maximum efficiency was achieved.

2.3.4. Calibration and video analysis

Each stereo-video system was calibrated prior to and after field work using the software 'CAL' (http://www.seagis.com.au/bundle.htm), following published procedures in Harvey and Shortis (1998) and Shortis and Harvey (1998). Imagery from each system was then viewed and analysed using the program 'EventMeasure' (http://www.seagis.com.au/event.htm). Fish observed within the transect sampling area (5 m width) were counted and identified to a species level when possible. Fish that were observed outside the transect area or measured further than 8 m distance from the system were automatically rejected by the software and not included in the analyses. We chose 8 m as the maximum distance for measurement because measurement accuracy and precision declines with the systems used beyond 8 m (Harvey et al., 2010). For BRUV footage, relative abundance was measured in terms of MaxN to avoid repeated counts and measures of the same fish at a single system (Cappo et al., 2001, 2003; Willis and Babcock, 2000). An 8 m field of view was also used during BRUV video analyses.

2.3.5. Statistical tests

A single factor PERMANOVA design (Technique*:* 4 levels, fixed: DOV, STV, ROV, and BRUVs) was used to assess statistical differences in the multivariate fish assemblages observed by the four methods. A single factor ANOVA design (Technique*:* 3 levels, fixed: DOV, STV, ROV) was used for univariate analysis to compare the numbers of fish, numbers of species and numbers of each focal fish species sampled by the three transect techniques. Statistical comparisons between BRUVs and transect methods were not included in any of the univariate analyses due to fundamentally different units of measurement (i.e. MaxN vs. density). BRUV data was included alongside univariate figures to illustrate the differences in the means sampled per replicate. Prior to statistical analysis we removed count data on *Trachinops noarlungae* and *Pempheris* spp. due to the high numbers of individuals recorded which skewed the statistical analysis and interpretation of data.

2.3.5.1. Multivariate analysis

To test for differences in the assemblage of fishes recorded by each technique, a non-parametric permutational analysis of variance (PERMANOVA) was performed using the PERMANOVA + add on to the Primer 6 statistical package (Anderson et al., 2008). To reduce the weighting of common species over rare species, the dataset was transformed using a square root-transformation. Analyses were based on a Bray Curtis resemblance matrix using 9999 permutations. Where a statistically significant test for Technique was returned, a post-hoc (pairwise comparison) was performed to determine statistical differences between each of the techniques. To reduce the chance of a false discovery (type 1 error) a Holm-Bonferroni (H-B) correction was applied to alpha values in pairwise comparisons and used to determine significance (Holm, 1979). Alpha values are unique to each post-hoc test depending on their p-value ranking, therefore a statistical difference is determined when p < αH-B. These formal test results were then interpreted by examining principal coordinate analysis (PCO) and constrained canonical analysis of principal coordinates (CAP) plots (Anderson and Willis, 2003). PCO and CAP analyses were completed between all techniques and repeated on the transect measures only. Vectors were overlaid onto the CAP plots which illustrate the strength and direction of the Pearson's correlation of the abundance of individual fish species to the CAP axes. A Pearson correlation of ± 0.5 was used to identify the most influential species on the CAP involving all techniques, while a value of ± 0.35 was selected for analysis focused on the transect techniques only. A leave-one-out allocation test was also used to estimate and classify how distinct samples were relative to each technique.

2.3.5.2. Univariate analysis

Separate PERMANOVA tests were used on single variables (univariate case) to determine whether the density and species richness of fishes, and densities of focal fish species differed between DOV, STV and ROV sampling techniques. Focal species were designated according to those identified in the transect CAP plot (Pearson correlations) and those considered to have management value (i.e. non-targeted and targeted indicator species). The non-target indicator species included *Coris auricularis* (Western king wrasse), *Chromis klunzingeri* (Blackhead puller) and *Neatypus obliquus* (Footballer sweep) (Metcalf et al., 2011), while the target species included *Glaucosoma hebraicum* (Western Australian dhufish), *Bodianus frenchii* (Foxfish), and *Epinephelides armatus* (Breaksea cod) (Cossington et al., 2010; Hesp et al., 2002; Moore et al., 2007). Where PERMANOVA returned a significant test for Technique, post-hoc tests were performed. To account for type 1 error in post-hoc tests, a Holm-Bonferroni correction was used to adjust alpha values to determine statistical significance (Holm, 1979). Statistical differences were determined when P < αH-B. In some cases post-hoc tests could not achieve reasonable permutation value (i.e. unique permutations <400) and Monte Carlo bootstrapping was conducted to obtain p-values, reported as *P(MC).* A Venn diagram was constructed to illustrate the cumulative percentage of species sampled with each of, and combinations of the techniques.

2.3.6. Power analysis

The statistical power of DOV, STV, ROV, and BRUVs to detect hypothetical changes in the observed means and variation of the numbers of individuals, species richness and abundance of focal species were estimated in G*Power (Faul et al., 2009), using a 1-way fixed effect ANOVA, with 2 levels ($\alpha = 0.05$). Changes of 20 and 50% were selected for analyses, where power ($1 - \beta$) for each technique could be predicted with increasing sample size. These effect sizes are considered small and moderate respectively for ecological changes (Harvey et al., 2001a, 2001b, 2012; Langlois et al., 2010).

2.4. Results

A total of 3180 individual fish belonging to 79 species were observed across the four techniques. BRUVs sampled a greater proportion of individuals and species (n = 1200; s = 65) followed by DOV (n = 915, s = 48), STV (n = 602, s = 40) and ROV (n = 463, s = 34). BRUVs also recorded the highest number of unique species (s = 21) with DOV (s = 4) having the next highest, followed by STV (s = 2), and ROV (s = 1) (Appendix; Table App. 2.1). When the number of species recorded on BRUVs were combined with a transect method, the cumulative percentage of observed species increased by ~10%. BRUV combined with STV recorded 92.41% of the total number of species observed across all methods, followed by BRUV combined with DOV or ROV (both yielding 91.14%; Fig. 2.2).

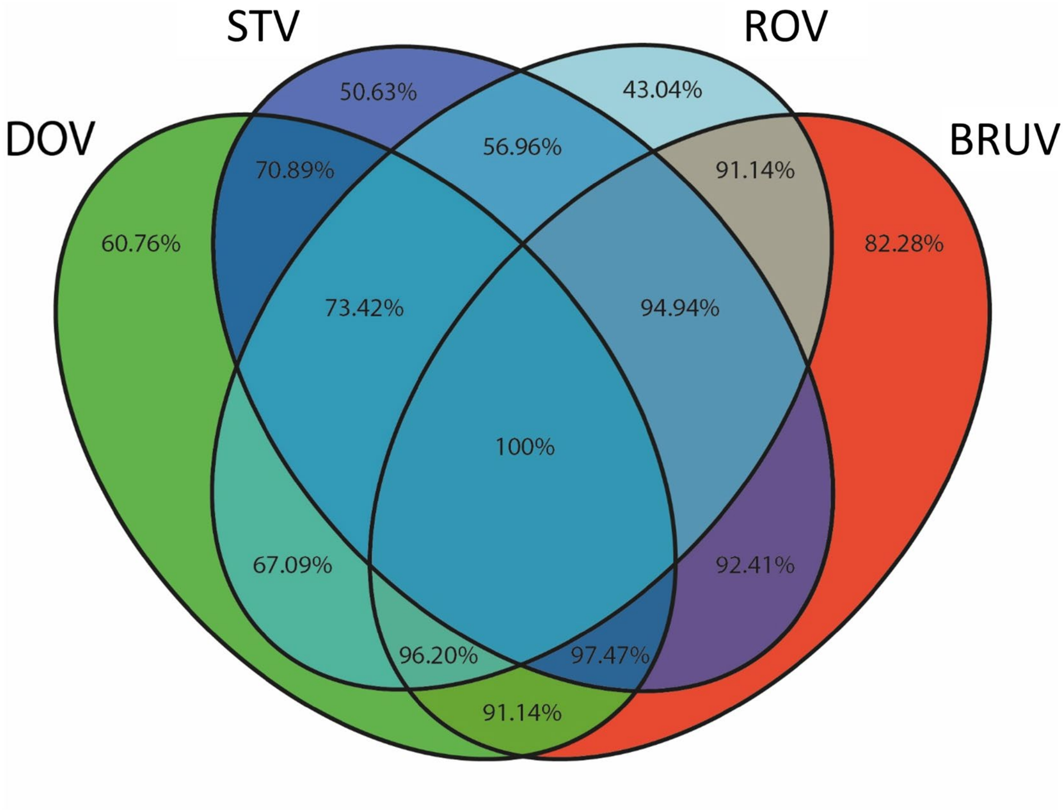


Fig. 2.2 Venn diagram showing the cumulative percentage of species recorded by each technique, and in combinations of techniques (DOV, STV, ROV, and BRUVs). Percentages are relative to the total number of species surveyed along the ridge, s = 79.

2.4.1. Fish Assemblages

There was a significant difference in the assemblages of fish observed between techniques (Technique $_{(3,\ 122)}$, MS = 17062, Pseudo-F = 10.111, $P < 0.001$), with a post-hoc pairwise test indicating all methods to be statistically different from each other ($P < \alpha$H-B). There was a distinct separation between transect and BRUVs replicates in the ordinations (Fig. 2.3A and B). This was driven by small carnivorous fishes such as *C. auricularis*, *O. lineolatus,* and *Pseudocaranx* spp. that mainly feed on benthic invertebrates and are attracted to bait (Fig. 2.3B). A leave one out allocation success test confirmed that BRUV sampled a unique suite of species, with 100% of BRUVs samples being correctly classified (Table 2.1). For transect methods there was some overlap, with correct allocation ranging from 58% to 80% (Table 2.1). When analysed without BRUVs, separation between transect methods was greater (Fig. 2.3D), but the unconstrained PCO illustrates considerable overlap between the transect methods (Fig. 2.3C).

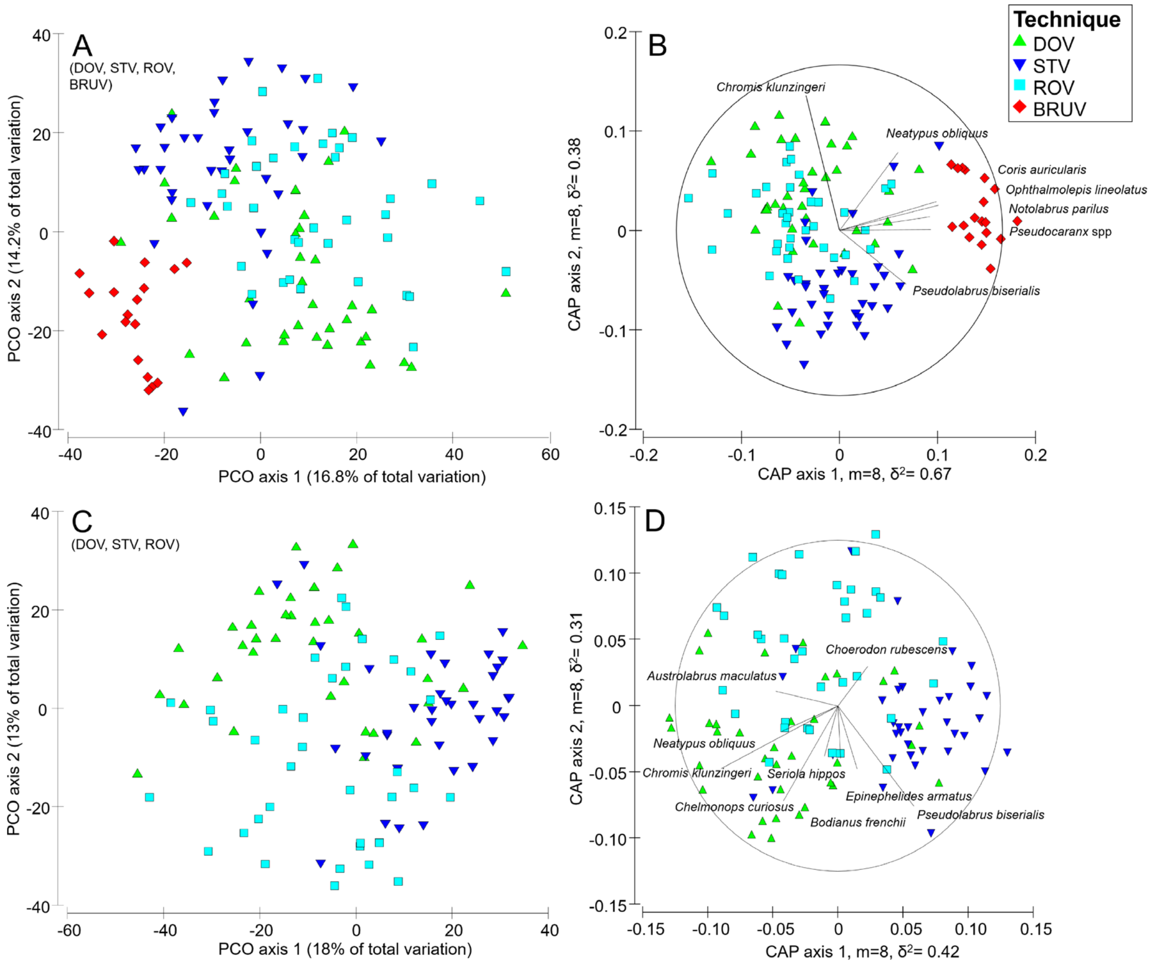


Fig. 2.3 Principle Coordinate Ordination (PCO) and Canonical Analysis of Principal Coordinates (CAP) plots of the fish assemblage sampled by all techniques: DOV, STV, ROV, and BRUVs (m = 8) (A, B); and transect only techniques: DOV, STV, ROV (m = 8) (C, D). Ordinations are based on a square root transformation and Bray-Curtis resemblance matrix. Characteristic species are shown on CAP vector overlays with (B)

Person correlations greater than ± 0.5 to either of the CAP axes and (D) Pearson correlations greater ± 0.35.

Table 2.1 Leave-one-out allocation success of observations to technique: Cross validation, (m = 8).

Technique	DOV	STV	ROV	BRUVs	Total	Success (%)
DOV	21	7	7	1	36	58.33
STV	2	29	4	1	36	80.57
ROV	7	5	23	1	36	63.89
BRUVs	0	0	0	18	18	100

2.4.2. Number of individuals

There was a significant difference in the mean density of fishes recorded between transect methods (Technique $_{(2,\ 105)}$, MS = 1488.9, Pseudo-F = 7.163, P < 0.001). A pairwise comparison showed that DOV recorded on average 49.29% more fish compared to ROV (t = 4.083, P(MC) < 0.001, αH-B = 0.017), and 34.06% more fish than STV, although this comparison was not statistically significant (t = 2.189, P(MC) = 0.037, αH-B = 0.025). No significant difference was found between STV and ROV (t = 1.259, P(*MC*) = 0.220, αH-B = 0.05; Fig. 2.4A). The power to detect a 20 and 50% change in the mean number of individuals was greater for BRUVs (Fig. 2.5A). For example, BRUVs sampling requires roughly half the number of samples to achieve adequate power (β = 0.8; n = ~ 7 for 50% change) compared to the other transect methods (e.g. ROV n = ~ 13 and DOV: n = ~ 17). Between transects methods, ROV had the greatest power, followed by DOV and STV.

2.4.3. Species richness

There was a significant difference in the mean number of fish species observed between transect methods (Technique $_{(2,\ 105)}$, MS = 72.898, Pseudo-F = 8.656, $P < 0.001$). A post-hoc test indicated that ROV recorded significantly fewer species per 125 m^2 in comparison to DOV ($t = 4.208$; $P < 0.001$, αH-B = 0.017) and STV ($t = 3.250$; $P = 0.002$, αH-B = 0.025), recording on average 29.17% and 15% fewer species, respectively. No significant difference was found between DOV and STV ($t = 0.639$; $P = 0.525$, αH-B = 0.05; Fig. 2.4B). A power analysis showed that DOV had a greater statistical power to detect changes in the numbers of species when compared to other transect techniques. However, unlike number of individuals, the power to detect changes was comparable across all techniques, especially at 50% detection (Fig. 2.5B).

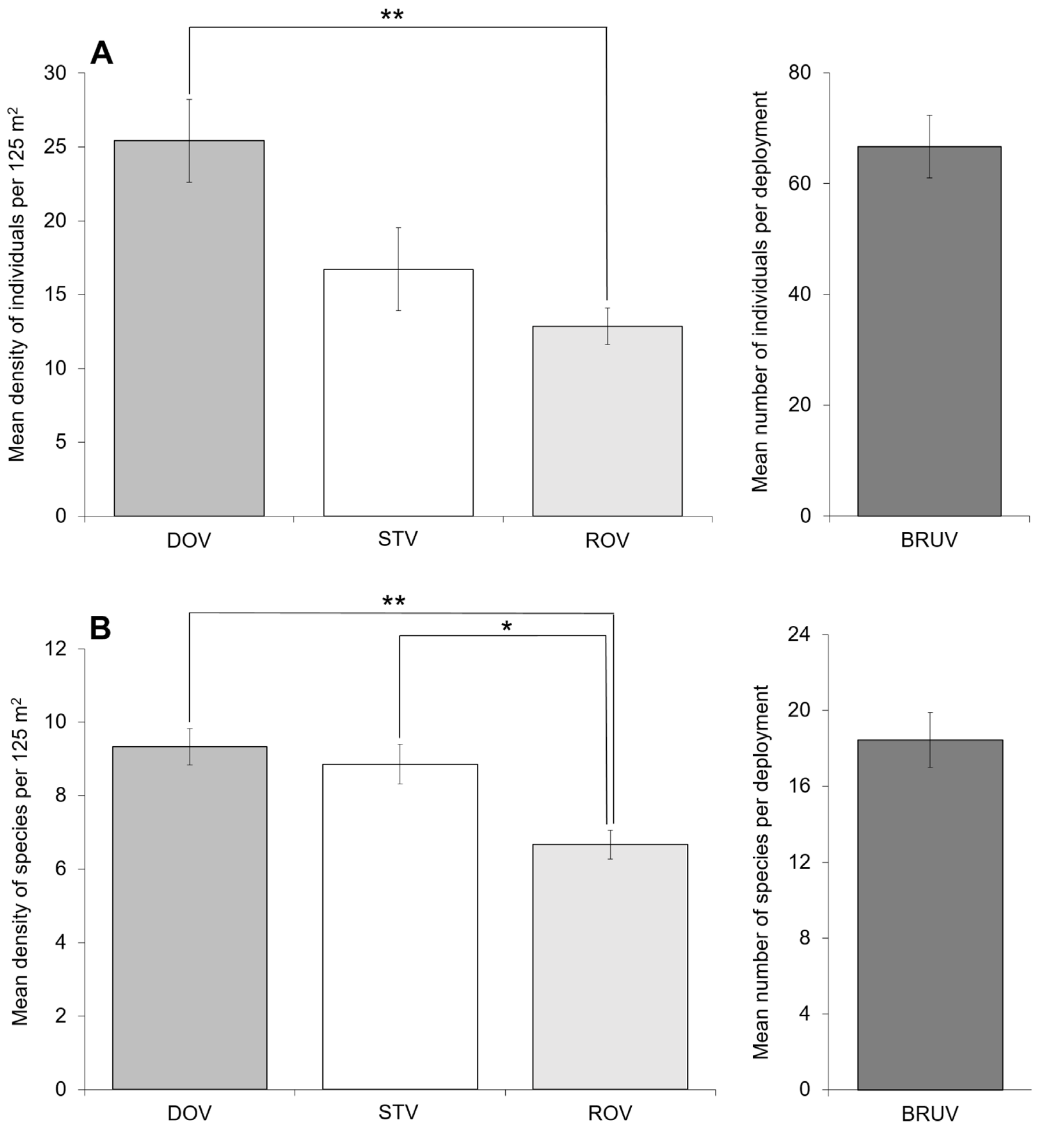


Fig. 2.4 (A) Mean (±1 SE) density of individuals recorded by transect methods DOV, STV, ROV, and mean (±1 SE) abundance of individuals per BRUV deployment, and (B) mean (±1 SE) density of species recorded by transect methods DOV, STV, ROV, and mean (±1 SE) number of species recorded per BRUV deployment. Probability values at ($P < 0.05$) and ($P < 0.001$) are represented as * and **, respectively.

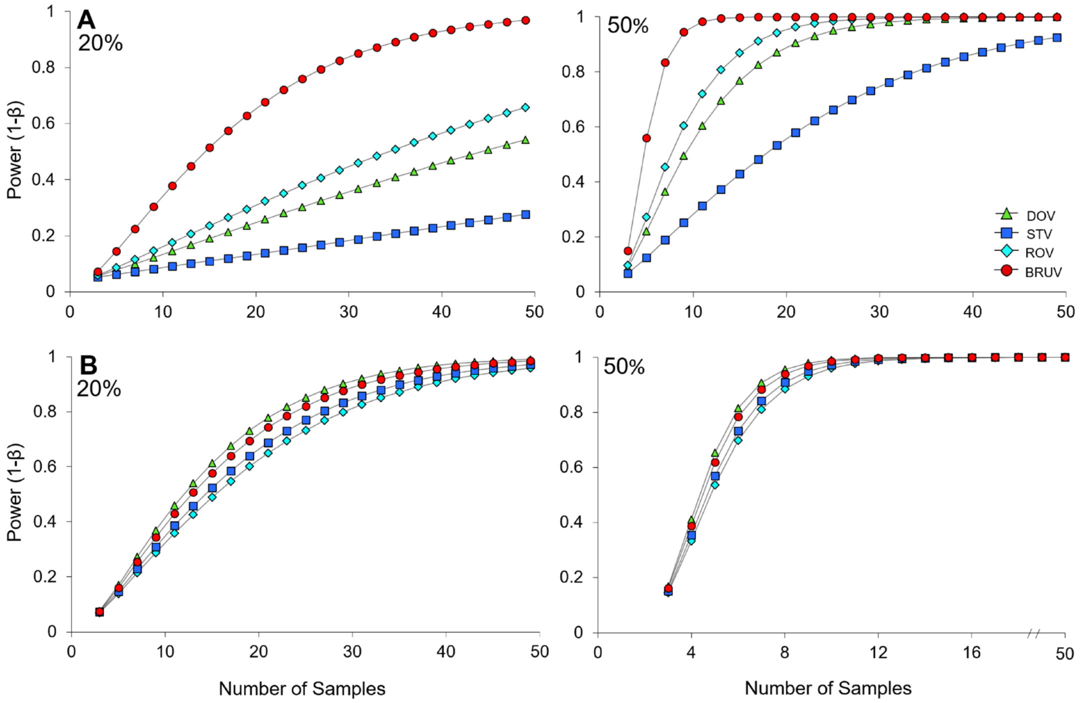
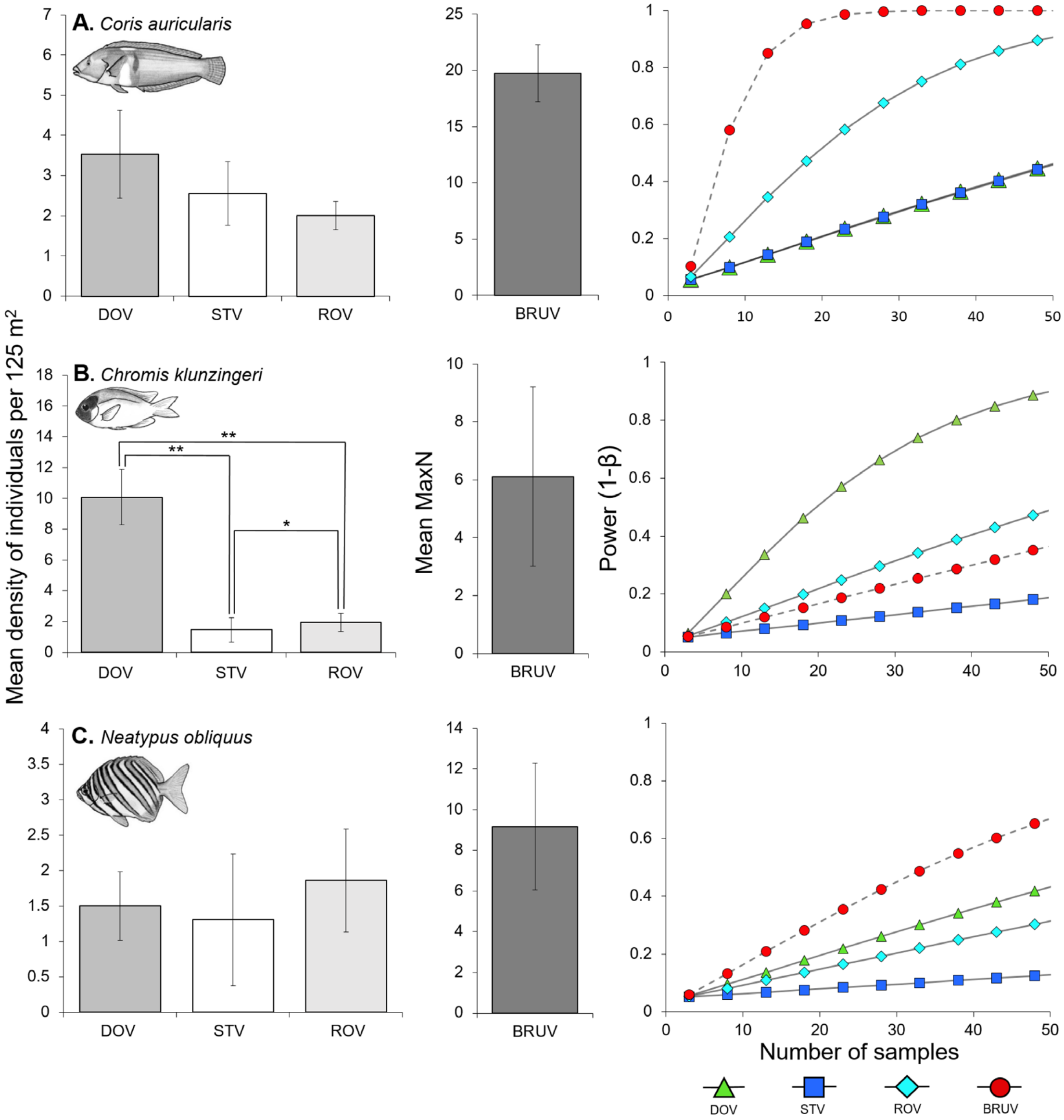


Fig. 2.5 Power to detect 20 and 50% change in the (A) mean number of individuals and (B) mean number of species for techniques DOV, STV, ROV, BRUV, with increasing number of samples ($\alpha = 0.05$).

2.4.4. *Focal species*

The mean density amongst focal species was statistically similar across transect methods ($P > 0.05$; Fig. 2.6A, C-F), with the exception of *C. klunzingeri* (MS = 44.811; $F_{2,105} = 21.804$; $P < 0.001$; Fig. 2.6B), with DOV recording on average 80% more individuals of this species than either STV and ROV ($P < \alpha$ H-B). For five of the six focal species, BRUVs sampling had the highest statistical power (Fig. 2.6A, C-F), requiring fewer samples to achieve adequate power ($\beta = 0.8$). Conversely, when it came to *C. klunzingeri,* greater power was achieved with DOV, with increasing sample size (Fig. 2.6B). Across all focal species, STV and ROV were less consistent in power to detect 50% change.


A. Coris auricularis
B. Chromis klunzingeri
C. Neatypus obliquus
Mean density of individuals per 125 m²
Mean MaxN
Power (1-β)
Number of samples
DOV
STV
ROV
BRUV
**
**
*
0
1
2
3
4
5
6
7
0
5
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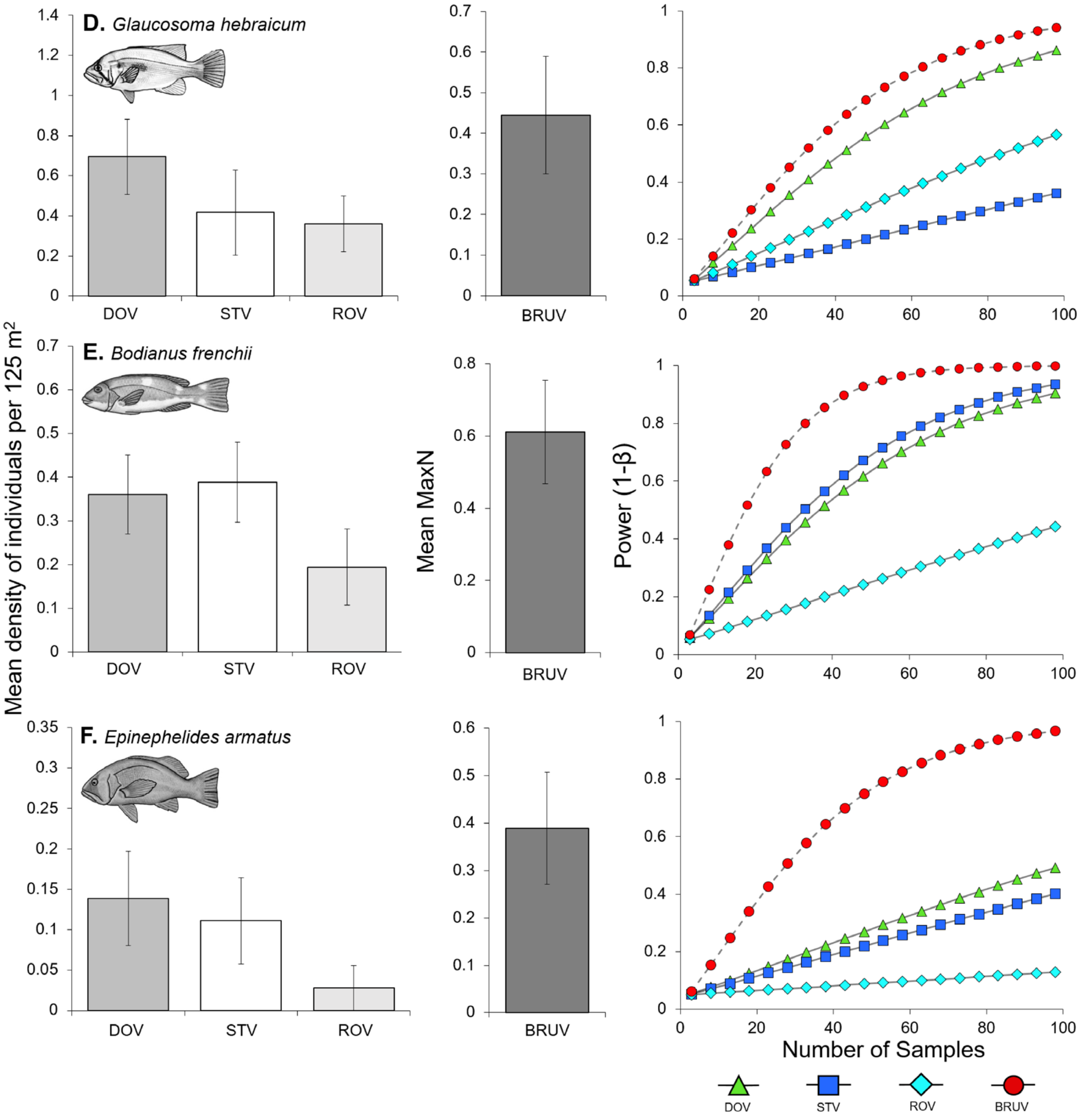


Fig. 2.6 Mean (± 1 SE) density sampled by transect methods DOV, STV, ROV, mean (± 1 SE) MaxN per BRUV deployment, and power analysis to detect 50% change in mean abundance (α = 0.05) of (A) *Coris auricularis*, (B) *Chromis klunzingeri*, (C) *Neatypus obliquus*, (D) *Glaucosoma hebraicum*, (E) *Bodianus frenchii*, and (F) *Epinephelides armatus*. Probability values at ($P < 0.05$) and ($P < 0.001$) are represented as * and **, respectively.

2.5. Discussion

We show that the combination of BRUVs with one or more transect based methods provides the most comprehensive assessment of this temperate reef fish assemblage. The comparison of data collected by DOV, STV, ROV and BRUVs demonstrated that no one technique sampled all species observed, and that a combination of techniques observed more species along the reef. This is consistent with previous studies, where a combination of BRUVs with; UVC (Willis and Babcock, 2000), DOV (Watson et al., 2010), DOV and UVC (Goetze et al., 2015), DOV and RUV (Watson et al., 2005), or towed video (Logan et al., 2017) was optimal in providing a more holistic interpretation of fish assemblages compared to a single method approach. As in other studies (Goetze et al., 2015; Langlois et al., 2010; Logan et al., 2017; Watson et al., 2010; Willis and Babcock, 2000), BRUVs sampled the greatest number of species, many of which were mobile species from higher trophic groups. However, some studies have found an opposing pattern, with UVC recording greater number of species than BRUVs, in particular smaller bodied and cryptic species (Colton and Swearer, 2010; Lowry et al., 2012). BRUVs sample higher counts of targeted predatory species due to the attractant properties of bait (Cappo et al., 2004, 2006; Harvey et al., 2007) as well as species that may actively avoid contact with divers (Cappo et al., 2003; Chapman et al., 1974; Chapman and Atkinson, 1986; Watson et al., 2005; Watson and Harvey, 2007). Avoidance behaviours of fish towards divers have been documented, showing more pronounced interactions for target species that inhabit areas with high fishing pressure (Kulbicki, 1998; Lindfield et al., 2014). Conversely, transect methods are more likely to count smaller bodied species that are habitat affiliated, less mobile, and not attracted to bait (Langlois et al., 2010; Watson et al., 2010).

Transects sample a greater area of habitat (125 m^2) compared to the field of view of a stationary BRUV (34.5 m^2). Consequently, transect methods sample fish that are associated with micro-scale habitat features (e.g. sponges, corals and other benthic components) that may not be sampled by a BRUV unless an individual's territory is encompassed in the field of view (Cundy et al., 2017). The ability to encompass a greater area of habitat in transect surveys also means habitat heterogeneity and fish-habitat associations can be explored in greater detail (e.g. Galaiduk et al., 2017; Saunders et al., 2014). As BRUVs are stationary and attract fish to them gaining such information is not possible.

In this study, the combination of any one of the transect techniques with BRUVs resulted in an approximate 10% increase in the numbers of species sampled compared to BRUVs alone. In addition, STV, DOV and ROV all provided comparable measures of density for five of the six indicator species. As each of the transect techniques performed similarly, it is important to look at the advantages and disadvantages of each in more detail.

DOV recorded higher densities of fish and number of species when compared to ROV, but was statistically similar to STV. This is in contrast to a previous study which reported significantly more fish species and individuals with DOV compared to STV (Warnock et al., 201*6*). Differences between techniques could be explained by variation in the height at which each system recorded imagery above the reef or differences in the behavioural reaction of fish towards each system. Due to operational considerations ROV and STV systems were operated higher above the reef than DOV, which may result in poor capacity to observe smaller species and individuals which were in close proximity to the reef. This effect was most pronounced for *C. klunzingeri* with higher counts being recorded by DOV possibly as a result of divers following the contours of the reef more closely. STV operations were dependent on a live feed from the system to the skipper where adjustments along the narrow reef were made. However, the skipper's reaction to the live feed was often delayed making it challenging to closely follow the reef's topography and react to depth changes (Warnock et al., 2016). STV imagery was also more difficult to analyse due to small discrete movements in response to swell and chop. This movement was observed to influence fish behaviour, particularly when there was a larger, more sudden movement of the system through the water column. The influence of surface conditions on the STV body makes the system challenging to use in swell, but the effect could be reduced with a dampener on the winch. The ROV does not suffer from this issue and the distance above the bottom can be controlled by the operator. In this instance we hypothesise that if the system had been recording closer to the benthos, we would have recorded a similar number of species and fish to DOV. This could be resolved by operator training and briefing to maintain a specified height above the reef similar to that of DOV (50 cm, Goetze et al., 2019). Avoidance behaviours towards vessel and/or noises associated with work-class ROV operations have also been shown to influence the number of fish observed, limiting encounters across the field of view (Logan et al., 2017; Stoner et al., 2008). It is likely that the presence of a tether, and or noise produced from the ROV and STV elicit different behavioural responses when compared to DOV. While the behavioural influence of ROVs remain poorly understood (Ryer et al., 2009; Stoner et al., 2008), it is possible to assess this by comparing the minimum approach distance (MAD) for different sizes of fish with other sampling techniques (Goetze et al., 2017). It is important that future studies assess these potential behavioural biases across a range of temperate and tropical habitats.

As reported in previous studies (Cundy et al., 2017; Harvey et al., 2007, 2012; Langlois et al., 2010; Watson et al., 2005), BRUVs proved to be the most powerful tool for assessing the characteristics of fish assemblages. This reflected the lower variance achieved between and within samples, and the higher counts of fish in response to bait (Cappo et al., 2003; Harvey et al., 2007). This lower variance could also be attributable to the use of MaxN which is considered to be conservative in its estimates of relative abundance of fish (Cappo et al., 2001; 2003; Ellis and DeMartini, 1995; Willis and Babcock, 2000). While statistical power was comparable between BRUVs and transect methods for species richness, the variability in the number of individuals recorded was relatively high within and among transect methods, resulting in lower achieved power. Transect methods produce high variances due to spatial heterogeneity within the target habitat, which is why it is important to optimise transect length and width for the habitat being sampled (McCormick and Choat, 1987). For five of the six indicator species, BRUVs also had greater power to detect change compared to the transect methods. Again, this is due to variation between sampling units, and that the majority of these species are attracted by the bait. However, *C. klunzingeri* was the exception, which is a schooling species that is associated with reef, and is not attracted to bait, resulting in lower variance and more power with DOV. As a general rule, species with these characteristics are more commonly observed by transect methods (Watson et al., 2010).

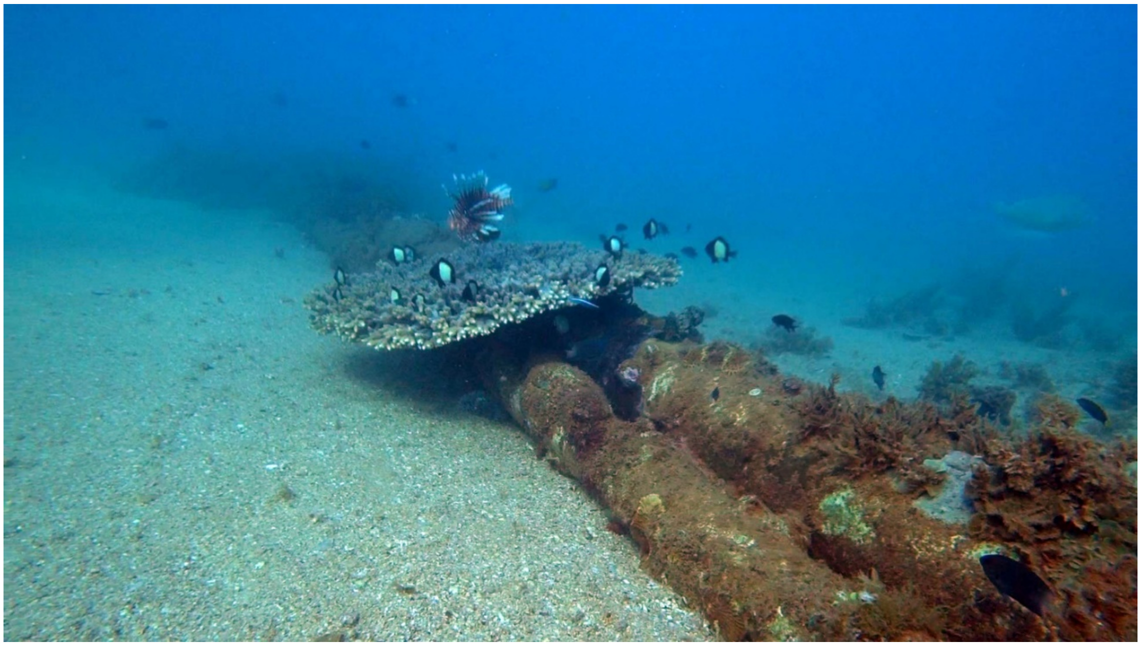
An important consideration of collecting and analysing data is evaluating the associated costs and effort of a method. Depending on the sampling method used, cost can vary considerably (Langlois et al., 2010). In this study, field and analysis costs were comparable between transect methods (Appendix; Table App. 2.2). However, safety restrictions on operating depth and repetitive diving can limit sampling effort using the DOV method. We were recording 12 (25 x 5 m) transects during one dive allowing a maximum of 36 transects per day per diver pair. By comparison STV and ROV techniques have the potential to sample a far greater number of transects per day, with the effect of increasing power, as they can be deployed multiple times during a day. BRUV systems have a similar advantage in that it is possible to make simultaneous deployments of multiple BRUV systems, which can greatly increase fieldwork efficiency (Cappo et al., 2001, 2003, 2006; Harvey et al., 2013). A method that has lower statistical power will require more replicates, which extends the field time required, resulting in higher associated costs. While DOV represents the most cost effective solution for detecting a 50% change in *C. klunzingeri* density estimates (based on field and analysis costs), the low power attained across the other focal species would result in higher sampling effort and cost when compared to BRUVs. For example, to detect a 50% change with 80% power in the relative abundance or density of *C. auricularis,* 13 BRUV deployments or >100 DOV transects would be required (Appendix; Table App. 2.3). This would involve one day of fieldwork for the BRUVs, and ~32.5 h of image analysis. By comparison, DOV would require three days in the field and ~50 h of processing. In addition to added time in the field and laboratory, DOV surveys are often limited to shallow water assessments (< 30m). The limitations of a diver based method often compromises fieldwork efficiency due occupational health and safety associated with fieldwork spanning multiple days. Having a lower statistical power compared to BRUVs, diver based stereo-video sampling will require a greater number of days in the field to obtain an equivalent statistical power. With increasing pressure to reduce the risks involved with SCUBA sampling, remote controlled techniques such as STV or ROV may represent a more desirable approach. In particular ROVs, which are becoming smaller, more affordable and accessible.

We highlight the importance of technique choice, and how biological interpretations can differ depending on the intrinsic sampling biases and limitations.

Selecting an optimal sampling tool is often dependent on the question and focal species (Watson et al., 2005). For conservation studies where diversity is an important measure, BRUVs are an effective tool as they cost-effectively sample the relative abundances of targeted and non-targeted species with good statistical power across a range of depths and habitats. However, for fine scale assessments where researchers are interested in the associations of fish with a particular type of habitat or physical structure (e.g. Galaiduk et al., 2017; Harman et al., 2003; Saunders et al., 2014), transect based methods may represent a more optimal approach due to the ability to constrain the size of the sample unit. As a result it is possible to measure density, and using length data calculate biomass per unit area (or grand biomass as reported in (Friedlander and DeMartini, 2002). Due to the challenges of calculating the area of attraction of fish to a BRUV (Cappo et al., 2006; Harvey et al., 2007) it is only possible to calculate a relative estimate of biomass. Similarly, length measures are made on the MaxN of fish resulting in these estimates being conservative (Willis and Babcock, 2000). However, BRUVs are commonly applied to survey targeted fisheries species as these are attracted to the bait, and the limitations of MaxN are outweighed by the advantages of increased statistical power.

This research demonstrates that transect based methods are complementary to BRUV surveys, as the combined data set provides a more robust assessment of the characteristics of reef fish assemblages. This is because BRUVs often underestimate site-associated species. With further technical refinements, fish transect data collected by micro ROV has the potential to extend into deeper waters, both day and night, while still being comparable to data collected by DOV. A combination of BRUVs and ROV are likely to become a promising tool for future management and monitoring programs that aim to measure the density and diversity of fish assemblages in a variety of habitat types.

Chapter 3: A comparison of stereo-BRUVs and stereo-ROV techniques for sampling shallow water fish communities on and off pipelines



Preface: This chapter has been published in the Journal of Marine Environmental Research (https://doi.org/10.1016/j.marenvres.2020.105198) and has been formatted to the journal's guidelines.

3.1. Abstract

We compared and contrasted fish assemblage data sampled by baited remote underwater stereo-video systems (stereo-BRUVs) and stereo-video remotely operated vehicles (stereo-ROVs) from subsea pipelines, reef and soft sediment habitats. Stereo-BRUVs sampled greater fish diversity across all three habitats, with the stereo-ROV sampling ~46% of the same species on pipeline and reef habitats. Larger differences existed in soft sediment habitats, with stereo-BRUVs recording ~65% more species than the stereo-ROV, the majority of which were generalist carnivores. These differences were likely due to the bait used with stereo-BRUVs attracting fish from a large and unknown area. Fish may have also avoided the moving stereo-ROV, an effect possibly magnified in open soft sediment habitats. As a result of these biases, we recommend stereo-ROVs for assessing fish communities on pipelines due to their ability to capture fish *in-situ* and within a defined sampling area, but caution is needed over soft sediment habitats for ecological comparisons.

3.2. Introduction

More than 7,500 offshore oil and gas installations exist worldwide, with the majority of these expected to cease production and require decommissioning over the coming years (Macreadie et al., 2011; Parente et al., 2006). The use of oil and gas platforms as habitat by fish has been well-documented (e.g. Ajemian et al., 2015a, 2015b; Claisse et al., 2014; Friedlander et al., 2014; Gallaway et al., 2009; Love et al., 2006; Streich et al., 2017; Torquato et al., 2017), however, investigation of the habitat value of pipelines to fish assemblages has received less attention (although see; Bond et al., 2018a, 2018b, 2018c; Love and York, 2005; McLean et al., 2017). Love and York (2005) reported that fish density was nearly six to seven times greater on pipelines compared to the adjacent seafloor in the Santa Barbara Channel, Southern California. McLean et al. (2017)

and Bond et al. (2018b) both documented a high diversity and abundance of fish on pipelines, including commercially important species, on the northwest shelf of Western Australia. These three studies all used different sampling techniques; Love and York (2005): a submersible, McLean et al. (2017): industry remotely operated vehicle (ROV) footage and Bond et al. (2018b): baited remote underwater stereo-video systems (stereo-BRUVs). These differences in techniques make it challenging to draw general conclusions about the broad ecological patterns of fish assemblages associated with pipelines in comparison to natural habitats and the ecological values of pipelines.

Selecting an appropriate sampling technique to assess fish associations with subsea pipelines has its own challenges. These structures extend into deep-water, often exceeding scientific scuba depth limits (>30 m) (Ajemian et al., 2015b; Andaloro et al., 2013) eliminating the use of traditional surveys that involve divers, such as underwater visual census (UVC) or diver operated stereo-video (stereo-DOV). Extractive techniques, such as trawls and traps are commonly used in deep-water surveys (Harvey et al., 2012; Williams et al., 2001), however, they have limitations including biases in catchability in relation to size and the type of species caught (Langlois et al., 2015; Murphy and Jenkins, 2010; Patterson et al., 2012; Wells et al., 2008). They also have negative impacts on the stocks of fisheries through the removal of fish and are subject to gear loss and entanglement in areas of high structural complexity (Newman et al., 2011).

Remote underwater video systems are non-extractive and enable surveys in deep-water, but also have their own biases and limitations. The most common of these techniques is stereo-BRUVs (Mallet and Pelletier, 2014; Whitmarsh et al., 2017) and has become a standard tool for recording the relative abundance of fish across a range of habitats (Cappo et al., 2006; Harvey et al., 2012, 2018; Langlois et al., 2020). This technique provides accurate and precise fish length measurements, allowing researchers to gather important information on fish population size structure (Harvey et al., 2001b, 2002a; Harvey and Shortis, 1998). The use of bait also allows observations of targeted species including those considered 'diver-shy' (Cappo et al., 2006; Goetze et al., 2015; Gray et al., 2016) and commercially important (Harvey et al., 2012). However, the area of attraction and bait plume can vary spatially and temporally making it near-impossible to calculate the sampling area, resulting in fish abundance estimates being expressed as a relative abundance (Cappo et al., 2001; Harvey et al., 2007). Although studies have used stereo-BRUVs to quantify fish association on subsea pipelines (Bond et al., 2018b, 2018c), the major limitation of this technique is that fish densities cannot be calculated. This is important if researchers or stakeholders are interested in a biomass per unit area estimate for a section of pipeline to compare fish stocks to other areas using a standard metric (Barrett et al., 2019; Friedlander and DeMartini, 2002). The stationary nature of stereo-BRUVs and the possible attraction of fish from nearby habitats also means that investigating fine scale fish-habitat associations is challenging (Cappo et al., 2006).

An approach which overcomes some of the limitations of stereo-BRUVs whilst permitting deep-water surveys, is the use of ROVs, equipped with a stereo-video setup (stereo-ROV). This technique allows accurate fish length measurements that can be converted to density by defining a sampling area (Schramm et al., 2020a). In recent years, ROVs have emerged as an alternative approach for assessing fish assemblages, particularly on artificial structures (Ajemian et al., 2015b; Andaloro et al., 2013; Consoli et al., 2016; Trenkel et al., 2004b), including pipelines (Bond et al., 2018a; McLean et al., 2017, 2020b). ROVs are regularly used by the oil and gas industry to inspect subsea infrastructure, including pipelines, for inspection and maintenance purposes. With recent advancements in ROV system design and reductions in costs, operations are now more accessible for scientific research (Consoli et al., 2016; Sward et al., 2019). Some species-specific behavioural biases of fish towards ROVs have been documented due to the effect of the lighting, noise from electric components and the thrusters, and the size of the ROV (Ryer et al., 2009; Stoner et al., 2008). Trenkel et al. (2004b) found that North Atlantic codlings (*Lepidion eques*) avoided an ROV in close proximity, with encounters being dependent on light intensities (i.e. lower light levels resulted in more encounters). Although similar interactions have been documented for other demersal fish (Laidig et al., 2013; Lorance and Trenkel, 2006; McLean et al., 2017; Trenkel et al., 2004b), in the majority of cases the ROVs used have been of industrial size (work class or inspection class ROVs), which either emit large amounts of artificial light or make loud noises at low frequencies (Sward et al., 2019). The use of a miniature ROV system that is quieter, may overcome some of these biases and dampen fish avoidances (Schramm et al., 2020a;

Warnock et al., 2016). Mounting a stereo-video system on the ROV enhances video capabilities, allowing accurate calculations of fish length and sampling area to an accuracy likely similar to those achieved using stereo-DOVs (Harvey et al., 2001b, 2002a, 2004).

With decommissioning activities anticipated to rise, information on the ecological value of oil and gas structures as marine habitat is required to facilitate decisions on decommissioning options. These could range from completed removal (current default for most nations) to leaving *in-situ* to form artificial reefs (one proposed alternative) (Bull and Love, 2019; Fowler et al., 2014; Schroeder and Love, 2004), the choice of which will be dependent on the environmental, social and economic outcomes of each option as demonstrated by titleholders (e.g. Australia; Department of Industry, Science, Energy and Resources, 2018). The use of a standardised sampling technique, which can provide representative and useful metrics on fish assemblages (e.g. species diversity, abundance, fish-length measurements and biomass of fish) across different types of infrastructure and natural habitats will aid comparative assessments of decommissioning alternatives. The objective of this research was to compare stereo-BRUVs and stereo-ROV techniques for assessing fish communities associated with exposed pipelines and surrounding natural habitats. In particular we compare and contrast data collected on the composition of fish assemblages, the number of individuals and species, and the biomass and length-frequency of fish.

3.3. Material and methods

3.3.1. Study area

Data on fish communities were collected from subsea pipelines and surrounding reef and soft sediment habitats located around Thevenard Island, Onslow, Western Australia (21°27'25"S, 114°59'55"E) (Fig. 3.1). The pipelines form a network which connects to nine platforms in depths of 9-19 m of water and have a combined length of 132 km. At the time of installation 80% of the pipeline was trenched. Over time a portion of the remaining pipeline has been covered and uncovered by sand. Of the 132 km of pipeline ~10% (14 km) were exposed above the substrate at the time of the study. Pipe diameters range from 90 mm (gas lift lines) to 610 mm (tanker loading line). The majority of pipelines were installed in the early 1990's, with the most recent instalment in 2001. To date, marine growth on these pipelines has not been removed, but has been subject to seasonal natural disturbances (e.g. cyclones). The natural habitats in the area were dominated by sand interspersed with sessile invertebrates (mainly sponges). Reefs consisted of a mixture of coral (porites bommies and plate corals) and limestone reef covered predominantly with brown algae.

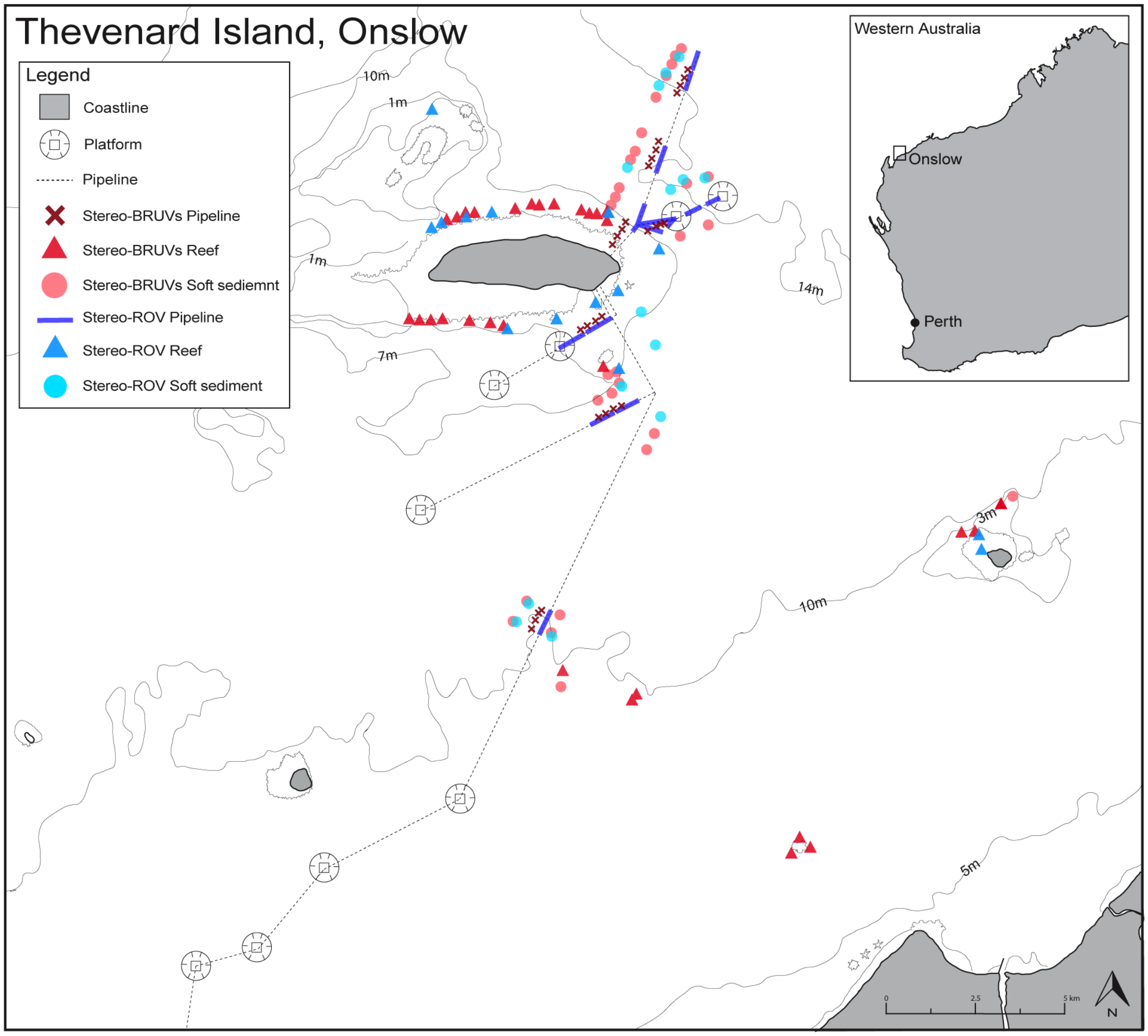


Fig. 3.1 Map of stereo-BRUVs and stereo-ROV fish surveys completed around Thevenard Island, Onslow, Western Australia.

3.3.2. Survey equipment

3.3.2.1. Stereo-video systems

Stereo-video systems followed the design and configuration presented in (Goetze et al., 2019; Harvey et al., 2010; Harvey and Shortis, 1995, 1998) and used SONY® FDRX3000 action cameras. A base separation of 475 mm between cameras was used on stereo-BRUVs, while a separation of 590 mm was used for stereo-ROV systems, both of which have shown to provide accurate and precise fish length measurements (Boutros et al., 2015). All cameras converged at a 5° angle and recorded at 60 frames per second set at medium view and in 1080p format.

3.3.2.2. Baited remote underwater stereo-video systems (stereo-BRUVs)

Eight stereo-BRUVs were baited with ~1 kg of crushed pilchards (*Sardinops sagax*) to maximise fish oil dispersal. Systems were deployed on the seafloor for a sampling period of 60 minutes.

3.3.2.3. Stereo-video remotely operated vehicle (stereo-ROV)

Stereo-ROV surveys were completed using two mini systems, SeaBotix vLBV300® and BlueROV2®. Pipeline surveys were carried out using the SeaBotix vLBV300® system (625 mm × 390 mm × 390 mm (l × w × h), ~18 kg) fitted with attached Tritech Ultra Short Baseline Positioning system (USBL) and Oculus 750D multibeam sonar to assist with system navigation and positioning. To increase fieldwork efficiency the BlueROV2® system (457 mm × 338 mm × 254 mm, ~11 kg) was used to simultaneously collect video imagery in natural habitats nearby the pipeline (reef and soft sediment). This ROV used a Seatrac X150 USBL and X010 Modem for positioning. Being similar in size and functionality it is unlikely there were any significant differences in the way that they sampled fish.

3.3.3. Data collection

3.3.3.1. Pipeline surveys

Exposed sections of pipeline were located using GIS maps and *in-situ* observations via the Seabotix vLBV300® with attached sonar, which could detect pipelines in low visibility or could track the direction of pipelines even when buried under shallow sand in order to reach the next exposed segment for surveys. Once exposed segments were located (Fig. 3.2B-D), stereo-ROV surveys proceeded. Stereo-BRUVs deployed over the same segment of pipeline were conducted on a different day after stereo-ROV surveys. We avoided concurrent sampling in the same area due to the possible influence of bait used to attract fish to the stereo-BRUVs affecting the stereo-ROV surveys. Surveys were done between 7:30 and 16:00 hrs to reduce diurnal variations in fish assemblages across the study (Bond et al., 2018a; Myers et al., 2016).

Stereo-BRUVs surveys were completed over seven segments of exposed pipe, with four replicates deployed on each segment, where possible (Fig. 3.1). Systems were dropped directly adjacent to the pipeline. This was achieved by utilising the vessel's sounder and GPS coordinates provided during stereo-ROV surveys. Systems were deployed more than 250 m apart to reduce the chance of highly mobile fish being recorded across neighbouring systems (Cappo et al., 2001). However, we acknowledge that this separation distance may be insufficient to avoid resampling of larger, transient species (e.g. sharks) (Harvey et al., 2018).

Stereo-ROV surveys were completed by flying the system approximately 1.4 ± 0.05 m from the pipeline in a parallel direction and positioned as close to the seafloor as possible (20 - 50 cm). The system was angled approximately ~25° (23.05 ± 0.77°) towards the pipeline (Fig. 3.3A and B). This flying configuration maximised the amount of fish recorded that may have been positioned underneath the pipeline. The mean speed of the stereo-ROV was 0.54 ± 0.04 m/s (1.9 km/h), with a 50 m transect taking approximately 1.54 minutes (92 s) to sample. This speed was as slow as practical while maintaining ROV control, and only slightly faster than the recommended speed for comparable diver-operated techniques (stereo-DOV ~0.33 m/s, 1.2 km/h, Goetze et al., 2019), where some speed is recommended to reduce the chance of over-estimating mobile fish (Goetze et al., 2015). Stereo-ROV footage of the pipelines was captured during live boating, where the stereo-ROV was piloted approximately 100-150 m ahead of the moving vessel. Continuous video footage was recorded until the pipeline was completely buried for an extended distance or reached a platform. In total, 14 segments of exposed pipeline were surveyed, with individual segment lengths ranging from 0.3 - 1.7 km.

For quantitative comparisons with nearby natural habitats, continuous footage was split into 50 m transects during analysis, with a 20 m separation between transects. This was calculated by using the mean speed of the stereo-ROV and sampling time for each site to determine an approximate 50 m and 20 m distance. Transects that involved >35% (>17.5 m) of buried pipeline (Fig. 3.2A) of the 50 m were excluded from the data set. On ten of the pipeline segments we sampled six transects, while the other four segments had three to five transects which was either due to the removal of buried transects along the surveyed area or the limited length of pipeline to complete six transects.

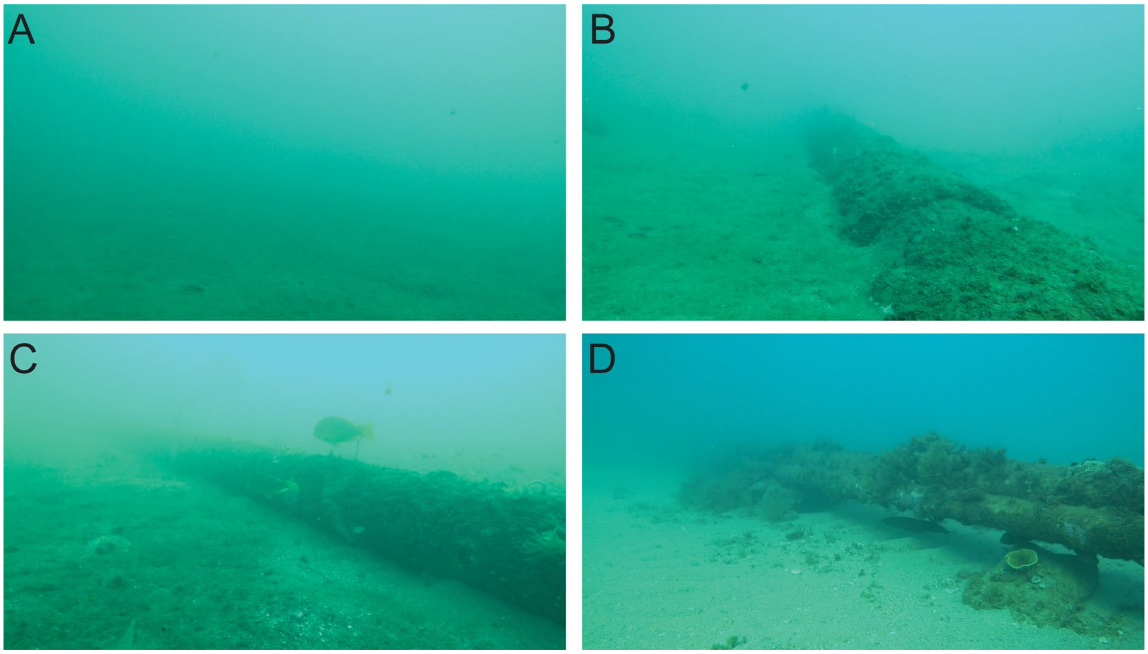


Fig. 3.2 Representative position of buried (A), more than half buried (B), resting on seafloor (C), and spanning (D) subsea pipeline.

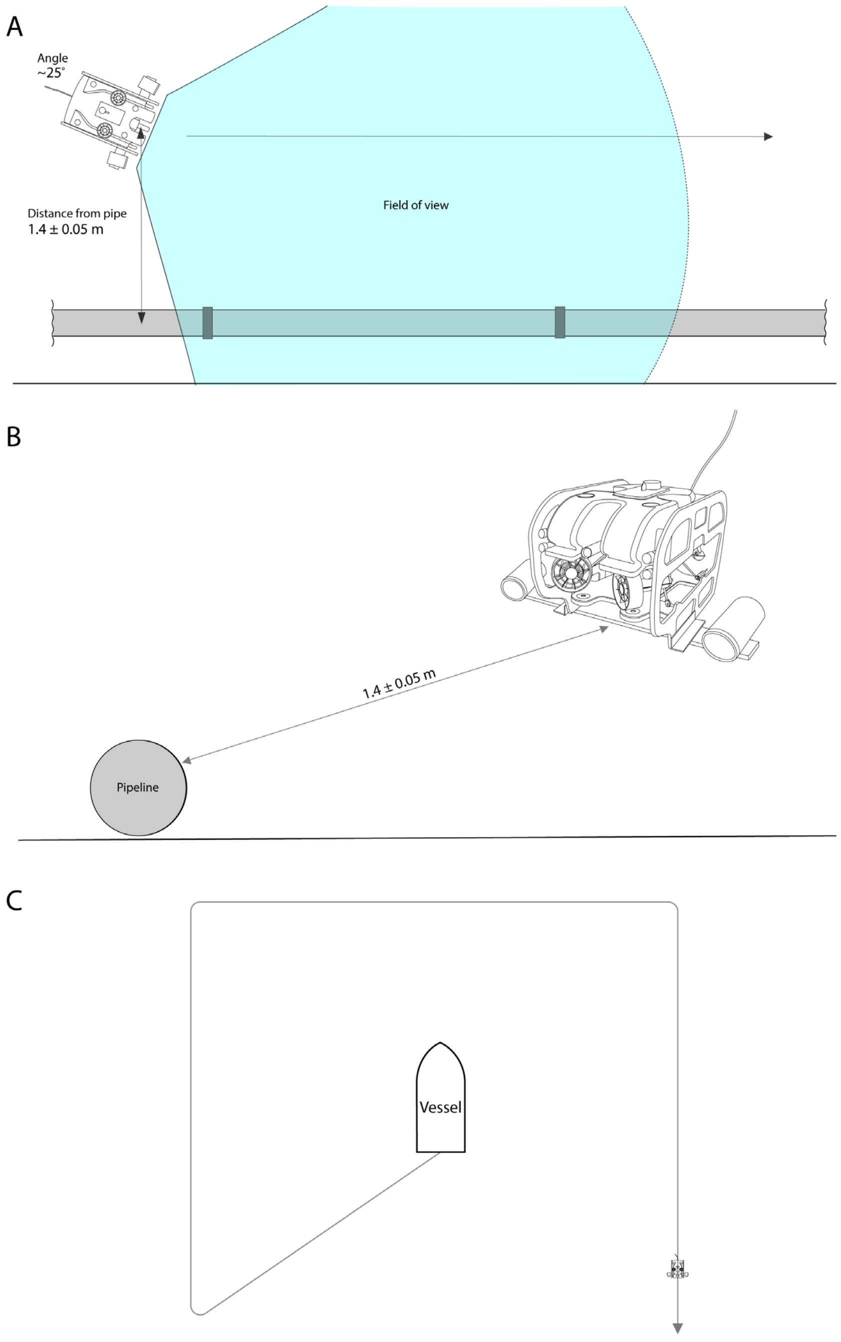


Fig. 3.3 Stereo-ROV method for surveying pipelines showing (A) plan view with approximated angle, mean distance from pipe, and field of view, and (B) front view, (C) flight path for natural reef and soft sediment habitats.

3.3.3.2. Natural habitat surveys

To compare fish association on pipelines to the surrounding environment, samples were collected >500 m away from any artificial infrastructure, over reef and soft sediment habitats. Reef was classified as hard substrate habitats covered with either coral or macroalgae (e.g. filamentous and branching). While soft sediment habitats consisted of bare sand or sand with sparse epibenthos (sponges predominantly) with underlying hard substrate not visible. Stereo-BRUVs and stereo-ROV sampling was done between 7:30 and 16:00 hrs. Stereo-BRUVs surveys involved 30 replicate deployments for reef and 28 for soft sediment habitats, with a separation distance of >250 m following the methods outlined for pipeline surveys. At natural habitat sites, the stereo-ROV was flown around the anchored vessel for ~25 mins with an average speed of ~0.5 m/s in an expanding square pattern covering new ground (Fig. 3.3C), similar to the polynomial pattern as described in Sward et al. (2019) and Gregoire and Valentine (2007). As live boating was not carried out for the natural habitat sites, the operating range of the stereo-ROV was dependent on the tether length (~150 m). When this range was reached the stereo-ROV was directed towards the vessel to avoid tension on the tether. Like the video imagery recorded on pipelines, the footage captured at each natural habitat site was split into 50 m transects with 20 m separation. In total, 14 sites were sampled for each habitat type, with 82 replicate transects recorded for reef sites and 84 replicate transects at soft sediment sites.

3.3.4. Calibration and video analysis

Stereo-video systems were calibrated using the software 'CAL' (http://www.seagis.com.au/bundle.htm), following procedures as described in Harvey and Shortis (1998) and Shortis and Harvey (1998). Calibrations were completed prior to and after fieldwork. Stereo imagery from each technique was then analysed in the programme 'EventMeasure'® (http://www.seagis.com.au/event.htm), where fish were identified to species level, counted and measured using fork length (FL; tip of snout to the fork of the caudal fin). Where fish could not be identified to species level individuals, were pooled to the next lowest taxonomic level (e.g. *Plectropomus maculatus* and *Plectropomus leopardus* pooled to a *Plectropomus* spp. identification). To achieve synchronization between cameras, left and right streams were adjusted to matching frames using a digital clapboard which showed real time, or by simply clapping in front of the system. Likewise, to obtain time and positioning information, synchronization between the on-board ROV camera and attached stereo video-system was carried out, which enabled GPS timestamps during video analyses.

For stereo-BRUVs surveys, the maximum number of individuals in one frame (MaxN) (Cappo et al., 2001, 2003; Willis and Babcock, 2000), was used as a relative assessment of abundance to ensure individuals were not recounted during the sampling period. Length measurements were recorded for individuals at the time of MaxN. Fish that exceeded a distance further than 7 m from the stereo-video system were rejected and not included in the data set, as accuracy and precision declines for fish measurements past this range (Harvey et al., 2010). The field of view of stereo-BRUVs was therefore estimated at 40.46 m^2. In the instance where a fish's head or tail was obscured by another fish, structure or other fauna and flora making it impossible to make a measurement, a 3D point was used to ensure the fish was within the field of view and included in MaxN estimates.

For stereo-ROV surveys, all individuals were counted and measured within a predetermined transect boundary (50 m × 5 m × 5 m) following the same analysis protocol as for diver operated stereo-video (Goetze et al., 2019). Fish that were identified as having left the area of a transect, which later re-entered, were only counted once. A 7 m depth of field of view with 5 m vertical and horizontal constrains was used for image analyses, with a field of view of 41.61 m^2. Fish that had a length measurement RMS precision greater than 20 mm for both techniques were rejected during the QA and QC process on the raw data.

3.3.5. Calculating biomass and habitat associations

Fork length (FL) was used to calculate biomass (g) using the equation: Weight (g) $= a \times \text{Length (cm)}^b$ (Taylor and Willis, 1998). The relevant a and b values were sourced from FishBase (Froese and Pauly, 2019). For fish that had no recorded FL due to visual obstruction, or had a poor RMS (>20 mm) we applied a mean length for fish of the same species at the level of habitat and technique to provide a more representative biomass estimate (Bach et al., 2019). Habitat preferences of fish were sourced from the ecology descriptions in FishBase (Froese and Pauly, 2019) to determine whether fish in soft sediment habitats were present due to presence of bait or naturally inhabit these areas.

3.3.6. Statistical tests

3.3.6.1. Multivariate analyses

To assess statistical differences in the fish assemblage recorded by techniques within habitats we used a two-way crossed PERMANOVA design (*Technique:* 2 levels; stereo-BRUVs, stereo-ROV, *Habitat*: 3 levels; pipeline, reef, soft sediment) based on 9999 permutations in Primer 7 with the PERMANOVA + add-on (Anderson et al., 2008). Because this statistical analysis involves a direct comparison between techniques with fundamentally different sampling units (density vs. MaxN) we used a presence/absence transformation. Analyses were done on a Jaccard resemblance matrix with dummy variable (+1) added to account for samples with zeros (Clarke et al., 2006). Statistically significant effects for *Technique, Habitat,* and *Technique* × *Habitat* were determined when $P < 0.05$. Where appropriate, a post-hoc pairwise test was done for each factor to distinguish where statistical differences existed. To illustrate the difference between techniques within each habitat a 2D nMDS (bootstrap average) ordination was produced.

3.3.6.2. Univariate analyses

To investigate statistical differences further than the presence/absence level one-way PERMANOVAs (*Habitat*: 3 levels; pipeline, reef, soft sediment) were completed for each technique with respect to the mean number of species, individuals, biomass, and abundance/relative abundance of focal species by habitat. The focal species were selected to represent different families and species that were abundant and commonly targeted by fishers in the Pilbara region (Ryan et al., 2019). Focal species were *Plectropomus* spp. (Coral trout) *Lethrinus laticaudis* (Grass emperor), and *Choerodon schoenleinii* (Blackspot tuskfish). As statistical comparisons were not completed between techniques at a univariate level, comparisons between techniques were made based on the patterns observed with habitat. Data were tested for dispersion using PERMDISP (Anderson, 2006) and were analysed using raw data based on a Euclidean resemblance matrix. Where a significant difference was found ($P < 0.05$), a pairwise test was performed, using 9999 permutations. If a low permutation value was returned (< 100) the P-value produced from a Monte Carlo bootstrapping correction P(MC) was used. Venn diagrams were constructed to illustrate the cumulative percentage of species shared between techniques within each habitat. Species accumulation curves, and pie charts showing dominant families found within each habitat sampled by each technique were also constructed.

3.3.6.3. Length Frequency

Kolmogorov-Smirnov tests ($P < 0.05$ used as a test for significance) were used to investigate differences in the length frequency distributions sampled between techniques among habitat, and for focal species (Zar, 1999). Length frequency distributions were illustrated for each technique.

3.4. Results

A total of 17,367 fish from 61 families and 286 species were observed across all habitats and techniques. Stereo-BRUVs observed 7,050 fish from 228 species, and stereo-ROV observed 10,317 fish from 194 species (Appendix; Table App. 3.1). Approximately half of the species observed in pipeline and reef habitat were detected by both techniques (43% and 49%, respectively, Fig. 3.4A, B) with only 24% of species sampled in soft sediment habitats being observed by both techniques (Fig. 3.4C). The proportion of species sampled by either stereo-BRUVs or stereo-ROV exclusively were comparable in pipeline and reef habitats, each recording between 21 and 33% of the observed species (Fig. 3.4B, A). Conversely, 68% of species in soft sediment habitat were unique to stereo-BRUVs, the majority of which were generalist carnivores (Appendix; Table App. 3.2), while 8% were unique to stereo-ROV surveys (Fig. 3.4C).

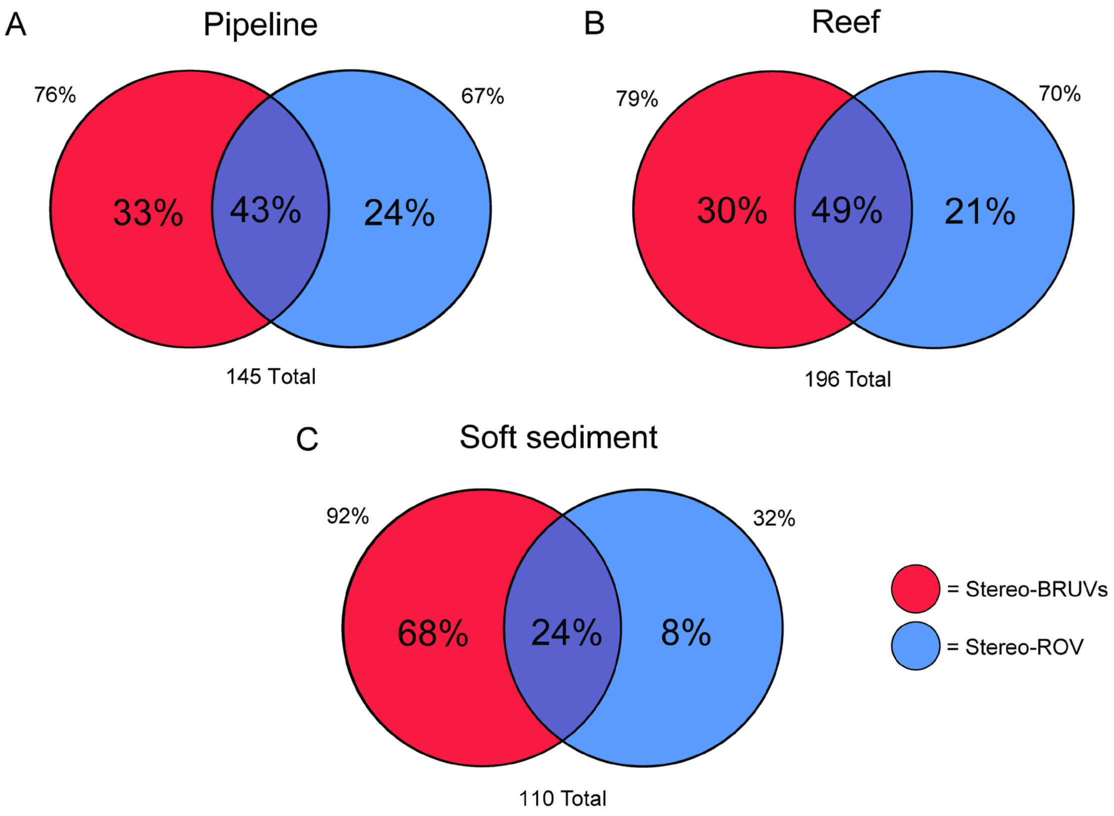


Fig. 3.4 Venn diagrams showing the cumulative percentage of species recorded by stereo-BRUVs and stereo-ROV at each habitat. Percentages are relative to the total number of species surveyed in each habitat: pipeline (A), reef (B), soft sediment (C).

3.4.1. Species richness

The accumulation of new species varied between techniques with stereo-BRUVs sampling a greater richness of fish than stereo-ROV with fewer samples required for all habitats (Fig. 3.5). The number of new species recorded per stereo-BRUVs sample continued to increase and the accumulation curve did not plateau, whereas the species accumulation curves for stereo-ROV were much flatter (Fig. 3.5). Patterns in the observed species richness among habitats differed between techniques (stereo-BRUVs, Habitat, MS = 766.35, Pseudo-F $_{(2,\ 85)}$ = 9.239, P < 0.001; Fig. 3.6A; stereo-ROV, Habitat, MS = 2,599.6, Pseudo-F $_{(2,\ 241)}$ = 80.125, P < 0.001; Fig. 3.6D). Stereo-ROV recorded ~84% more species on pipeline habitat than on soft sediment habitat (P < 0.001; Fig. 3.6D). In contrast, no difference between these habitats was detected using the stereo-BRUVs (P =

0.891; Fig. 3.6A). Both techniques sampled more species in reef habitat compared to pipeline and soft sediment habitats ($P < 0.05$; Fig. 3.6A, D).

3.4.2. Abundance

The techniques differed in the patterns of fish abundance and relative abundance observed within each habitat. No difference was found among habitats in the mean relative abundance of fish recorded during stereo-BRUVs surveys (Habitat, MS = 3,210.6 Pseudo-$F_{(2, 85)} = 0.187$, $P = 0.944$; Fig. 3.6B). By comparison, there was a difference in the mean abundance of fish recorded among habitats using stereo-ROV (Habitat, MS = 84,453, Pseudo-$F_{(2, 241)} = 18.126$, $P < 0.001$; Fig. 3.6E). Stereo-ROV recorded on average ~88% fewer fish in soft sediment compared to pipeline and reef habitats (both $P < 0.001$; Fig. 3.6E) which were similar ($P = 0.282$; Fig. 3.6E).

3.4.3. Biomass

Patterns in the biomass of fish observed by each technique varied among habitats. No differences were detected among habitats for stereo-BRUVs (Habitat, MS = 790.09, Pseudo-$F_{(2, 85)} = 1.253$, $P = 0.293$; Fig. 3.6C), however, differences were detected for stereo-ROV (Habitat, MS = 952.64, Pseudo-$F_{(2, 241)} = 10.929$, $P < 0.001$; Fig. 3.6F). A pairwise comparison for stereo-ROV indicated that biomass sampled in pipeline and reef habitats were similar ($P = 0.971$), and were both on average ~93% greater than soft sediment habitat (both $P < 0.001$; Fig. 3.6F).

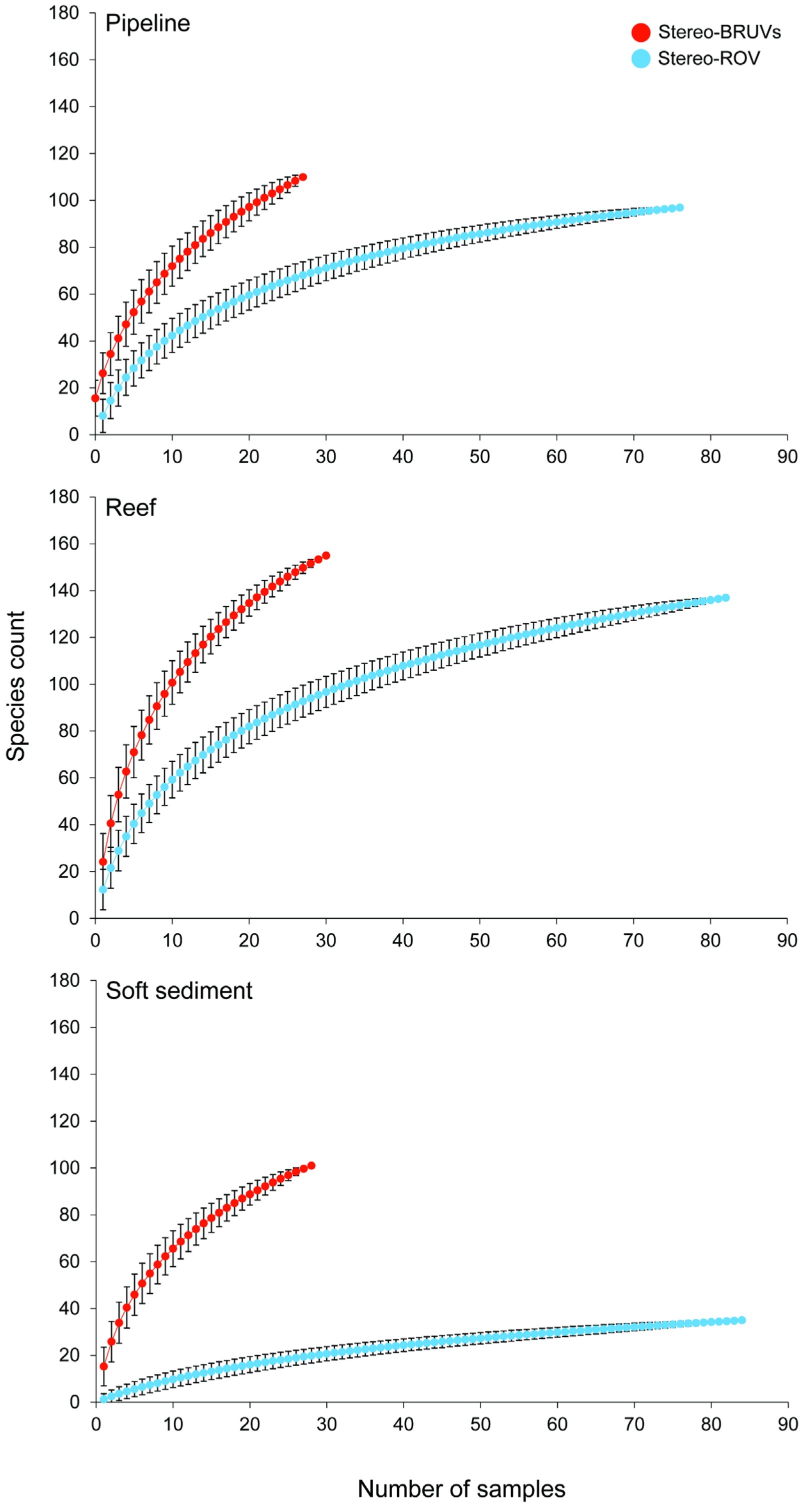


Fig. 3.5 Species accumulation curves with increasing number of samples as sampled by stereo-BRUVs and stereo-ROV across pipeline, reef and soft sediment habitat.

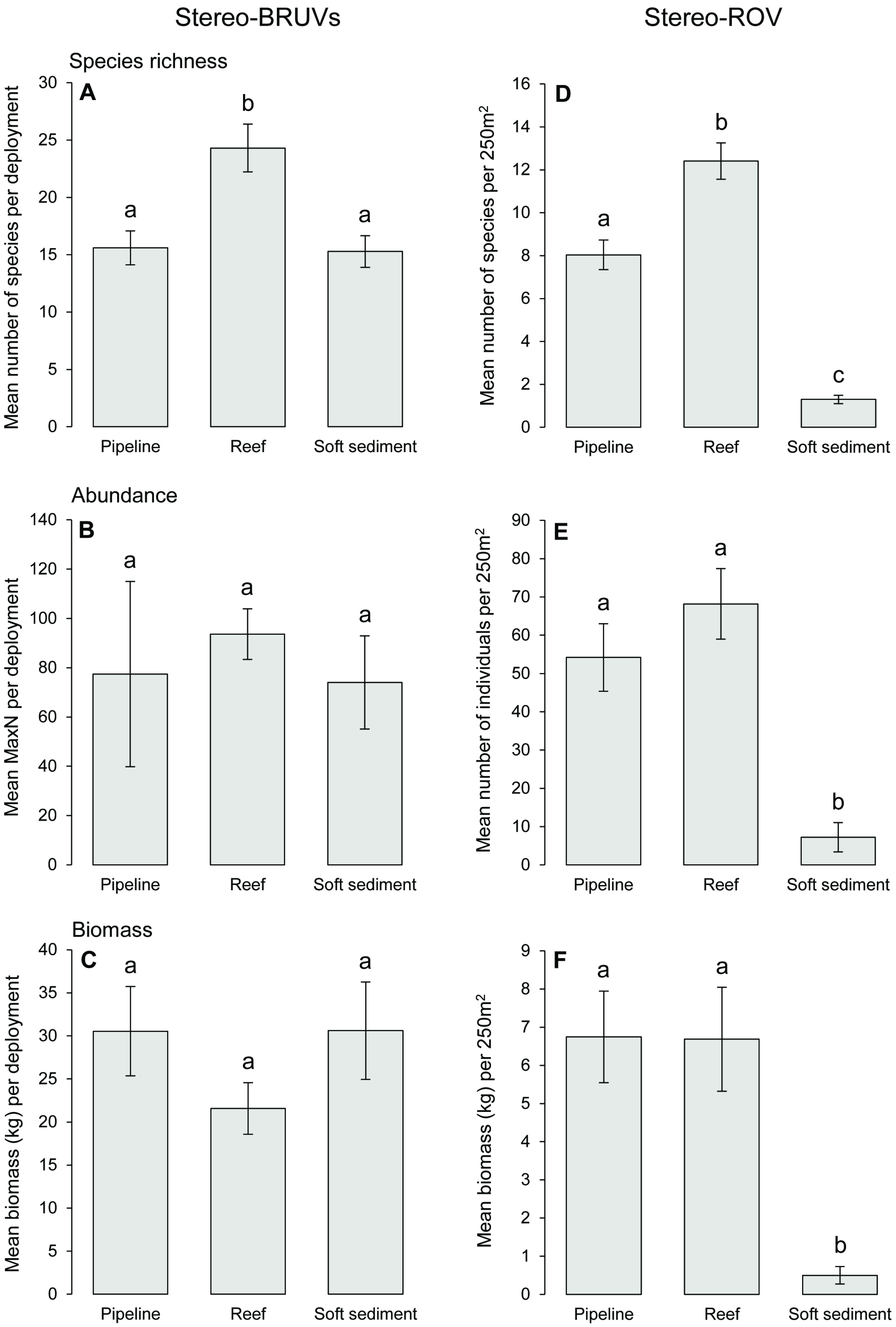


Fig. 3.6 Mean (± SE) number of species (A), relative abundance (B), and relative biomass (kg) of fish (C) per stereo-BRUVs deployment (60 mins); Mean (± SE) number of species

(D), abundance (E), and biomass (kg) of fish (F) per stereo-ROV transect (50 m x 5 m) across pipeline, reef, and soft sediment habitat. Statistically similar means are indicated by the same letter (e.g. a). Scales are not equivalent for each technique.

3.4.4. Fish assemblages

An nMDS (bootstrap average) ordination on species presence/absence data shows distinct grouping between techniques and separations between reef, pipeline, and soft sediment habitat samples within technique (Fig. 3.7). PERMANOVA detected an interaction between technique and habitat (Technique × Habitat, MS = 23,219, Pseudo-F $_{(2,\ 327)}$ = 7.923, P < 0.001; Fig. 3.7). Pairwise comparisons indicated that all combinations of techniques and habitats were different (P < 0.001). However, the stereo-ROV soft sediment habitat samples deviate away from the pattern displayed for techniques and other habitats (Fig. 3.7), this deviation drives the Technique x Habitat interaction.

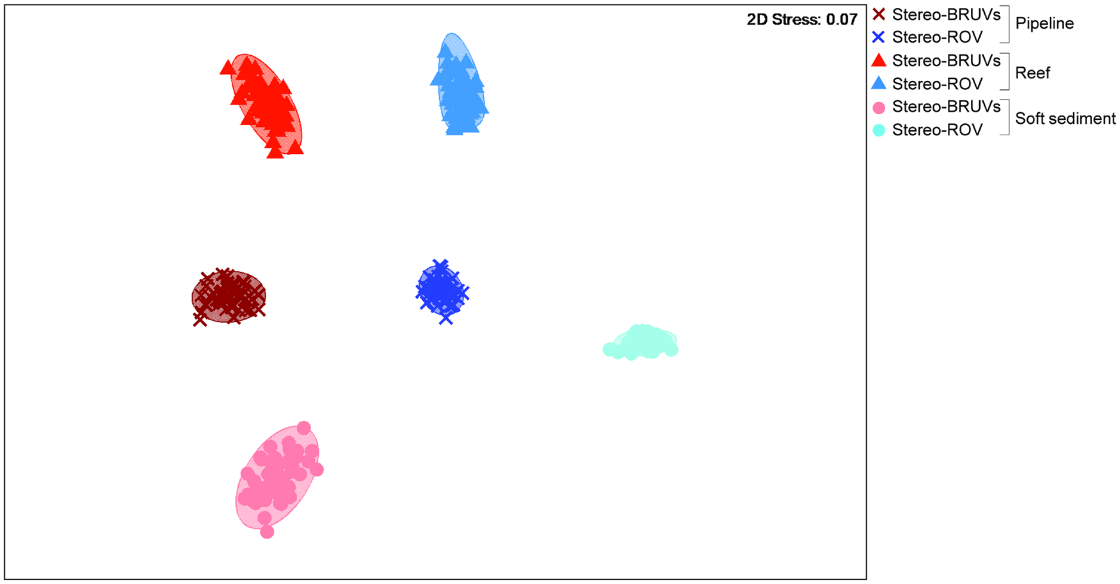


Fig. 3.7 Two dimensional bootstrap average nMDS ordination showing the differences in recorded fish assemblage by technique: stereo-BRUVs and stereo-ROV, and habitat:

pipeline, reef and soft sediment, based on a presence/absence and Jaccard similarity resemblance matrix.

Both techniques recorded similar families of fish among habitats, but in different proportions of the total fish assemblage (Fig. 3.8). For pipelines, both techniques observed a large percentage of Pomacentrids (stereo-BRUVs 47% of the total fish abundance, stereo-ROV 50%; Fig. 8A(i), B(i)). However, for stereo-BRUVs this was a reflection of detecting one large school of *Chromis fumea* (~1,000) in a single deployment, whereas the stereo-ROV detected numerous individuals and smaller schools consistently along transects. In reef habitat, stereo-BRUVs sampled a greater proportion of fish outside of the eight most dominant families with 15% of the recorded abundance belonging to "other families" (Fig. 3.8A(ii)). In comparison, 96% of fish sampled by stereo-ROV in reef habitats belonged to the eight most dominant families (Fig. 3.8B(ii)). Differences between techniques in the composition of families by abundance was more pronounced in soft sediment habitat (Fig 3.8A(iii), B(iii)). By numbers of fish, stereo-BRUVs surveys were dominated by the family Leiognathidae (26%; Fig. 3.8A(iii)), whereas stereo-ROV surveys were dominated by Lutjanidae (38%; Fig. 3.8B(iii)). However, both of which were the result of encountering a large school. With the exclusion of these schools the composition of dominant families was generally similar between techniques (Fig. 3.8A(iii), B(iii)). Stereo-ROV encountered two shark species (*Triaenodon obesus*, n = 2, *Nebrius ferrugineus*, n = 1) in reef habitat, and two ray species (*Neotrygon australiae*, n = 2, *Pateobatis jenkinsii*, n = 1) across reef and soft sediment habitat. In contrast, Stereo-BRUVs recorded 59 individual sharks, rays and wedgefish from seven families and eighteen species across all habitats.

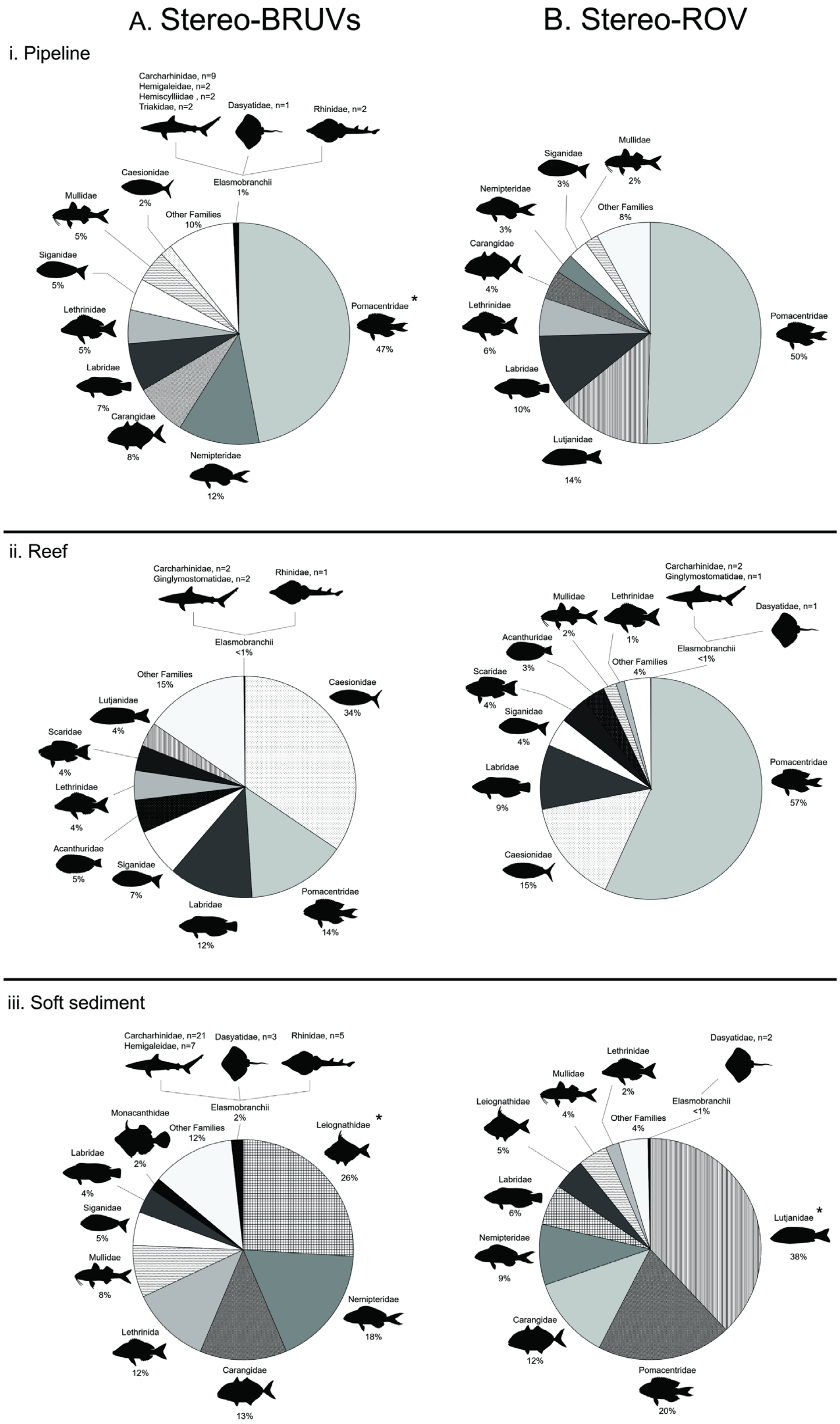


Fig. 3.8 Percentage of abundance by families sampled by stereo-BRUVs (A) and stereo-ROV (B) across habitat: pipeline (i), reef (ii), and soft sediment (iii). Proportions are

calculated from the total fish assemblage sampled by technique at each habitat, stereo-BRUVs: pipeline; n = 2,168, reef; n = 2,809, soft sediment; n = 2,073; stereo-ROV: pipeline; n = 4,118, reef; n = 5,591, soft sediment; n = 608.*Includes single schools of fish.

3.4.5. Length Frequency

Length frequencies were constructed from a total of 3,859 fish sampled by stereo-BRUVs surveys (55% of the total number recorded), and 4,106 fish lengths sampled by the stereo-ROV (40% of the total number recorded). The length distribution for stereo-BRUVs ranged from 1.4 cm (*C. fumea*) to 2.5 m (*Rhynchobatus australiae*). For stereo-ROV surveys, the smallest fish measured was 1.5 cm (*C. fumea*), with the largest being 1.4 m (*Nebrius ferrugineus*). Kolmogrov-Smirnov tests indicated that the length distribution differed between techniques for pipeline ($D = 0.190$, $P < 0.001$; Fig. 3.9A), reef ($D = 0.389$, $P < 0.001$; Fig. 3.9B) and soft sediment ($D = 0.271$, $P < 0.001$; Fig. 3.9C), with stereo-BRUVs recording on average larger fish (pipeline: mode = 200-250 mm, Fig. 3.9A; reef: mode = 100-150 mm, Fig. 3.9B; soft sediment: mode = 150-200 mm, Fig. 3.9C) than stereo-ROV (pipeline: mode = 150-200 mm, Fig. 3.9A; reef: mode = 50-100 mm, Fig. 3.9B; soft sediment: mode = 100-150 mm, Fig. 3.9C).

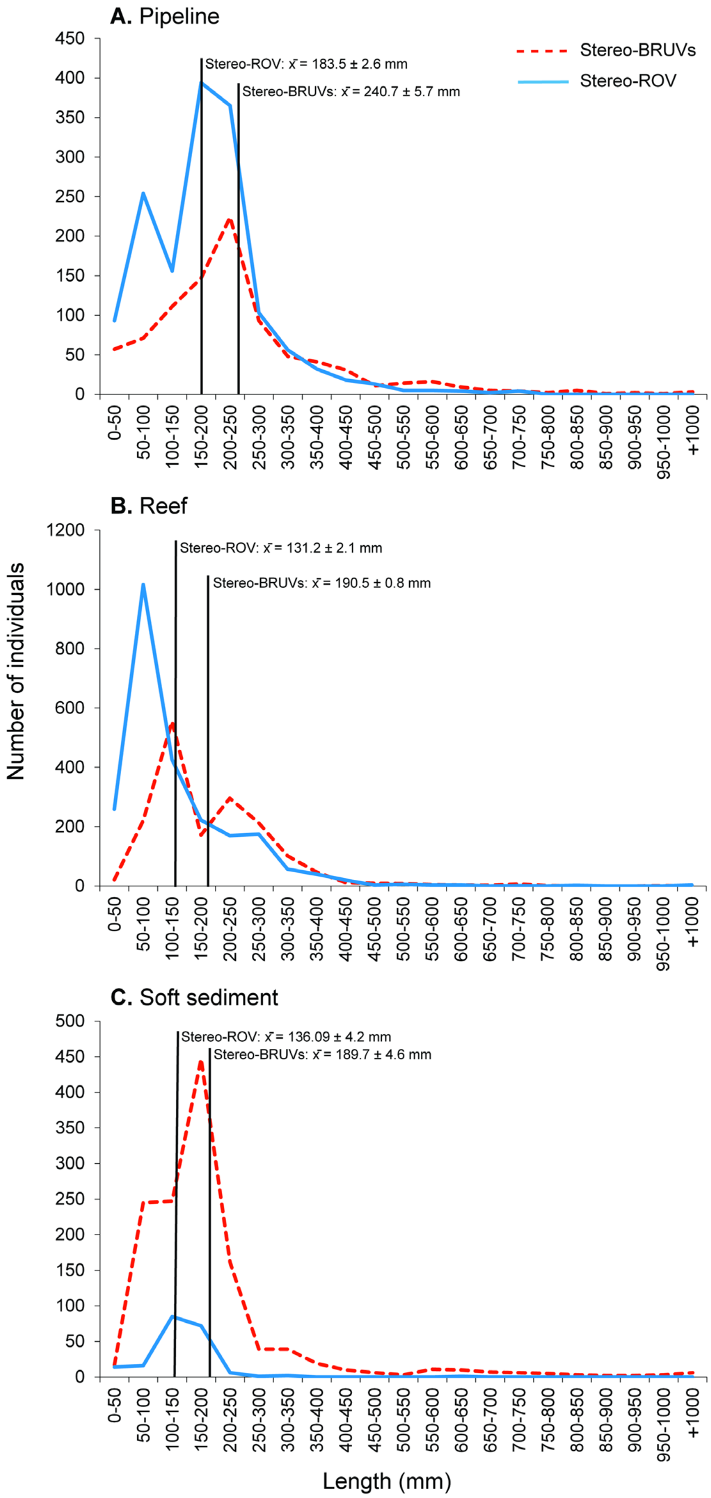


Fig. 3.9 Length frequency distribution of the observed fish assemblage, with the mean fish length (mm) for pipeline (A), reef (B), soft sediment (C) as sampled by stereo-BRUVs and stereo-ROV.

3.4.6. Focal Species

3.4.6.1. Plectropomus spp. (coral trout)

We observed a significant difference in the average number of *Plectropomus* spp. sampled among habitats using stereo-BRUVs (Habitat, MS = 2.946, Pseudo-F $_{(2, 85)}$ = 4.006, P = 0.015; Fig. 3.10A(i)) and stereo-ROV surveys (Habitat, MS = 0.420, Pseudo-F $_{(2, 241)}$ = 3.124, P = 0.032; Fig. 3.10A(ii)). Stereo-BRUVs observed more *Plectropomus* spp. on the pipeline (90% more) and on reef (85% more), than in soft sediment habitat (P(MC) = 0.012; P(MC) = 0.009, respectively Fig. 3.10A(i)). Using stereo-ROV, no *Plectropomus* spp. were recorded in soft sediment habitat, and were rarely recorded on reef and pipeline habitats (Fig. 3.10A(ii)). While the low abundance on reef was statistically similar to soft sediment (P = 0.131), the abundance on pipeline was significantly greater than soft sediment habitat (P(MC) = 0.014; Fig. 3.10A(ii)). No differences were detected among pipeline and reef habitat using either technique ($P > 0.05$; Fig. 3.10A(i, ii)).

3.4.6.2. Lethrinus laticaudis (grass emperor)

The mean abundance of *L. laticaudis* differed among habitats using stereo-BRUVs and stereo-ROV (Habitat, MS = 7.325, Pseudo-F $_{(2, 85)}$ = 4.028, P = 0.016; Habitat, MS = 4.665, Pseudo-F $_{(2, 241)}$ = 5.237, P = 0.003, respectively; Fig. 3.10B(i, ii)). Approximately 70% more *L. laticaudus* were recorded in pipeline habitat than reef habitat using stereo-BRUVs (P(MC) < 0.001; Fig. 3.10B(i)), and ~80% more fish using stereo-ROV (P(MC) = 0.027; Fig. 3.10B(ii)). A greater abundance of *L. laticaudis* were recorded on pipelines than on soft sediment habitat using stereo-ROV (P(MC) = 0.009; Fig. 3.10B(ii)), but mean abundances on these habitats were similar when sampled using stereo-BRUVs (P(MC) = 0.935; Fig. 3.10B(i)).

3.4.6.3. Choerodon schoenleinii (blackspot tuskfish)

The mean abundance of *C. schoenleinii* differed with habitats (stereo-BRUVs, Habitat, MS = 7.037, Pseudo-F $_{(2, 85)}$ = 11.999, P < 0.001; stereo-ROV, Habitat, MS = 6.816, Pseudo-F $_{(2, 241)}$ = 11.848, P < 0.001; Fig. 3.10C(i,ii)), but this pattern differed between techniques. Stereo-BRUVs detected ~91% fewer *C. schoenleinii* in soft sediment habitat than pipeline and reef habitat (both P(MC) < 0.05; Fig. 3.10C (i)), in which similar high abundances were recorded (P(MC) = 0.074). In contrast, stereo-ROV surveys detected higher abundances of *C. schoenleinii* on pipelines than in both reef and soft sediment habitats (P(MC) = 0.002, P(MC) < 0.001) which were similar (P(MC) = 0.061; Fig. 3.10C(ii)).

3.4.6.4. Length frequency for focal species

The recorded size structure between techniques was similar across all focal species (*Plectropomus* spp.: D = 0.438, P = 0.077, Fig. 3.10A(iii); *L. laticaudis* D = 0.221, P = 0.152, Fig. 3.10B(iii); *C. schoenleinii:* D = 0.109, P = 0.952, Fig. 3.10C(iii)).

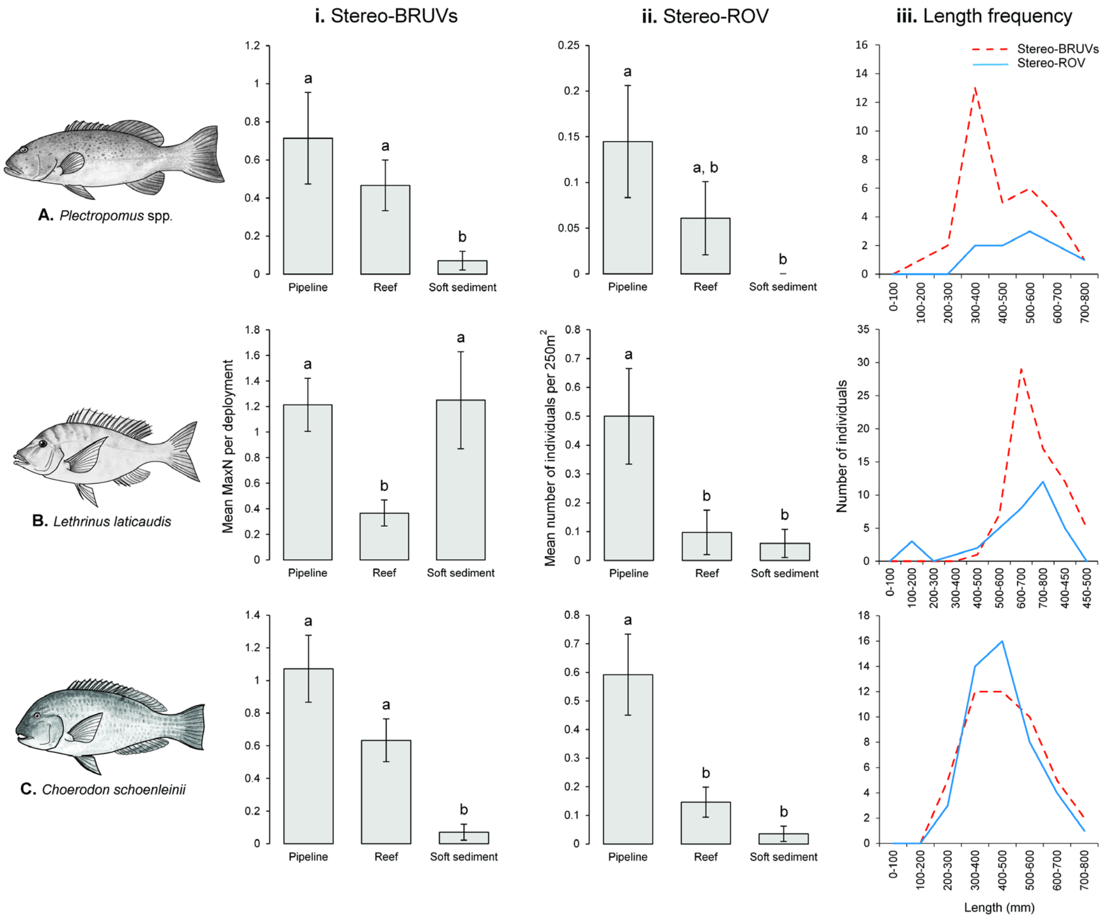


Fig. 3.10 Mean (± SE) MaxN per deployment (i), mean (± SE) number of individuals per 250 m^2 transect (ii), length frequency distribution (mm) as sampled by each stereo-BRUVs and stereo-ROV (iii) for *Plectropomus* spp. (A), *Lethrinus laticaudis* (B), and

Choerodon schoenleinii (C). Statistically similar means are indicated by the same letter (e.g. a). Scales are not equivalent for each technique or species.

3.5. Discussion

Assessing the ecological value of offshore infrastructure is a key component of comparative assessments of decommissioning alternatives. This study highlights that quantifying and defining the habitat value of subsea infrastructure to fish in comparison to natural habitats requires careful consideration of the data sources and their interpretation. For example, the sampling techniques used here detected contrasting patterns in soft sediment habitat, which require careful interpretation. Stereo-BRUVs and stereo-ROV techniques detected similar patterns in the fish assemblages for pipeline and reef habitats, while large differences were observed in soft sediment habitats. On average stereo-BRUVs sampled more individuals and species across all three habitats than stereo-ROVs, which is similar to other studies reporting that BRUVs sample a greater diversity of fish in comparison to UVC (Goetze et al., 2015; Willis and Babcock, 2000), DOV (Goetze et al., 2015; Langlois et al., 2010; Watson et al., 2010), stereo-ROV (Schramm et al., 2020a), towed video (Logan et al., 2017), traps (Harvey et al., 2012) and environmental DNA (Stat et al., 2018) (although see Colton and Swearer, 2010; Lowry et al., 2012, where UVC sampled greater species richness, particularly cryptic species, than BRUVs). However, we were surprised by the magnitude of the difference between techniques in soft sediment habitats, where stereo-BRUVs recorded ~65% more species than stereo-ROV. This leads us to question why stereo-BRUVs sampled a greater diversity (especially in soft sediment habitats) and whether sampling of fish by stereo-BRUVs reflects the true diversity of the local (i.e. area around structure or close to the BRUVs unit) or overestimates it, and similarly, whether stereo-ROV under-represents the true diversity.

One of the strengths of BRUVs is that the bait placed in front of the cameras attracts fish from the surrounding area, including out of crevices and caves. The use of bait has been shown to increase the number and diversity of fish sampled without excluding the sampling of herbivores and other trophic groups not attracted to bait (Harvey et al., 2007; Watson et al., 2005). The challenge with BRUVs is that it is extremely difficult to calculate the area from which fish are attracted to the system. This area of attraction is influenced by level of feeding activity (Dorman et al., 2012; Harvey et al., 2007; Watson et al., 2005, 2010) and the dispersion of the bait, which is then dependent on soak time, current velocity and direction, and the benthic topography of the area (Cappo et al., 2001, 2003, 2006). Even small scale site-specific variation in currents can alter the observed fish assemblage amongst BRUVs deployments; potentially confounding interpretations (Taylor et al., 2013). However, the number of fish and diversity of species sampled using BRUVs is also a function of the size and mobility of fish species and individuals within a species, and the speed at which they can swim towards a BRUV system once they detect the scent of the bait (Cappo et al., 2003). As a result, the area of attraction is not equal for all species, or even within different sizes of a species, making it extremely challenging or impossible to calculate a density estimate for fish sampled by BRUVs. As a consequence, data generated from BRUVs are best used for broad scale spatial and temporal comparisons of the relative abundance or composition of fish assemblages (but see Rizzari et al., 2014 who calculated the density of reef sharks sampled by BRUVs). As soft sediment habitats are relatively sparse with low typographic complexity the area of the bait plume is likely to be larger than on reefs with the added possibility of attracting species from a larger area, which may include other nearby habitats. Therefore, stereo-BRUVs samples are likely to inflate the number of fish observed in comparison to the true density or abundance per unit area that fish naturally range over. Although the majority of fish sampled by stereo-BRUVs in soft sediment habitats were normally associated with bare sand, sponge gardens or were species with a large home range, ~10% of the species observed are generally considered coral or rocky reef associated (e.g. *Plectropomus* spp. and *Gymnothorax* spp.) (Appendix; Table App. 3.2) suggesting they were attracted out onto the soft sediment from nearby reefs and outcrops. This is problematic when researchers are interested in sampling a particular structure with limited spatial coverage, such as pipelines or other fine-scale habitats features (e.g. corals and sponges).

A byproduct of the use of bait is that individual BRUV deployments must be spatially separated to maintain independence of sampling units (Cappo et al., 2001). Here we used a separation of at least 250 m, but this spacing restricted the number of replicates that could be placed within the limited area of habitat available. This is especially difficult on pipeline, which are linear and in our study only some sections were exposed above the substrate. The species accumulation curves for stereo-BRUVs illustrate that the curves had not reached species saturation in any habitat (similar to Cappo et al., 2001, 2004). Therefore, additional sampling would likely have continued to increase the diversity of species measured. In contrast, the sampling was closer to species saturation with stereo-ROV. In this study, an optimal sampling design for BRUVs would involve a greater number of samples, but this was not physically possible to achieve. Increased sampling using BRUVs would have increased the number of species recorded, and likely lead to a greater difference in the measured diversity between the techniques.

Avoidance behavioural responses of fish to the stereo-ROV may explain why fewer species were observed compared to stereo-BRUVs, particularly over soft sediment habitats. Fish avoidance behaviours towards ROVs have been documented (e.g. Ryer et al., 2009; Stoner et al., 2008; Trenkel et al., 2004b) with changes in response towards light and size of the ROVs. The behavioural responses of fish towards an ROV is likely to vary and be species-specific (Laidig et al., 2013) and also influenced by the size of the ROV. For example, meso-predators, such as *Lethrinus bohar* and *Lethrinus nebulosus*, modify their behaviour further away from shelter, becoming more cautious in response to predation risks (Lester et al., 2020). This may explain why we sampled *L. laticaudis* in pipeline and reef habitats, but very few in soft sediment. The presence of a physical habitat structure likely provided a refuge within which they could still be observed by the stereo-ROV. In the absence of structure, avoidance or flight behaviours are likely to be magnified, and it might be that the stereo-ROV sampling may not detect the presence of the fish. Similarly, it is possible the stereo-ROV under-sampled fish in more complex habitat such as natural reefs which have greater structural complexity and interstitial spaces. Fish fleeing or sheltering from the stereo-ROV in structurally complex habitats would likely be out of view and not recorded. Fish that were sampled uniquely in stereo-ROV surveys on soft sediment were mainly localised, site-attached species, such as *Hoplolatilus cuniculus*, *Gobiidae* spp., and *Ptereleotris microlepis*, suggesting that stereo-ROV may have been better at capturing the presence of these sand dwelling site attached species. Further work is needed to understand the behavioural interactions between stereo-

ROVs and fish assemblages in different habitat types, especially as few researchers undertake transect based visual observations of fish in soft sediments (but see Spencer et al., 2005) and so researchers are probably largely unaware of the habitat/assemblage differences. One avenue to extend our 'field of view' and to detect fish before they flee might be to explore the use of active acoustics (imaging sonar). By equipping stereo-ROV systems with imaging sonar, fish can be detected up to a 30 m radius from the system (Harvey et al., 2018), which might allow researchers to identify distances over which fish are exhibiting avoidance behaviour prior to the ROV flying past. Further advancements in reducing ROV thruster and tether vibration noise could also mitigate avoidance behavioural responses. Fish avoidance towards noise produced by bubbles from standard open circuit diving systems were moderated by the use of quieter semi-closed rebreather systems (Lindfield et al., 2014; Stamoulis et al., 2020). A comparison between untethered AUV (autonomous underwater vehicle) and ROV could separate the effects from thruster noise and tether vibrations. Another consideration could be the vessel noise associated with ROV operations (De Robertis and Handegard, 2012). We used both anchored and live boating for surveys and future research should control for the effect of engine noise of an operating vessel. Moderating the noise produced from ROVs may dampen fish avoidance interactions, particularly in open habitats where it is likely magnified, allowing samples to reflect the true diversity and abundance of fishes.

Sharks and rays were only recorded in any great abundance using stereo-BRUVs. Similar disparities between elasmobranch observations have been reported on pipelines, with fewer sharks observed using ROVs (McLean et al., 2017) in comparison to stereo-BRUVs (Bond et al., 2018b). Stereo-BRUVs is thought to be more suitable for sampling sharks than transect based methods due to avoidance behaviour and bait attraction (Harvey et al., 2018). Another explanation may be due to the longer sampling time of a stereo-BRUV system being left on the seafloor at one location (1 hr) in comparison to the relatively quick ROV transect (~1.5 min to sample a 50 m transect). The mean arrival time of sharks and rays at a BRUV system was 36.82 (± 1.91 min) with the number of species and relative abundance increasing over time.

Despite measuring similar numbers of fish (stereo-BRUVs 3,859 vs 4,106 individuals for stereo-ROV) the length distribution of fish varied between techniques, with stereo-BRUVs tending to sample larger fish than those sampled by stereo-ROV. This result is probably due to stereo-BRUVs recording larger species (e.g. *Carcharhinus plumbeus*, *G. cuvier,* and *R. australiae*), while stereo-ROV recorded more smaller-bodied species (e.g. *C. fumea*, *Neopomacentrus azysron*, *Neopomacentrus aktites*), rather than either technique targeting different sized individuals of the same species. This was evident with *Plectropomus* spp*.,* *L. laticaudis* and *C. schoenleinii* having comparable length distributions between techniques. The use of BRUVs may introduce biases into the observed size structure of a fish assemblage (Langlois et al., 2015). For example, the use of MaxN in BRUVs surveys can over or under-estimate the true length distribution of a fish species, whereby smaller individuals are displaced by larger individuals, or vice versa depending on the dominant size at time of MaxN (Coghlan et al., 2017). This may be the result of larger individuals being displaced by juvenile aggregations (Andradi-Brown et al., 2016; Harvey et al., 2012), competitive exclusion where smaller fish do not enter the field of view due the presence of larger predatory fish (Cappo et al., 2004, 2007; Harvey et al., 2007), or small individuals being displaced by larger ones over time (Cappo et al., 2009). Because fish lengths are measured at time of MaxN*,* length data can be lost on individuals displaced, or those seen at other stages of the video and as a result potentially reflect a biased size distribution. Transect surveys may avoid this issue as they measure all individuals encountered within the transect, with the exception of fishes that might flee from the ROV. It is important that representative length frequency distributions are obtained or that the biases in the frequency distributions between methods are well understood. In the case of subsea pipelines, establishing information about the size structure of species of fish could provide insights into the life stages utilising these structures in comparison to natural habitats.

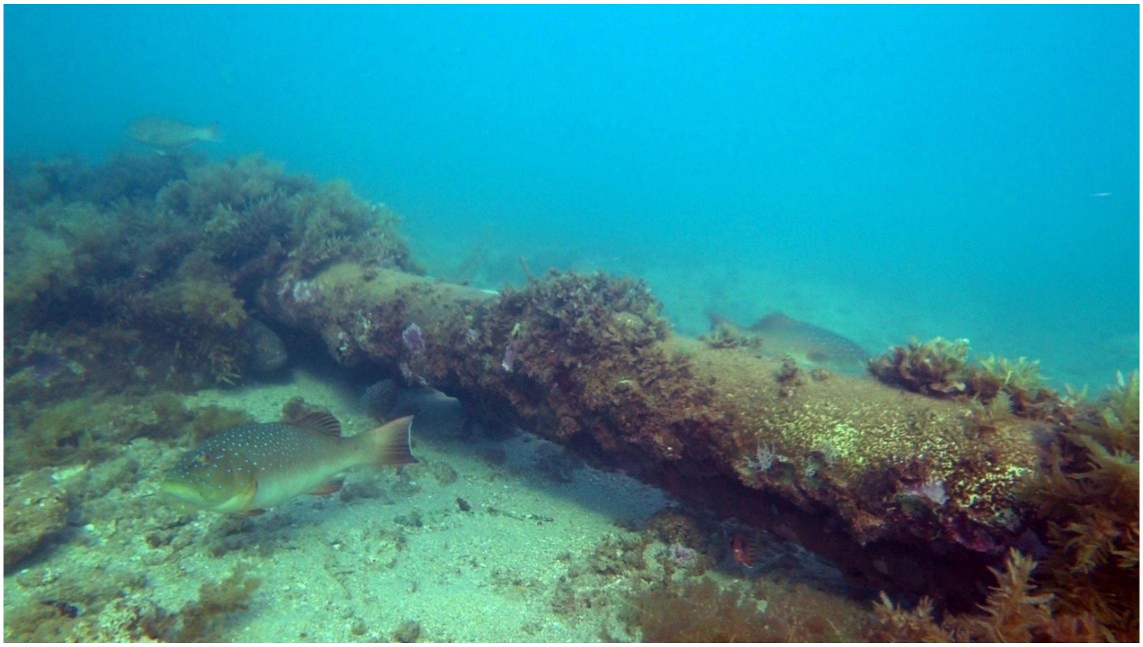
The biomass of fish is related and another important measurement for assessing the potential fishery value of oil and gas structures. While stereo-BRUVs recorded a greater relative biomass of fish across all three habitats in comparison to stereo-ROV, largely because they sampled larger, more mobile species (e.g. sharks) that were attracted to the bait, the unknown sampling area means estimates are restricted to a relative measurement only. This makes it impossible to constrain the sampling area to the footprint of a pipeline and scale up estimates to represent a particular length of pipeline or an entire network. By comparison the lack of bait and use of transects in stereo-ROV surveys makes it possible to calculate biomass to a unit per area providing contextual data of fish associated with particular habitats (Friedlander and DeMartini, 2002) or locations (Barrett et al., 2019; Friedlander and DeMartini, 2002; Salinas de León et al., 2016). For commercially important species, biomass estimates sourced from unit per area measurements can be converted into catch values ($/kg), allowing researchers to quantitatively compare the fishery value of oil and gas structures to natural habitats.

Identifying key species of fisheries value (commercially and recreationally) associated with pipelines is also important for stakeholder engagement when considering decommissioning alternatives. A greater diversity of lutjanidae (snapper) was recorded by the stereo-ROV in comparison to stereo-BRUVs along the pipeline. This is likely a result of ROVs ability to sample fish *in-situ*. High abundances of lutjanidae have been reported in undercut sections of pipeline with minimal exposure (i.e. pipeline spanning, < 0.5 m) (McLean et al., 2017). As ROVs have the ability to fly along the pipeline and in close proximity, it may be that the field of view more frequently captures fish such as snapper, which utilise these features for refuge, in comparison to stationary stereo-BRUVs. Greater abundances of *Epinephelus* species (groupers) were also recorded by the stereo-ROV and were frequently observed utilising similar undercut and span habitat features along the pipeline. If these species are not mobile during the time of sampling they are likely to be under-represented using stereo-BRUVs, as systems may not be deployed directly adjacent or near to sections of pipeline that have spanning, or appropriate physical structure. The ability to sample spatial heterogeneity is necessary in order to capture habitat features, such as spans and other features (i.e. mattress and rock dumps) along the pipeline, that may serve as favourable habitat for particular species, including those that are important to commercial and recreational fisheries.

The abundance and biomass measures reported using the stereo-ROV in this study are for transects of known dimensions (50 m x 5 m, 250 m^2), whereas it is not possible to define the unit area sampled with stereo-BRUVs (Harvey et al., 2018). To contextualise the ecological and fishery value of oil and gas infrastructure, and indeed natural habitats, a measure of sampling unit is required, such as the surface area or volume of water sampled (Barrett et al., 2019; Friedlander and DeMartini, 2002), which cannot be derived from the relative abundance and biomass measures derived from stereo-BRUVs. With estimates of sampling unit, value metrics can be scaled to represent a whole structure, such as platforms, wellheads, a length of pipeline, mooring blocks or other assets which have become fish habitat, and compared among structures and to natural habitats. Being able to quantify differences between sampling techniques, their strengths, limitations and biases, will only strengthen comparative assessments inferences and ultimately the ability to accurately describe fish patterns to the range of stakeholders.

To facilitate decisions on decommissioning alternatives, an appropriate sampling technique that can be used to accurately assess fish communities on and off oil and gas infrastructure is essential. Previous research on subsea pipelines have either used submersibles (Love and York, 2005), industry ROV footage (Bond et al., 2018a; McLean et al., 2017, 2020b), or stereo-BRUVs (Bond et al., 2018b, 2018c) to evaluate fish associations. We highlight the importance of technique choice, and how intrinsic biases and limitations of a technique can result in different biological interpretations, particularly over different habitats. Stereo-BRUVs are an effective tool for sampling high diversity of fishes over a range of depths and habitats, including targeted and highly mobile species. However, due to their stationary nature and use of bait to attract fish, fine scale fish-habitat associations cannot be explored in great detail. Difficulties in calculating the extent of bait dispersion also limits fish estimates to a relative measure, rather than a unit per area. As subsea pipelines are narrow linear structures it is important that the sampling tool selected has the ability to constrain its sampling area to the structure. It is also important that the sampling technique can record position specific fish-habitat associations that are necessary to explain the key drivers of fish occupancy on such structures (e.g. pipeline position and epifaunal growth). For this reason, stereo-ROV may be more useful on pipelines, as it provides position-specific contextual information, and has a defined sampling area. In addition, the magnified flight response of fish likely experienced in open soft sediment habitat due to ROV presence was not observed on pipelines. However, consideration is needed for studies that aim to compare pipeline habitats with nearby natural habitats as stereo-ROV techniques are likely to underestimate fish communities in open environments such as bare sand, due to potential fish avoidance behaviours.

Chapter 4: Fish associations with shallow water subsea pipelines compared to surrounding reef and soft sediment habitats



Preface: This chapter has been accepted for publication in Scientific Reports and has been formatted to the journal's guidelines.

4.1. Abstract

Offshore decommissioning activities are expected to increase as oil and gas subsea infrastructure becomes obsolete. Decisions on decommissioning alternatives will benefit from quantifying and understanding the marine communities associated with these structures. As a case study, fish assemblages associated with an inshore network of subsea pipelines located on the North West shelf of Western Australia were compared to those in surrounding natural reef and soft sediment habitats using remotely operated vehicles fitted with a stereo-video system (stereo-ROVs). The number of species, the abundance, biomass, feeding guild composition and the economic value of fishes were compared among habitats. The community composition of fish associated with pipelines was distinct from those associated with natural habitats, and was characterised by a greater abundance and/or biomass of fish from higher trophic levels (e.g. piscivores, generalist carnivores and invertivores), including many species considered to be of value to commercial and recreational fishers. Biomass of fish on pipelines was, on average, 20 times greater than soft sediments, and was similar to natural reefs. However, the biomass of species considered important to fisheries recorded on the pipelines was, on average 3.5 times greater than reef and 44.5 times greater than soft sediment habitats. This study demonstrates that fish assemblages on the pipeline infrastructure exhibit high ecological and socioeconomic values.

4.2. Introduction

There are approximately 7,500 oil and gas structures in the marine environment globally, many of which will require decommissioning in the near future as they reach the end of their production life (Macreadie et al., 2011; Parente et al., 2006). In most countries, current decommissioning policies require the complete removal of infrastructure. This is in alignment with international obligations (i.e. the United Nations Convention on the Law of the Sea (UNCLOS) and the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention)) (Fowler et al., 2014;

Hamzah, 2003). In Australia, the default policy position on decommissioning is complete removal (Chandler et al., 2017). However, with growing evidence that oil and gas structures have the potential to function as artificial reefs (e.g. Claisse et al., 2014; Fowler and Booth, 2012; Friedlander et al., 2014; Gallaway et al., 2009; Love et al., 2006; McLean et al., 2020a; Schramm et al., 2020b), there is a compelling environmental case for consideration of *in-situ* decommissioning alternatives. These alternatives include leaving the infrastructure in place, partial removal, toppling onto the seafloor or relocating to a designated reefing site (Bull and Love, 2019; Fowler et al., 2014; Schroeder and Love, 2004). Alternative decommissioning strategies could be supported if there is information demonstrating equal or better environmental and safety outcomes in comparison to complete removal (Department of Industry, Science, Energy and Resources, 2018).

There is demonstrated evidence that oil and gas infrastructure can support a high abundance and diversity of sessile invertebrates and fish, including species that are considered commercially and recreationally important (Fowler et al., 2014; Love et al., 2006), and/or are of high conservation value (Bell and Smith, 1999; Friedlander et al., 2014). The option of leaving these structures in the water may potentially benefit fisheries through increased catch (APPEA, 2017). They may also provide opportunities for diver-based tourism where structures are readily accessible (Stolk et al., 2007). Some of these structures have also been documented as having potential conservation benefits (Friedlander et al., 2014; Moore et al., 2016; Scarborough-Bull et al., 2008). However, there are concerns that the aggregation of fish may also lead to overfishing and depletion of fish stocks, especially if attraction is driving these associations (Bohnsack, 1989; Cripps and Aabel, 2002; Grossman et al., 1997; Matthews, 1985), although see (Fowler and Booth, 2012). Additionally, it has been suggested that offshore infrastructure may facilitate the propagation of invasive species by providing a mechanism for connecting habitat mosaics (Page et al., 2006; Pajuelo et al., 2016). Other considerations often associated with *in-situ* decommissioning alternatives include potential leaching of contaminants, snagging risk for trawl fisheries and shipping navigational hazards (Rouse et al., 2020; van Elden et al., 2019). From an ecological perspective, it is important that rigorous scientific data is collected to characterise fish assemblages associated with these structures in order to weigh the environmental, social and economic value of retaining these habitats against other benefits and risks.

Subsea pipelines are an integral component of oil and gas operations and form extensive networks on the seafloor. Despite their prevalence in our oceans there are few environmental studies that assess the ecological role of subsea pipelines as habitat (although see (Bond et al., 2018a, 2018b; 2018c; Love and York, 2005; McLean et al., 2017, 2020b), with the majority of literature focused on oil and gas platforms (Bull and Love, 2019). Research is now beginning to demonstrate the potential role subsea pipelines may serve in the marine environment. For example, a colony of Australian fur seals (*Arctocephalus pusillus doriferus*) in the Bass Strait, south-eastern Australia, has been documented using subsea pipelines to search for prey (Arnould et al., 2015). Similarly, fish have been documented utilising subsea pipelines as habitat, with Love and York (2005) reporting that fish densities were six to seven times greater on pipelines compared to the adjacent seafloor in the Santa Barbara Channel, Southern California. Studies specific to north Western Australia (Bond et al., 2018b; McLean et al., 2017, 2020b) have also documented a high diversity and abundance of fish on pipelines, including species that are considered important to fisheries (e.g. lutjanids (snappers) and epinephelids (groupers)). Higher distributions of fish were observed near spanning pipelines (i.e. unsupported pipe where seabed sediment has been removed by water flow scouring), suggesting that these structures may be favourable places for refuge and access to food (e.g. ambush behaviours) for some species (Bond et al., 2018b; McLean et al., 2017, 2020b).

Studies that have assessed fish associations with subsea pipelines have either used existing industry remotely operated vehicle (ROV) video footage (Bond et al., 2018a; McLean et al., 2017, 2020b), small submersibles (Love and York, 2005), or baited remote underwater stereo-video systems (stereo-BRUVs) (Bond et al., 2018b, 2018c) as a means of sampling. The use of a mini-ROV fitted with a stereo-video system (stereo-ROV) may be a more appropriate sampling approach for assessing fish associations on pipelines, as the stereo camera setup provides per unit area measurements of fish and accurate length data for biomass estimates *in-situ* (Schramm et al., 2020b). Furthermore, the majority of studies have assessed pipelines in relatively deep waters, ranging from 56 – 230 m (although see Bond 2018c, where fish surveys started at 15 m depth), and where surrounding habitat consisted predominantly of soft sediment. There is little information describing how fish communities interact with subsea pipelines at shallow depths (< 30 m), particularly where surrounding habitats are complex, such as coral reefs.

The North West Shelf of Western Australia encompasses the Northern Carnarvon and Roebuck Basins, where several thousand kilometres of subsea pipeline exist across multiple oil and gas projects (DMP, 2020). These structures are predominantly situated over soft sediment habitat and sparse, sponge garden communities. However, at shallower depths (< 30 m) they also lie within, or adjacent to complex reef systems, which include communities of hard and soft corals. Studies in this region have shown the inshore fish assemblage to be highly diverse and include species which are endemic (e.g. *Lethrinus punctulatus*), protected (e.g. *Epinephelus lanceolatus*), and of importance to commercial and recreational fishers (McLean et al., 2016; Travers et al., 2006, 2012, 2018).

With decommissioning activities expected to increase in the future, understanding the ecological roles of oil and gas structures, including subsea pipelines, will contribute valuable information to the decision-making process on decommissioning alternatives. This study aims to compare fish assemblages on exposed shallow-water subsea pipelines to those observed in nearby natural reef and soft sediment habitats using stereo-ROVs. We assessed the number of species, abundance, biomass, feeding guild composition and the potential economic value of fish communities associated with pipelines near Thevenard Island, Western Australia. We also surveyed nearby natural reef and soft sediment habitats in order to contextualise the value of pipelines as habitat for shallow-water fish communities.

4.3. Methods

4.3.1. Study area

Surveys were carried out in September 2018 on a network of subsea pipelines located near Thevenard Island, Western Australia, ranging in water depths of 10.6 - 20 m (Fig. 4.1). The majority of the pipelines were installed between 1989 – 1994, with the most recent installation in 2001 prior to cessation of operations in 2014. The network of pipelines has a combined length of 132 km in depths ranging from 0 - 20 m and connects nine platforms (three tripods and six monopods) to onshore facilities. During installation, ~80% of the pipelines were trenched and backfilled. At the time of the study approximately 14 km (~10%) of pipelines were exposed above the substrate and ranged from more than half-buried, more than half-exposed to spanning above the seafloor (Fig. 4.2B-D). The proportion of pipeline buried (Fig. 4.2A) and how exposed above the seabed unburied sections are is likely to change over time (especially in high current areas) due to sediment transport and scouring processes in the shallow-water environment. For the purposes of this study, other structures associated with the pipeline such as concrete mattresses and tie downs were considered part of the pipeline. Fish seen on these structures were included in counts from the pipelines (Fig. 4.2E-F). The outer diameter of the pipelines ranged between approximately 89 - 720 mm. Marine growth had not been cleaned from the pipelines since installation, but had been subject to natural disturbances such as cyclones.

We also surveyed natural reefs and soft sediment habitats in the surrounding area in water depths of 3.7 - 18.5 m at the same time to provide ecological context to the data obtained from the pipelines. GIS maps of the region with habitat overlays were used as guides for the selection of reef and soft sediment sites (Chevron, 2016). Reef habitat consisted of hard substrate with coral cover and/or macro algae (Fig. 4.3A and B), while soft sediment consisted of bare sand (Fig. 4.3C) or sand with patchy epibenthos (e.g. sponges and gorgonians) with underlying hard substrate not visible (Fig. 4.3D). The inshore waters where the focal pipelines are located are closed to commercial fisheries, expect for a small-scale trawl fishery that targets banana prawns (*Fenneropenaeus indicus*) and a pelagic fishery for Spanish mackerel (*Scomberomorus* spp.) (Gaughan et al., 2019). This region is subject to recreational fishing activities, however due to low human population sizes along the Pilbara coast, fishing pressures are minimal (Ryan et al., 2019).

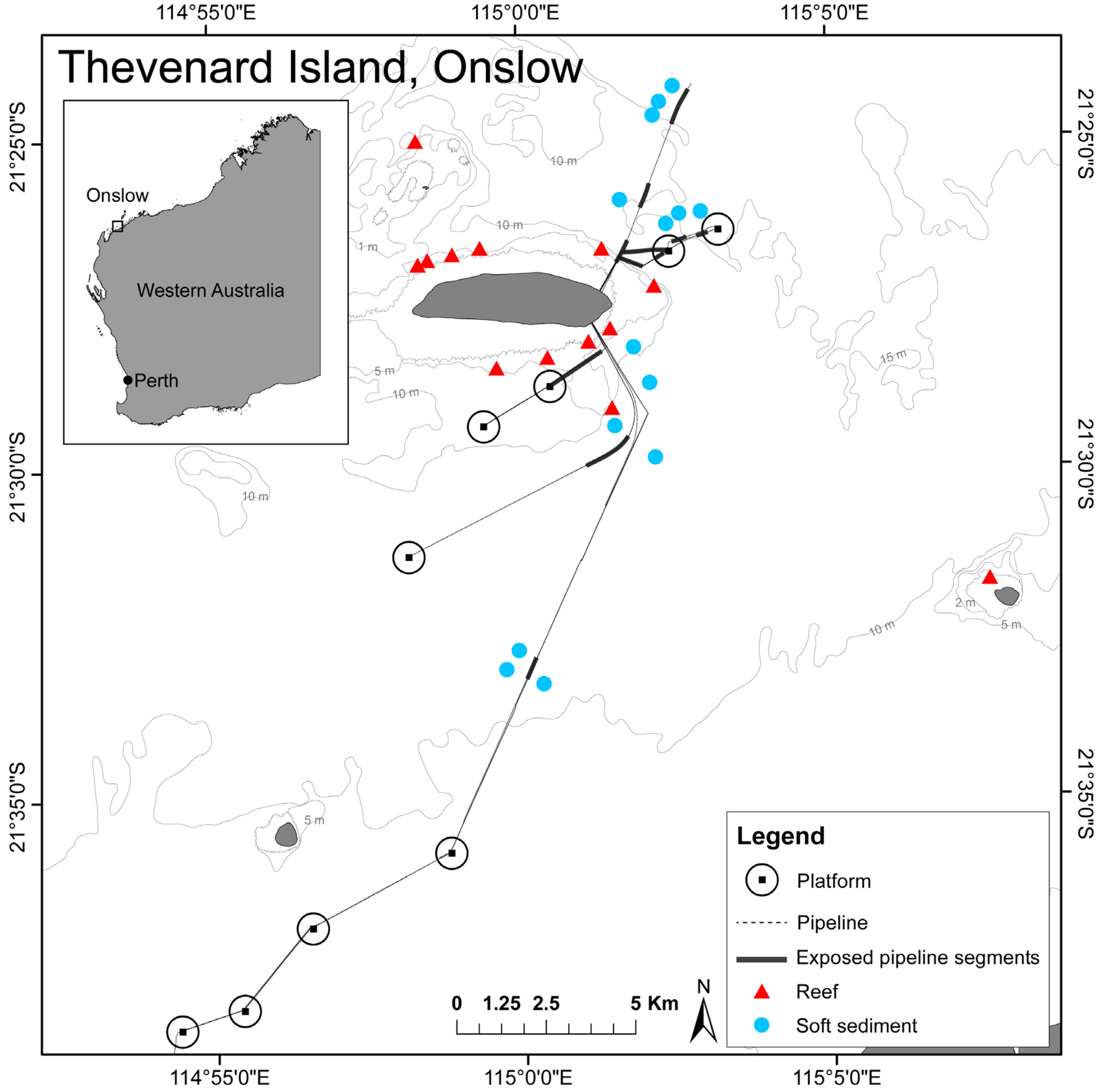


Fig. 4.1 Location of exposed subsea pipelines surveyed and surrounding reef and soft sediment sites, in the vicinity of Thevenard Island, off Onslow, Western Australia (generated using ArcMap v10.7.1, https://desktop.arcgis.com, Memory-Map v1.2, https://memory-map.com, and Adobe Illustrator v25.0.1, https://www.adobe.com/au/products/illustrator.html).

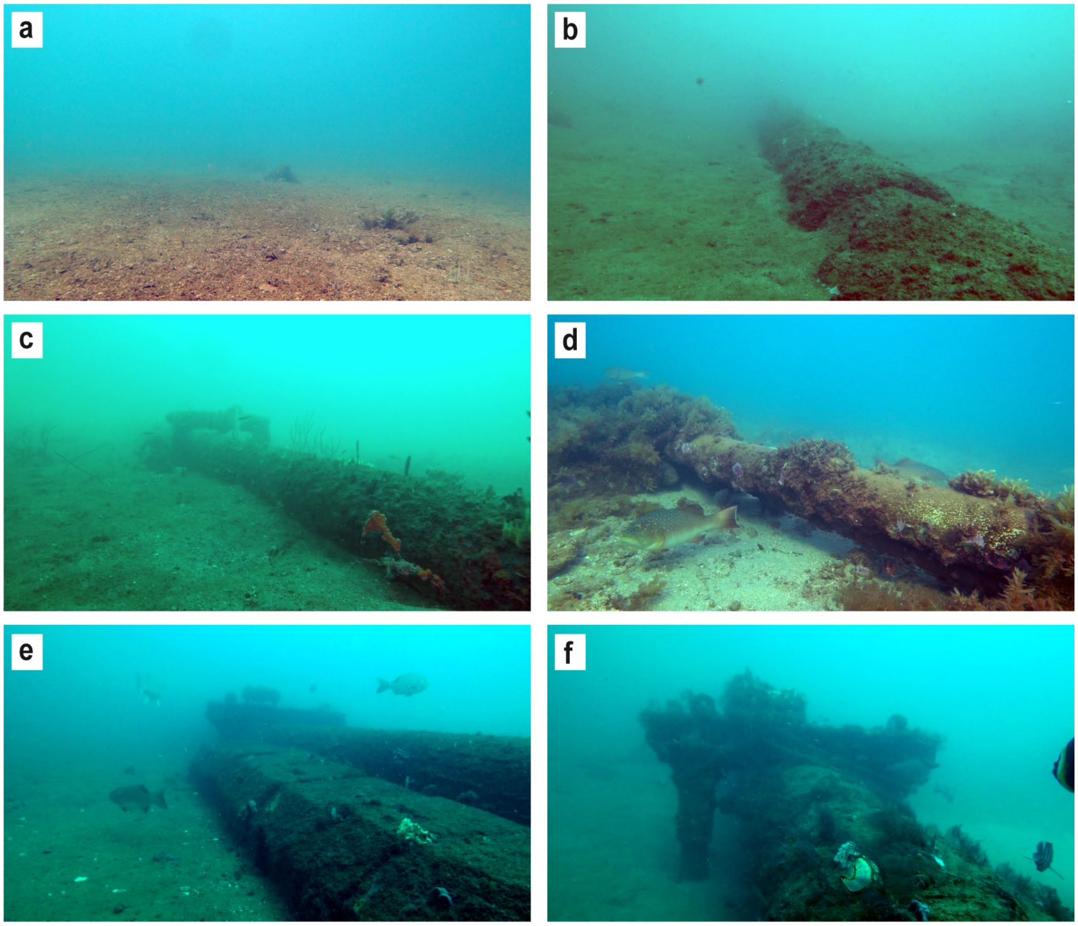


Fig. 4.2 Representative (A) buried, (B) more than half buried, (C) more than half exposed, (D) spanning above the seafloor pipeline positions and associated structures along the pipe, (E) concrete mattress, (F) tie down (generated using Adobe Illustrator v25.0.1, https://www.adobe.com/au/products/illustrator.html).

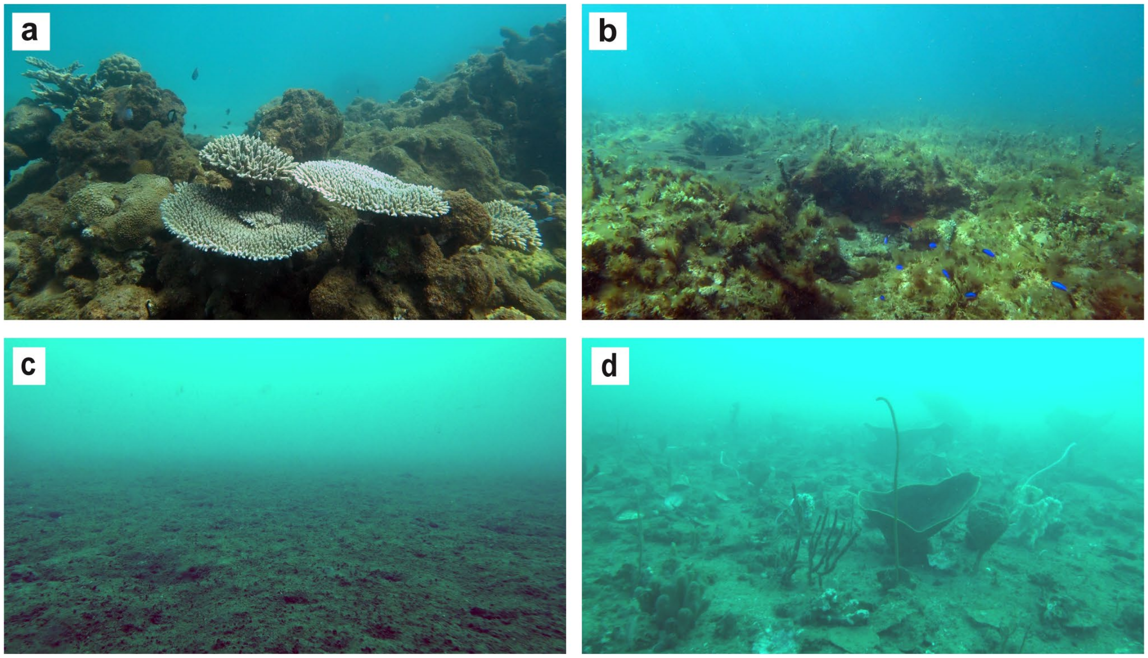


Fig. 4.3 Representative reef (A & B) and soft sediment (C & D) habitats surrounding pipelines in the area around Thevenard Island, Western Australia (generated using Adobe Illustrator v25.0.1, https://www.adobe.com/au/products/illustrator.html).

4.3.2. Sampling technique

Two mini ROVs of a similar size and functionality (SeaBotix vLBV300: 625 mm × 390 mm × 390 mm (l × w × h), ~18 kg; and BlueROV2 Heavy Configuration; 457 mm × 338 mm × 254 mm, ~11 kg) fitted with a stereo-video system (Goetze et al., 2019; Harvey et al., 2010) were used to survey fish associated with pipelines, natural reefs and soft sediment habitats. The two ROVs were operated simultaneously, with the SeaBotix vLBV30 used to survey pipelines and the BlueROV2 used in natural habitats. The stereo-video systems on both ROVs used Sony FDR-X3000 ActionCams mounted in purpose-built housings with a base separation of 590 mm and an inward convergence of 5°. Cameras were set to record at 60 frames per second in a 1080p format. The SeaBotix vLBV300 ROV used a Tritech Ultra Short Baseline Positioning system (USBL) and an Oculus 750D multibeam sonar to help with positioning and navigation, while the BlueROV2 was equipped with Seatrac X150 USBL and X010 Modem.

4.3.3. Pipeline surveys

GIS maps combined with recent hydrographic survey data were used to identify the position of exposed segments of pipeline around Thevenard Island. Live feed from the ROV camera and attached multibeam sonar were then used to locate pipelines *in-situ*. Once located, the ROV operated approximately 1.4 ± 0.05 m from the pipeline on one side only, with the system angled approximately 25° (23.05 ± 0.77°) towards the pipeline to enable a field of view of any undercut sections between the pipe and the seafloor. The system was operated at an average flight speed of approximately 0.54 ± 0.04 m/s (similar to the recommended velocity of stereo-DOV transects, 0.3 m/s (Goetze et al., 2019). Continuous footage of exposed pipelines was collected during active boating, with the vessel trailing behind the stereo-ROV, 100-150 m away. In total, eleven segments of exposed pipeline were surveyed with segment lengths varying between 0.3 - 1.7 km, which was dependent on the level of exposure of the pipeline. Pipeline surveys were completed between 08:30 - 17:00 h to minimise the effects of diel changes in fish behaviour on data collected (Bond et al., 2018a; Myers et al., 2016).

Quantitative comparisons between the reef and soft sediment habitats were undertaken by dividing continuous footage of the pipelines into 50 m transects, with a minimum 20 m separation between transects to ensure independence. To do this, the average flight speed of the stereo-ROV for each segment of pipeline was used to determine the time taken to complete a 50 m transect. Each pipeline transect surveyed encompassed an area 5 m wide × 50 m (250 m^2). The level of pipeline exposure varied across the 88 transects. If a 50 metre transect contained more than 17.5 metres (35% of a 50 m transect) of buried pipeline, it was excluded from image analysis. In total, 88 independent 50 m transects were retained for analyses. For pipeline segments that were included in analyses, on average per transect: 5% (2.33 ± 0.72 m) was in free-span above the seafloor, 55% (27.62 ± 2.10 m) was more than half-exposed, 33% (16.40 ± 2.02 m) was more than half-buried and 7% (3.66 ± 0.93 m) was completely buried.

4.3.4. Reef and soft sediment habitat surveys

Concurrent surveys in natural habitats were undertaken >500 m away from the pipeline or any artificial structure, such as platforms. The ROV was operated from an anchored vessel and was continuously flown for approximately 25 min. As the vessel was stationary during these surveys, the operating range of the stereo-ROV was limited by the tether length (150 m). To avoid tension on the tether and ensure new ground was covered, the ROV was maneuvered in an expanding square around the vessel (similar to polygonal patterns described in Gregoire and Valentine (2007) and Sward et al. (2019). During image analysis, the imagery was split into 50 m × 5 m transects (250 m^2) with a 20 m separation between the end and start of transects as per the pipelines. Determining the start and end of these transects followed the procedure described above for pipeline transects. Imagery was analysed from 150 transects derived from 14 sites in reef habitat, and 148 transects from 14 sites in soft sediment habitat. Surveys in natural habitats were similarly undertaken between 08:30 - 17:00 h.

4.3.5. Stereo-video calibration and video analysis

Stereo-video systems were calibrated before and after fieldwork using the software package ‘CAL’ (https://www.seagis.com.au/bundle.html) following well established protocols and guidelines in (Boutros et al., 2015; Harvey and Shortis, 1998; Shortis et al., 2009; Shortis and Harvey, 1998). All fish counts, identifications, and fork length (FL) measurements (tip of snout to mid of forked caudal fin) were made in EventMeasure Stereo Version 5.25 (https://www.seagis.com.au/event.html). Where fish could not be identified to a species level, individuals were pooled to the next highest taxonomic level, i.e. genera/family. To maintain a defined unit area of sampling across image analyses a horizontal and vertical constrained field of view was set to 2.5 m in either direction of the centre point (x = 5 m, y = 5 m), with a depth (z) range to 7 m. All individuals were counted within this defined sampling area, and those that were observed outside this area were not included in the data set (Goetze et al., 2019). Fish that were identified as having left the area of the transect, which later re-entered (where they could be identified) were only counted once. In order to obtain positioning and time information, the video imagery from the stereo system was synchronised with the video footage collected by the onboard ROV camera. Synchronisation was achieved by referring to the timecode overlay and manually defining the start time of the ROV footage. Both the ROV footage and stereo footage were then paused at the same unique synchronisation point (i.e. a digital clapperboard or physical clap). Calculating the difference in the elapsed time between the stereo and ROV video footage allowed us to define the start time of the high definition footage. Using the event logs collected in the field we were able to skip to the time at which the ROV commenced its survey. For quantitative analyses, this also allowed us to identify fish counts and measurements that were observed within the timed 50 m transects using their corresponding time stamp.

4.3.6. Calculating biomass, feeding guilds and economic value of fish

Fish length was used as a proxy of weight (biomass), using the equation: Weight (g) = $a\times\text{Length (cm)}^b$ (Taylor and Willis, 1998). Relevant slope (a) and intercept (b) values for different species/genera were sourced from FishBase (Froese and Pauly, 2019). Where fish could not be measured due to visual obstruction from other fish or structure (pipe and reef etc.), or were oblique to the camera with neither the head nor tail visible, we allocated that fish a mean length which was based on that specific species within the same habitat (similar to Bach et al., 2019). Classification of feeding guilds for fish were sourced from FishBase based on the ecology and/or diet descriptions (Froese and Pauly, 2019) (see Table App. 4.1). The value of commercial and recreational fish ($AUD/kg) was calculated using the mean wet weight market value for commercial species for 2017/2018 (Gaugham et al., 2019). In total, 39 species for which a corresponding market value was available were recorded (see Table App. 4.2).

4.3.6. Statistical analysis

Statistical analyses were undertaken in PRIMER 7 with PERMANOVA + add on (Anderson et al., 2008). A one way PERMANOVA was used to test for differences among habitats in the numbers of species, overall fish abundance, the overall biomass, differences in feeding guilds and focal species (*Habitat*: 3 levels; pipeline, reef, soft sediment) (Anderson et al., 2001, 2008). Focal species were considered to be those species identified as being dominant in a SIMPER analyses, as well as those species commonly targeted by fishers in the Pilbara area (Ryan et al., 2019). Focal species were *Neopomacentrus aktites* (Western Australian demoiselle), *Pomacentrus coelestis* (Neon damsel), *Thalassoma lunare* (Moon wrasse), *Parupeneus indicus* (Yellowspot goatfish), *Scarus ghobban* (Bluebarred parrotfish), *Pentapodus porosus* (Northwest threadfin bream), *Choerodon cauteroma* (Bluespotted tuskfish), *Choerodon schoenleinii* (Blackspot tuskfish), *Plectropomus spp*. (Coral trout), *Lutjanus carponotatus* (Stripey snapper) and *Lethrinus laticaudis* (Grass emperor). Data were tested for dispersion using PERMDISP (Anderson, 2006) and were analysed using the untransformed data based on a Euclidean distance matrix. When a statistical difference was found ($P < 0.05$, using 9999 permutations), a post-hoc pairwise comparison was completed. P-values from pairwise tests are indicated using $P_{(pairwise)}$. A Monte Carlo bootstrapping correction was used in instances where a low permutation value was obtained for post-hoc tests (< 100), and indicated using P(MC). A Venn diagram showing the number of species shared between habitats was constructed.

To test for statistical differences in the fish assemblage recorded between habitats a one-way PERMANOVA was used (*Habitat*: 3 levels; pipeline, reef, soft sediment). A fourth root transformation was applied to down weight the influence of more common species over those rarely recorded across the data set. A Bray Curtis similarity matrix was used for analysis with a dummy added variable (+1) to account for transects in which no fish were observed. A significant difference was determined when $P < 0.05$ using 9999 permutations, followed by a pairwise comparison to distinguish which habitats were statistically different from one another ($P_{(pairwise)}$). Principal coordinates analysis (PCO) and constrained canonical analysis of principal coordinates (CAP) plots were used to visually represent differences in the abundance and biomass of fish assemblages among habitats. Overlays onto CAP axes were done using SIMPER analyses and selecting the top five species that contributed to group differences (based on similarity/standard deviation values). A leave-one-out allocation test was also used to estimate and classify how distinct samples were relative to each habitat.

4.4. Results

A total of 13,883 fish from 46 families and 207 species were recorded in surveys of pipelines and surrounding reef and soft sediment habitats. The mean number of species varied between habitats (PERMANOVA: $F_{2,382} = 125.82$, $P < 0.001$) with reef having significantly more species than pipeline ($P_{(pairwise)} = 0.008$) and soft sediment habitats ($P_{(pairwise)} < 0.001$), which were also different from one another (pipeline and soft sediment: $P_{(pairwise)} < 0.001$) (Fig. 4.4A). Pipeline and reef habitats however were more similar in the composition of the fish assemblage, sharing 44 species (21% of the total fish assemblage), than pipeline and soft sediment habitats, which shared only ten species (5%) (Fig. 4.5). Fish abundance varied between habitats (PERMANOVA: $F_{2, 382} = 33.339$, $P < 0.001$), but pipeline and reef habitats had similar abundances of fish ($P_{(pairwise)} = 0.904$). Soft sediment habitat had lower abundances of fish than pipeline or reef habitats ($P_{(pairwise)} < 0 .001$) (Fig. 4.4B). A similar pattern was observed for biomass (PERMANOVA: $F_{2,382} = 24.641$, $P < 0.001$), with soft sediment having lower biomass compared to pipeline and reef habitats ($P_{(pairwise)} < 0.001$), which were similar ($P_{(pairwise)} = 0.461$) (Fig. 4.4C).

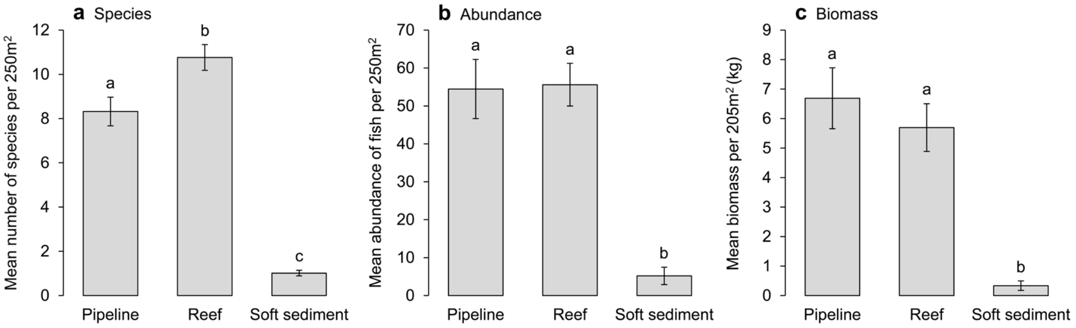


Fig. 4.4 Mean (± SE) number of species (a), abundance (b), and biomass of fish (kg) (c) per transect (50 m × 5 m, 250 m^2) for pipeline, reef, and soft sediment habitats. Statistically similar means are indicated by the same letter (e.g. a) (generated using Microsoft Excel v16.0.5122.1000, https://www.microsoft.com/).

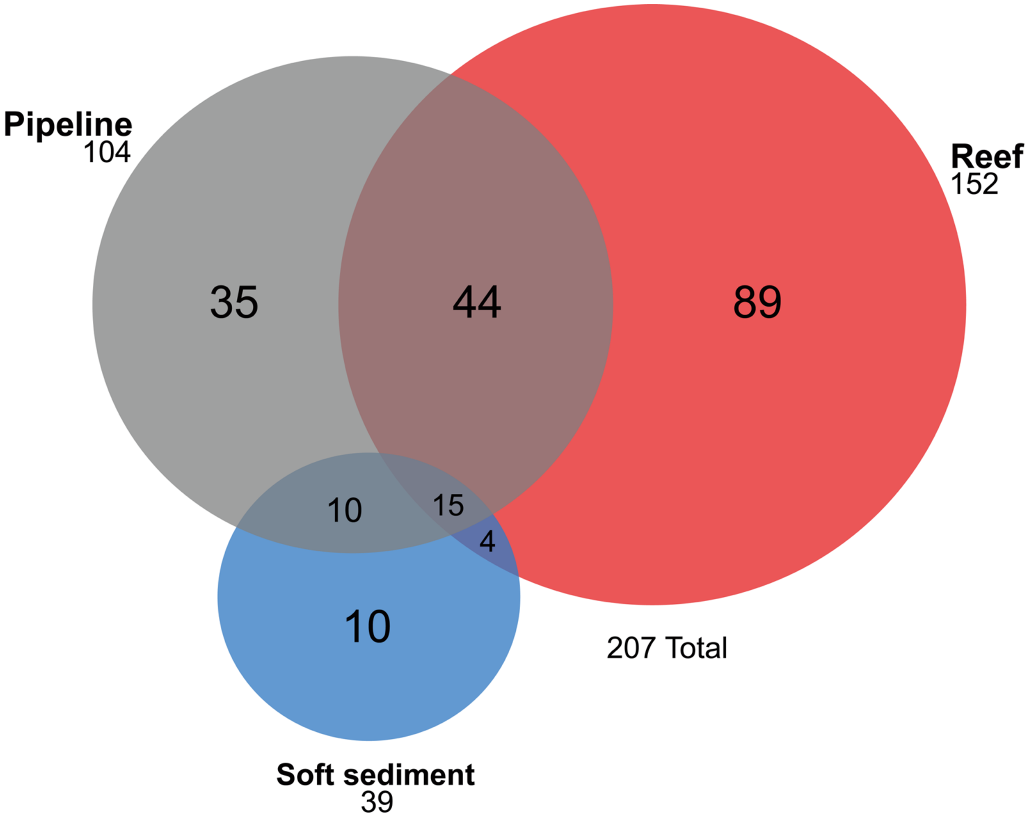


Fig. 4.5 Total number of species recorded at each habitat: pipeline, reef, soft sediment, and across combinations of habitats (generated using Venn Diagram Plotter v1.5.5228.29250, http://omics.pnl.gov/).

4.4.1. Fish assemblage composition

The composition of the fish assemblage differed among habitats (PERMANOVA: $F_{2,382} = 64.833$, $P < 0.001$) with each habitat being distinct from one another ($P_{(pairwise)} < 0.001$). The leave one out allocation success between pipeline, reef, and soft sediment habitats was high overall with 85.38% of samples (327/383) being correctly classified to the correct habitat (Table 4.1). However, there was higher overlap between pipeline and soft sediment habitats, than between reef and pipeline habitats (Table 4.1). A PCO and CAP plot showed a distinct separation between reef and the other habitats (pipeline and soft sediment), which were more similar, but still statistically different from one another ($P < 0.001$; Fig. 4.6A). This separation was driven by high abundances of damselfish species (*P. coelestis* and *Pomacentrus milleri*), *Acanthurus grammoptilus* and *T. lunare* that were observed on natural reefs. Separations between pipeline and soft sediment were less distinct on CAP axis 2 and were driven by fish that are reef associated (*Chromis fumea, N. aktites,* and *Labroides dimidiatus*). The separation was also driven by fish that occupy a combination of sandy areas over or near reef areas (*C. cauteroma*, *P. indicus,* and *P. porous*) (Fig. 4.6A). A similar pattern was detected for biomass at an assemblage level (PERMANOVA: $F_{2,382} = 63.303$, $P < 0.001$), where all habitats differed from one another ($P_{(pairwise)} < 0.001$) with an overall high (86.16%) allocation success for samples by habitat (Table 4.1). Separation and grouping by habitat in the ordinations was driven by high abundances of *P. porosus, C. cauteroma and C. fumea* on the pipeline, and larger *P. indicus* individuals, which created a greater biomass on pipelines than in natural reef and soft sediment habitats (Fig. 4.6B). Conversely, reef samples were driven by the high abundance of damselfishes (*P. milleri*, *Pomacentrus limosus* and *P. coelestis*), as well as *A. grammoptilus,* and *T. lunare*, which were more abundant on the reef in comparison to pipeline and soft sediment habitats (Fig. 4.6B).

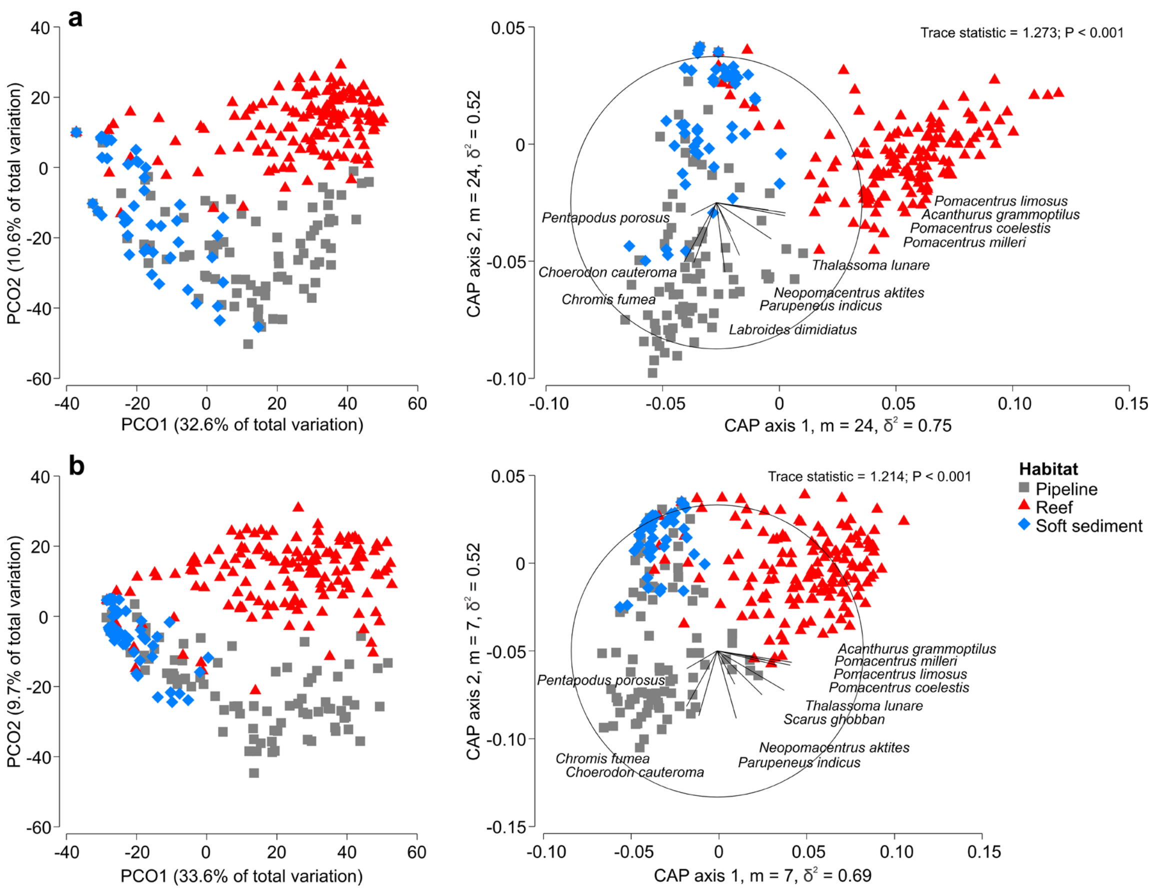


Fig. 4.6 Principal Coordinates Analysis (PCO) and Canonical Analysis of Principal Coordinates (CAP) plots for the abundance (a) and biomass (b) of the observed fish assemblage with SIMPER species overlay showing the differences between habitats: pipeline, reef, and soft sediment. Ordinations are based on four root transformations and Bray Curtis similarities (generated using PRIMER 7 v7, https://www.primer-e.com/).

	Abundance (m:24, total correct: 327/383)					**Biomass (m:7, total correct: 330/383)**				
Habitat	Pipeline	Reef	Soft sediment	Total	Success (%)	Pipeline	Reef	Soft sediment	Total	Success (%)
Pipeline	67	0	21	88	76.14	67	0	21	88	76.14
Reef	2	130	18	150	86.67	4	125	21	150	83.33
Soft sediment	14	1	130	145	89.66	7	0	138	145	95.17

Table 4.1 Leave-one-allocation success of observations to habitat: cross validation for fish abundance and biomass

4.4.2. Feeding guilds

Guild-specific abundance was greater on pipelines than in reef habitats for piscivores ($P(MC)_{(pairwise)} = 0.009$; Fig. 4.7A), generalist carnivores ($P_{(pairwise)} < 0.001$; Fig. 4.7B), and invertivores ($P_{(pairwise)} = 0.048$; Fig. 4.7C), while similar abundances were found between these habitats for planktivores ($P_{(pairwise)} = 0.727$; Fig. 4.7G). Biomass was greater on pipelines than in reef habitats for piscivores ($P_{(pairwise)} = 0.006$; Fig. 4.7A) and invertivores ($P_{(pairwise)} < 0.001$; Fig. 4.7C), but similar for generalist carnivores ($P_{(pairwise)} = 0.195$; Fig. 4.7B) and corallivores ($P_{(pairwise)} = 0.092$; Fig. 4.7F). Reef habitats had greater abundances of omnivore ($P_{(pairwise)} < 0.001$; Fig. 4.7D), herbivores ($P_{(pairwise)} = 0.002$; Fig. 4.7E) and corallivores ($P(MC)_{(pairwise)} = 0.016$; Fig. 4.7F) than pipeline habitats. A greater biomass of herbivores ($P_{(pairwise)} = 0.001$; Fig. 4.7E) and planktivores ($P_{(pairwise)} = 0.010$; Fig. 4.7G) was found in reef habitats compare to pipeline habitats. This differed for omnivores where a greater biomass was found on pipeline habitats ($P_{(pairwise)} = 0.014$; Fig. 4.7D). In general, abundance and biomass of all guilds were lowest in soft sediment habitats ($P_{(pairwise)} < 0.05$), with the expectation of the abundance of generalist carnivore, which was similar to reef ($P_{(pairwise)} = 0.999$; Fig. 4.7B), and the abundance and biomass of corallivores, which were similar to pipeline ($P(MC)_{(pairwise)} = 0.195$; ($P(MC)_{(pairwise)} = 0.202$; Fig. 4.7F).

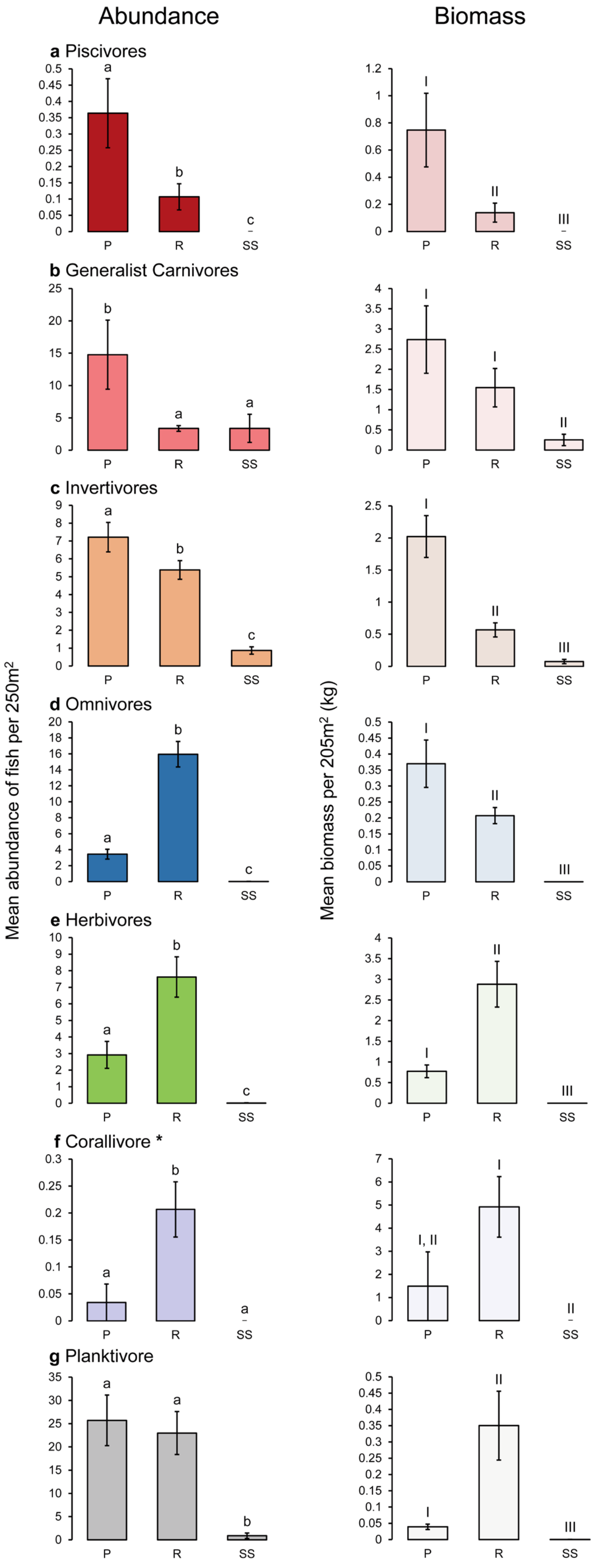

Abundance
Biomass
a Piscivores
b Generalist Carnivores
c Invertivores
d Omnivores
e Herbivores
f Corallivore *
g Planktivore
Mean abundance of fish per 250m²
Mean biomass per 205m² (kg)
P
R
SS
a
b
c
I
II
III
I, II

Fig. 4.7 Mean (± SE) abundance and biomass of fish per transects (50 m x 5 m x 5 m) for feeding guilds: piscivores (a), generalist carnivores (b), invertivores (c), omnivores (d), herbivores (e), corallivores (f), and planktivores (g), between habitats: pipeline (P), reef (R), soft sediment (SS). Statistically similar means are indicated by the same letter for abundance (e.g. a), and roman numerals for biomass (e.g. I). * Biomass of corallivores are represented in grams (g) (generated using Microsoft Excel v16.0.5122.1000, https://www.microsoft.com/).

4.4.3. Focal species

The mean abundance and biomass of focal species differed between habitats (PERMANOVA: $P < 0.05$; Fig. 4.8). For *N. aktites*, *P. indicus* and *S. ghobban* the mean abundance was similar between pipeline and reef habitats ($P_{(pairwise)} = 0.565$; $P(MC)_{(pairwise)} = 0.835$; $P_{(pairwise)} = 0.309$, respectively) with both habitats having a greater abundance of fish than soft sediment habitat ($P_{(pairwise)} < 0.001$; Fig. 4.8A,D,E). A similar pattern was observed for the biomass of these species, with the exception of *P. indicus* where a greater biomass was recorded on the pipeline in comparison to reef ($P_{(pairwise)} = 0.010$) and soft sediment ($P_{(pairwise)} < 0.001$), which also differed from one another ($P_{(pairwise)} < 0.001$; Fig. 4.8D). By contrast, the mean abundance and biomass of *P. coelestis* was greater in reef habitat than on the pipeline ($P_{(pairwise)} < 0.001$) and soft sediment habitats ($P_{(pairwise)} < 0.001$) where no individuals were encountered (Fig. 4.8B). A greater abundance and biomass of *T. lunare* was detected on reefs than pipelines ($P_{(pairwise)} < 0.05$) and soft sediment habitats ($P_{(pairwise)} < 0.001$), which also differed from one another ($P_{(pairwise)} < 0.001$; Fig. 4.8C). A greater abundance of *P. porosus* was found on the pipeline in comparison to natural habitats (reef: $P_{(pairwise)} < 0.001$; soft sediment: $P_{(pairwise)} = 0\ .002$), where soft sediment had a higher abundance than reef habitat ($P(MC)_{(pairwise)} < 0.001$; Fig. 4.8F). A similar pattern was observed for the biomass of *P. porosus* (Fig. 4.8F).

Species that are commonly targeted or retained by recreational fishers in the Pilbara region (*C. cauteroma*, *C. schoenleinii*, *Plectropomus* spp., *L. carponotatus*, and *L. laticaudus*) were more abundant on pipelines than reef and soft sediment habitats (P $P_{(pairwise)} < 0.05$; Fig. 4.8G-K), which were also different from one another. *Choerodon* *cauteroma* and *L. laticaudus* were exceptions where reef and soft sediment had a similar abundance of individuals ($P(MC)_{(pairwise)} = 0.414$; $P(MC)_{(pairwise)} = 0.623$, respectively; Fig. 4.8G and K), which were less than the pipeline. The biomass of these species was also consistently greater on pipelines than in natural habitats ($P_{(pairwise)} < 0.05$; Fig. 4.8F-K) with soft sediment having a lower biomass than reefs. Again, *C. cauteroma* and *L. laticaudus* were exceptions with natural habitats having a similar biomass of fish ($P_{(pairwise)} = 0.056$; $P(MC)_{(pairwise)} = 0.153$, respectively; Fig. 4.8G and K) which were less than the pipeline.

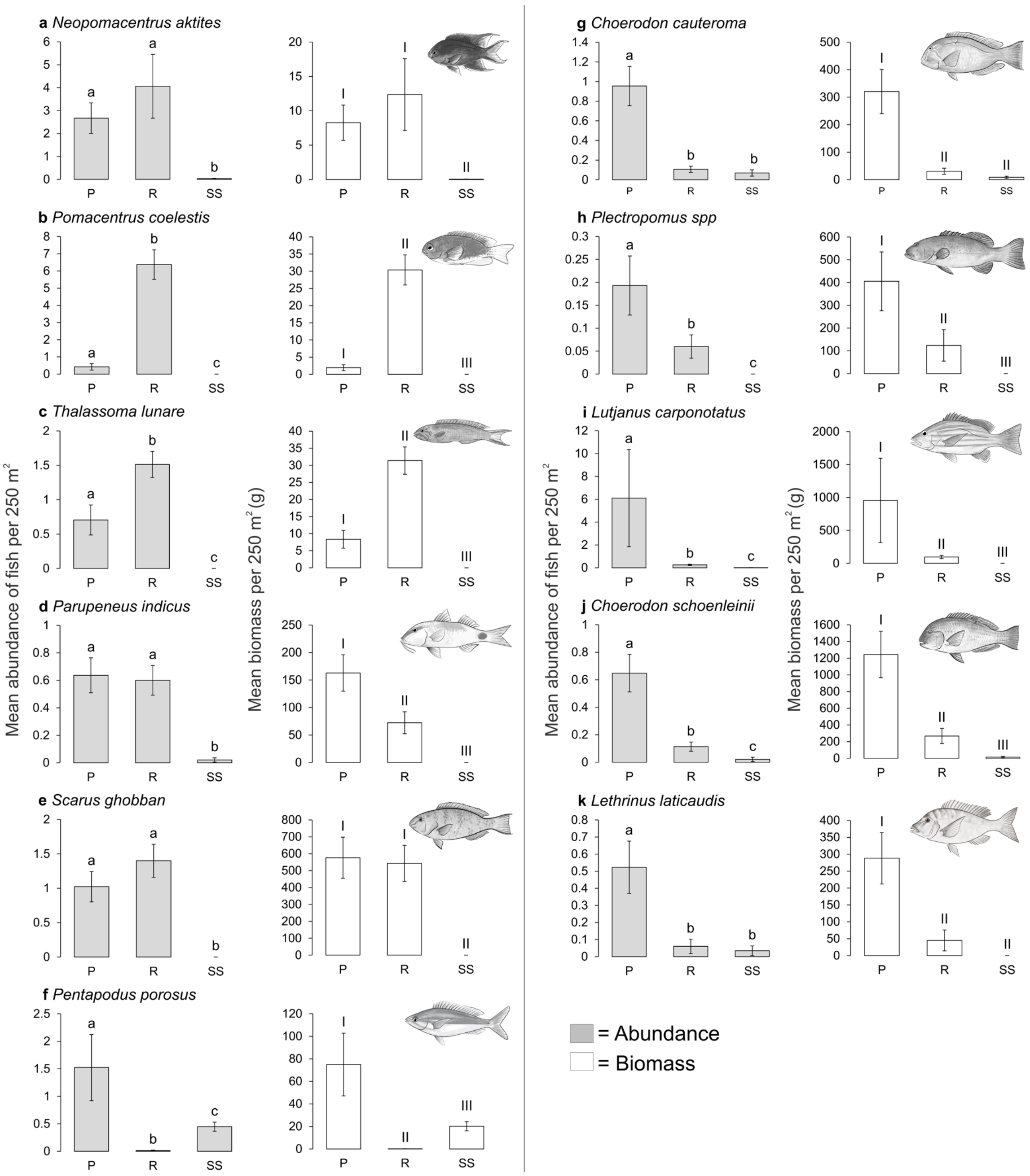


Fig. 4.8 Mean (± SE) abundance and biomass of *Neopomacentrus aktites* (a), *Pomacentrus coelestis* (b), *Thalassoma lunare* (c), *Parupeneus indicus* (d), *Scarus ghobban* (e), *Pentapodus porosus* (f) *Choerodon cauteroma* (g), *Plectropomus* spp. (h), *Lutjanus carponotatus* (i), *Choerodon schoenleinii* (j), *Lethrinus laticaudis* (k) among habitats: pipeline (P), reef (R), soft sediment (SS). Statistically similar means are indicated

by the same letter for abundance (e.g. a), and roman numerals for biomass (e.g. I) (generated using Microsoft Excel v16.0.5122.1000, https://www.microsoft.com/).

4.4.4. Economic value

An equivalent area (250 m^2) of pipeline had ~71% more biomass (3.5 times more) than natural reefs, and ~98% more biomass (44.5 times more) than soft sediment habitat for species commonly retained by commercial and recreational fishers in the region (see Table App. 4.2). When converted into an economic dollar value ($AUD) based on market prices for wet weight (see Table App. 4.2), an equivalent area of pipeline contained an economic value of fish that was 3.4 times greater than the adjacent natural reef habitat and 57 times more than the adjacent soft sediment habitat (Table 4.2).

	Pipeline	Reef	Soft sediment
Mean biomass per 250 m^2 (kg)	4.90 ± 0.92	1.40 ± 0.25	0.11 ± 0.07
Mean catch value per 250 m^2 ($AUD)	30.20 ± 4.76	8.82 ± 1.88	0.53 ± 0.31
Number of transects	88	150	145

Table 4.2 Economic value of species retained by commercial and recreational fishers

4.5. Discussion

Fish assemblages associated with pipelines were distinctly different from nearby natural reefs and soft sediment habitats. Pipelines were characterised by a greater abundance and or/biomass of species from higher trophic levels (i.e. piscivores, generalist carnivores, and invertivores) compared to reef and soft sediment habitats, which resulted in them having a higher fisheries value than equivalent areas of reef and soft sediment. In comparison, natural reefs had a greater proportion of omnivores, herbivores, and corallivores than pipelines and soft sediment habitats, which was likely due to the greater cover of benthic communities and associated food sources (i.e. coral and macroalgae) observed on reefs. Soft sediment habitat was dominated by generalist carnivores and invertivores, but overall had much lower abundances than other habitats.

These findings differ from a previous study undertaken in the same region at similar depths, which indicated that fish assemblages were similar in both the pipeline and soft sediment habitats (Bond et al., 2018c). However, the disparity in findings between this study and that of Bond et al. (2018c) are likely attributable to the sampling technique (stereo-BRUVs) attracting fish from other habitats. For example, attracting fish from the soft sediment habitat to the baited camera located near the pipeline. Stereo-BRUVs have been shown to sample a broad range of species (Harvey et al., 2007; Watson et al., 2005), and are particularly effective at sampling large, highly mobile carnivorous fishes. The distance these species travel to a bait is unknown (Schramm et al., 2020b), and fish do tend to aggregate around the bait in numbers which are much higher than count data collected by other sampling techniques (Langlois et al., 2010; Watson et al., 2005). By comparison, the stereo-ROV collects data at a much finer scale than stereo-BRUVs and is particularly good for habitat affiliated fishes (Schramm et al., 2020b). It is acknowledged that the stereo-ROV technique may have some avoidance biases due to the noise associated with the ROV thrusters, electronics and tether vibrations (Schramm et al., 2020b). This avoidance may have been heightened within soft sediment habitat where there was limited structure for fish to take shelter. Consequently, it is possible that in soft sediment habitat fish avoided the ROV, increasing the difference between soft sediment areas and pipelines (Schramm et al., 2020b).

Density-dependent mechanisms, such as habitat availability, competition, and predation have likely contributed to the observed abundance of fish along the pipeline. The presence of predatory fish on artificial reefs has been associated with food availability (Bohnsack, 1989), both on and off the structure, where searching for prey is likely optimised with lower energy expenditure (Simon et al., 2011, 2013). For resident predatory species, such as *Plectropomus* spp., the limited spatial area of the pipeline may enhance prey encounters, whereby food sources are potentially concentrated along the structure. This may also be true for invertivores that consume sessile invertebrates that were associated with pipelines and will likely be influenced by patterns in epifaunal growth (McLean et al., 2017). Habitat forming biota, particularly sponges, support a range of marine fauna (e.g. fish, crustaceans and echinoderms) and likely contribute a link between species of a higher trophic level (Wulff, 2006).

Fish that use reefs for shelter by day, but forage in different habitats by night, such as in seagrass or macroalgae beds or open sand, are also likely to benefit from the physical presence of structure within open and sparse habitats, where suitable food resources may be prevalent (Bohnsack and Sutherland, 1985) as opposed to shelter sites in natural reef habitat which may be distant from foraging habitat. Networks of pipelines are typically situated on sandy substrates and foraging efficiency may be increased for some species that feed on infaunal burrowing organisms (e.g. crustaceans, polychaetes and molluscs). Lutjanid species display this foraging behaviour at night by migrating to nearby habitats away from reefs to feed on invertebrates (Harvey et al., 2012; Nagelkerken et al., 2000; Newman and Williams, 1995; Travers et al., 2018). Some lethrind species also migrate away from reefs to forage over soft substrate during the night (Currey et al., 2015; Newman and Williams, 1995, 2001; Travers et al., 2006). Similar diel variations have been documented on subsea pipelines, with fewer encounters of fish and number of species at night, compared to during the day using industry ROV footage (Bond et al., 2018a; McLean et al., 2017). Therefore, the high abundance of lujtanids and certain lethrinids on the pipeline may not be due to prey availability on pipelines, but rather due to the physical structure acting as a daytime shelter. Fish that forage in other habitats and return to pipelines for shelter may play an important role in concentrating nutrients around pipelines via waste excretion (Bond et al., 2018a; Layman et al., 2013; Marnane and Bellwood, 2002; Shantz et al., 2015), which may, in itself, result in increased species, abundance and biomass of a range of species. Foraging fish returning to the pipeline may also be preyed upon by resident piscivores, which in turn excrete nutrients at the pipeline. Future work assessing diurnal variations both on and off the pipeline will provide better insights into the behaviour of fish that occupy these structures during the day and their role in facilitating nutrient and energy transfer onto the pipeline from surrounding areas.

Prey availability, structural complexity, and habitat characteristics influence the distribution of reef fishes (Friedlander and Parrish, 1998; Wen et al., 2013a). Pipeline features such as span length (Love and York, 2005; McLean et al., 2017), wellheads (McLean et al., 2018; Pradella et al., 2014) and field joints (McLean et al., 2020b) have been associated with high abundances of fish, particularly of predatory species (e.g. lujtanids, epinephelids, and sebastids). Our findings were consistent with previous subsea pipeline studies (Bond et al., 2018a, 2018b, 2018c; Love and York, 2005; McLean et al., 2017, 2020b), whereby predatory fish (e.g. in this study, *L. carponotatus* and *Plectropomus* spp.) were commonly observed near pipe spans or utilising small interstitial spaces created between concrete mattresses and the pipe. Ambush predators, such as *Plectropomus* spp., likely rely on structural features that limit visibility to prey, thus increasing capture success (Frisch, 2006; Wen et al., 2013b). Juvenile *Plectropomus* spp. display a strong association with *Acropora* corals over sandy substrates as the morphological complexity of *Acropora* skeletons provides shelter (Wen et al., 2013b), and importantly this *Acropora* edge habitat has a variety of food sources, with a high prevalence of small cryptic fishes around the coral, and invertebrates in the sandy substratum (Wen et al., 2013a, 2013b). Shelter size requirements are also likely to change as fish grow due to the effectiveness of shelter and ease of rapid escape from predators (Kerry and Bellwood, 2012). Although pipeline features, such as spans and concrete mattresses, may not represent typical refuges (e.g. live coral) for some predatory fish, their structural complexity may serve as favourable habitats for both access to prey and refuge from predators.

Population estimates of some commercially and recreationally important fish species are influenced by behavioural biases towards divers, particularly in areas exposed to fishing pressure where fish exhibit more pronounced avoidance responses (Gray et al., 2016; Guidetti et al., 2008; Kulbicki, 1998; Lindfield et al., 2014). Behavioural reactions of avoidance or attraction to ROVs vary and are likely to be species-specific (Laidig et al., 2013). It is possible that fish responses to the ROV varied across habitat types. In particular, natural reefs have high structural complexity, which allows fish to shelter and potentially be obscured in surveys. Likewise, flight responses as the ROV approaches may be heightened in open and sandy habitats due to the lack of structure (Schramm et al., 2020b). As pipelines are often the only shelter in an open and sandy environment, any avoidance towards the ROV is likely to result in fish fleeing near the structure in close proximity, where they can still be observed in the video imagery (Schramm et al., 2020b). However, this is likely to be dependent on whether cautious fish take shelter on the side of the pipeline where ROV operations are taking place, still permitting observations. In some cases, fish may flee to the opposite side of the pipeline and be out of the field of view, limiting pipeline fish counts to a more conservative estimate. This is likely to be more of an issue where segments of pipeline are fully exposed, creating a larger obstruction in video imagery, in comparison to free-spanning segments of pipeline where fish may still be captured in the field of view underneath the pipe. Therefore, predatory fish that are commonly targeted by fishers may have been underestimated in the reef and soft sediment habitats due to potential avoidance behaviours in areas of high structural complexity or lack of structure, but may also have been underestimated where pipelines were fully exposed and when fish fled to the opposite side of the structure out of view of ROV cameras.

Previous studies have documented that pipelines can provide habitat for a greater abundance of fish than adjacent soft sediment habitat (Bond et al., 2018a, 2018b). However, to our knowledge, the present study is the first to demonstrate that pipelines can hold a similar abundance and biomass of fish per unit area compared to natural reefs. Furthermore, the biomass of species commonly targeted by fishers that were recorded on pipelines was approximately 3.5 times greater than natural reefs, highlighting the potential fisheries value of these structures (Bond et al., 2018a, 2018b, 2018c; Love and York, 2005; McLean et al., 2017, 2018, 2020b). Bond et al. (2018b, 2018c) reported a higher biomass of targeted fish species associated with pipelines compared to adjacent natural habitats (predominantly soft bottom areas), ranging from two to eight times greater. The present study demonstrated even greater disparities in biomass between pipeline and natural habitats (3.5 times more than reef and 44.5 times more than soft sediment) than previous studies. We also note that our pipeline estimates may be conservative as we effectively only surveyed one side of the larger pipelines. However, it is possible that fish estimates in reef habitat were also similarly conservative and potentially under represented due to fish seeking shelter in or around the reef, obscuring their view from the stereo-ROV (Schramm et al., 2020b). These disparities are likely attributable to the biases of the sampling method (stereo-BRUVs) used by Bond et al. (2018b, 2018c) in comparison to the biases of the stereo-ROV method used in the present study (discussed in Schramm et al., 2020b).

Predatory fish, such as *L. carponotatus*, *L. laticaudus, C. schoenleinii, Plectropomus maculatus* and *Plectropomus leopardus,* are some of the most commonly retained species by recreational anglers in the Pilbara region (Ryan et al., 2019). The prevalence of these species associated with pipelines, which were higher in abundance and biomass than surrounding natural habitats, suggests that these structures offer an extractive value for fishers, similar to purposely deployed artificial reefs (Florisson et al., 2020; Sutton and Bushnell, 2007). In the North Sea, oil and gas pipelines are commonly targeted by commercial fishers, with ~36% of trips taking place within 200 m of a pipeline, and >1% which actively target these structures (Rouse et al., 2018). The predatory fish assemblages that characterise pipeline infrastructure in the Pilbara region of north-western Australia exhibit high ecological and socioeconomic value, indicating that retaining these structures *in situ* offers significant ecological and community benefits.

Whether the higher fish numbers and biomass we recorded on these oil and gas pipelines are caused by fish production or is due to attraction from other nearby habitats is not clear (the production vs. attraction debate (Bohnsack, 1989). This is partly due to difficulties involved in demonstrating an overall increase in regional fish biomass after the installation of such structures, whilst controlling for natural variation, external fishing pressures and possible immigration (Fowler and Booth, 2012). Claisse *et al.* (2014) demonstrated that the secondary production on oil and gas platforms in California was 27.4 times more than natural rocky reefs at similar depths. To assess secondary production (the formation of new animal biomass from growth for all individuals in a given area during a period of time) we would need to resample these pipelines repeatedly. However, given these structures have been *in situ* for 17-29 years (at the time of this study), it is likely that they contribute directly to biomass production rather than simply attraction from the surrounding area. Removal of these structures would therefore likely result in a net habitat loss, resulting in a net loss of production in this region.

Identifying particular habitat features of pipelines that drive fish associations would be of benefit for decommissioning of structures, including planning and understanding their ecological value. The distinct fish assemblage observed on the pipelines suggests these structures are not surrogates of natural reefs or soft sediment habitats, but may offer additional structural complexities and conditions that are favourable for certain species to seek refuge, particularly those species of a high trophic level that are also considered of value for fishing activities. In general, the more complex a habitat is, the greater the species richness will be (Charbonnel et al., 2002; Gratwicke and Speight, 2005), as it provides a variety of niche microhabitats/structural features suitable for particular taxa to inhabit (e.g. caves, crevices, and other interstitial spaces). Hence the high species count observed within natural reef habitats. Fish communities on pipelines will only mimic aspects of natural reefs if they share similar habitat features favourable for refuge (Perkol-Finkel et al., 2006) and while differences in habitat features remain, different fish assemblages will also be found (Burt et al., 2009; Folpp et al., 2014). Anecdotally we observed that where habitat complexity increased around small structures, such as spans, rock dumps, and concrete mattresses along the pipeline, in addition to high epifaunal growth, the abundance and number of species of fishes increased (similar to Bond et al., 2018a, 2018b; 2018c ; Love and York, 2005; McLean et al., 2017, 2018, 2020b). Understanding how fish utilise these small structures along the pipeline as habitat may be useful for enhancing artificial reef designs (McLean et al., 2020a). Further work is needed on nearshore pipeline systems, focusing on covariates such as pipeline features, diameter, depth, and distance from natural reefs, and a better understanding of day/night residency of fish, all of which would provide greater clarity around the ecological, social and economic value of structures associated with subsea pipelines.

Chapter 5: General Discussion

Photo credit: John Totterdell and Chevron

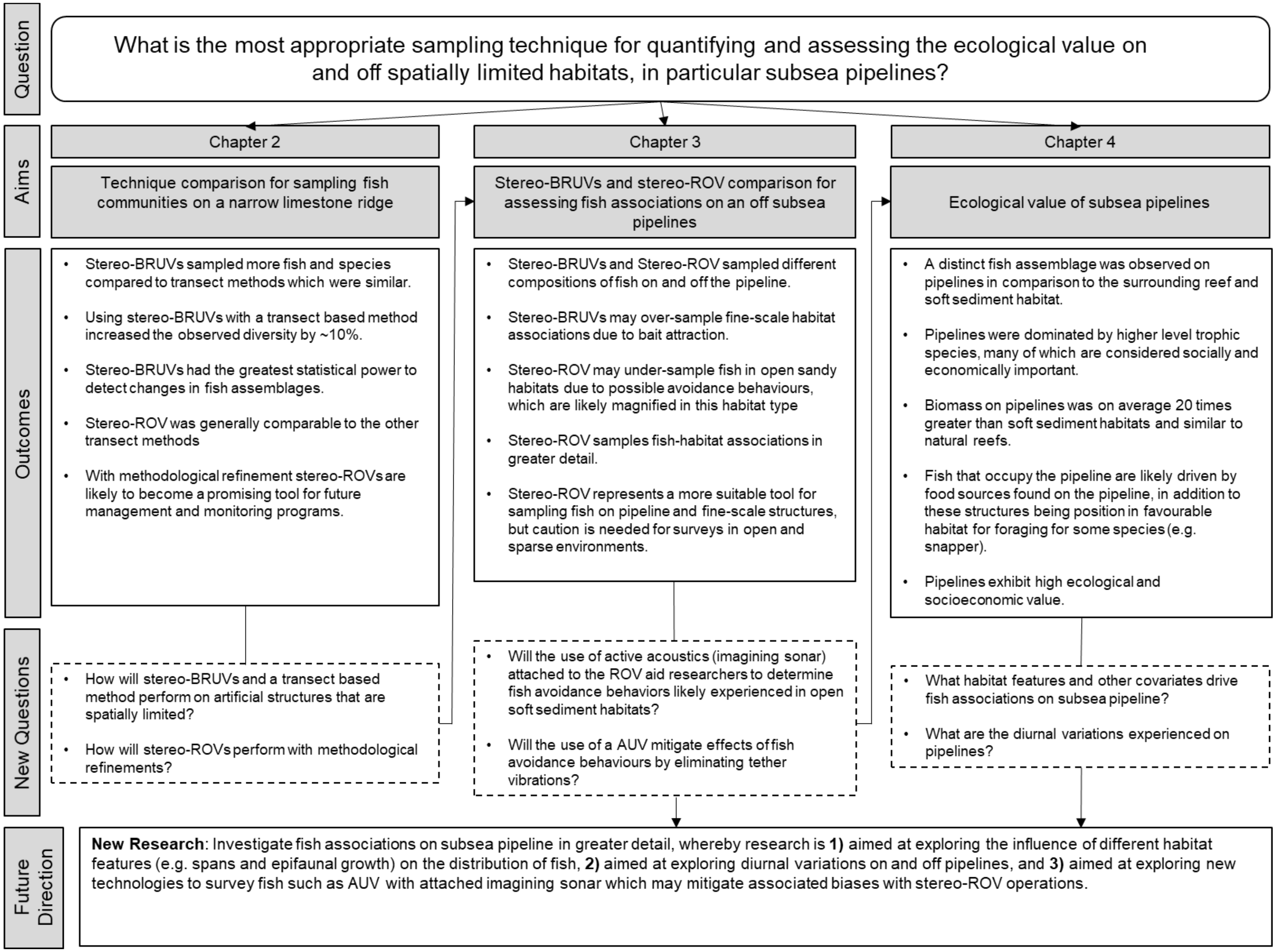
5.1. Summary of findings

The aim of this thesis was to determine the most appropriate technique for sampling fish communities on and off spatially limited habitats, and employ this technique on and off subsea pipelines to investigate patterns in the distribution, abundance and biomass of fish. With oil and gas decommissioning activities expected to increase in the coming decades, regulators will need to make informed decisions on appropriate decommissioning outcomes. While these decisions will most likely be made on a case-by-case basis, it is essential that standardised and repeatable sampling techniques are used to collect accurate and reliable ecological data. This data can be used to communicate what is living on and around these structures to both stakeholders and regulators. Multiple sampling techniques have been used to assess fish associations on subsea pipelines (e.g. stereo-BRUVs, industry ROV and AUV footage, and now stereo-ROVs) and the limitations and biases of each technique may influence their ability to sample such spatially restricted habitats in a representative manner. I conclude that stereo-ROVs are the most appropriate sampling tool for this type of application as the imagery they collect can be used to count and measure the lengths of fish *in-situ* with a defined sampling area (Fig. 5.1). They also allow sampling in waters too deep for SCUBA diving and have the fine scale controlled manoeuvrability that other techniques such as a towed or drift video do not have.

In chapter two, I compared the performance of four stereo-video techniques (diver operated stereo-video, slow towed stereo-video, stereo-video remotely operated vehicle, and baited remote underwater stereo-video systems) along a narrow limestone reef. The outcomes of this study showed that stereo-BRUVs recorded a greater diversity and abundance of fish than the three transect based techniques, which were more similar (Fig. 5.1). These differences were most likely attributable to the use of bait in stereo-BRUVs surveys and longer sampling periods per replicate, extending arrival times of species (e.g. shark and ray encounters). Lower variance experienced between and within stereo-BRUVs samples (likely a function of MaxN) also resulted in greater statistical power to detect change. Having greater power can often mean lower field and lab cost as fewer replicates are required to detect change. However, this can be compromised if the sampling technique is not remote (i.e. diver based) as the amount of replicates allowed per day is restricted due to health and safety guidelines. I suggest here that a mini stereo-ROV could be a promising tool for surveying fish remotely with some simple methodological refinements (Fig. 5.1). In chapter three, I compared the sampling abilities of stereo-BRUVs and stereo-ROVs on a network of subsea pipelines and nearby natural reef and soft sediment habitats. Similar to chapter two, the two sampling techniques observed different compositions of fish, with stereo-BRUVs sampling greater species count, abundance and biomass of fish across habitats than stereo-ROVs. However, greater disparities were observed in soft sediment sites where stereo-BRUVs sampled ~68% more species. This large difference was likely a reflection of using bait to attract fish in a sparse and open habitat and/or fish avoidance likely experienced with ROV operations due to thruster, electronic and tether vibration noises (Fig. 5.1).

Collecting fish data on spatially limited habitats such as pipelines, requires the use of an appropriate technique capable of collecting data at an appropriate spatial scale. It is important that the data collected is representative of the targeted habitat and not drawn from the surrounding area. While stereo-BRUVs have become a standardised approach for surveying fish across a range of habitats (Cappo et al., 2006; Langlois et al., 2020), particularly those that exceed scientific SCUBA limits (>30 m), their application on spatially limited habitats, such as pipelines, may not represent best practise. My research demonstrated that stereo-BRUVs could often be considered to outperform transect based sampling techniques (stereo-DOV, stereo-STV (chapter 2), and stereo-ROV (chapter 2 and 3)) as they recorded greater species richness and often a greater abundance and biomass of fish. However, these measurements of fish may not be representative of a segment of narrow limestone ridge or pipeline as fish estimates are most likely inflated due to area of attraction from the bait, which may exceed the footprint of the structure. This area of attraction is influenced by the dispersal of the bait plume (a function of soak time, current velocity and benthic topography)(Cappo et al. 2001, 2003, 2006) and the feeding activity around the camera (Dorman et al., 2012; Harvey et al. 2007; Watson et al. 2005, 2010). It also needs to incorporate and account for the behaviour of fish, where the mobility and site fidelity among species and individuals are likely to vary (Cappo et al., 2003). As a result, the area of attraction can be site-specific and not be equal for all species, or different class sizes within the same taxa. While it is possible to model the bait plume and the behaviours of some fish that are bait affiliated, to apply this to a whole community of fish for a particular area would be logistically difficult and misleading. As discussed, transect based approaches have the ability to define the area sampled and can be constrained to the footprint of a targeted habitat. This is necessary for researchers if they want to scale up estimates to represent a particular length of pipeline or an entire network of pipelines to demonstrate the ecological or fisheries values to stakeholders and regulators. The advantage of mobile sampling also allows researchers to incorporate spatial heterogeneity into sampling which is necessary to capture the ecological values of fine-scale habitat features (e.g. Galaiduk et al., 2017; Saunders et al., 2014). Features such as pipeline spans and epifauna growth influence fish distributions (Bond et al., 2018a, 2018b, 2018c; Love and York, 2005; McLean et al., 2017, 2020b), and these features are likely better represented through stereo-ROV transects than stereo-BRUVs. The stationary nature of stereo-BRUVs means fish estimates are based on MaxN which can compromise the ability to capture all individuals within the data set as some may be displaced during the sampling period (Coghlan et al., 2017). For example, small fish may be displaced by larger fish of the same species or vice versa, influencing abundance and biomass estimates at the time of MaxN (Andradi-Brown et al., 2016; Cappo et al., 2009; Harvey et al., 2012). This is not a problem for transect sampling, but may not survey fish which flee from the sampling area avoiding divers, towed video systems, ROVs or AUVs.

In chapter four, I investigated the fish assemblages associated with subsea pipelines, in comparison to those inhabiting nearby reef and soft sediment habitats. This chapter used stereo-ROVs and a more complete data set than that used for comparison between methods in chapter three. A distinct fish assemblage was observed on the pipelines characterised by high abundances and biomass of fish from higher trophic levels (e.g. piscivores, generalist carnivores, and invertivores) (Fig. 5.1). The associations of these higher trophic level species on the pipeline are likely attributable to ease of access to prey on these habitat restricted structures. Pipelines are often situated in favourable foraging areas for some invertivore species (e.g. snapper; Harvey et al., 2012; Nagelkerken et al., 2000; Newman and Williams, 1995; Travers et al., 2018) and have habitat features which serve as shelter and refuge to support both predator and prey. The high diversity and biomass of fished species found on the pipelines in comparison to the surrounding natural habitat suggests these structures hold significant potential value to recreational fishers (Fig. 5.1). However, whether the presence of these fish is through attraction or production is still debatable (Bohnsack, 1989) and should be explored in further research, particularly if the outcome is to keep these structures in the water.



What is the most appropriate sampling technique for quantifying and assessing the ecological value on and off spatially limited habitats, in particular subsea pipelines?

Chapter 2

Technique comparison for sampling fish communities on a narrow limestone ridge

- Stereo-BRUVs sampled more fish and species compared to transect methods which were similar.
- Using stereo-BRUVs with a transect based method increased the observed diversity by ~10%.
- Stereo-BRUVs had the greatest statistical power to detect changes in fish assemblages.
- Stereo-ROV was generally comparable to the other transect methods
- With methodological refinement stereo-ROVs are likely to become a promising tool for future management and monitoring programs.

- How will stereo-BRUVs and a transect based method perform on artificial structures that are spatially limited?
- How will stereo-ROVs perform with methodological refinements?

Chapter 3

Stereo-BRUVs and stereo-ROV comparison for assessing fish associations on an off subsea pipelines

- Stereo-BRUVs and Stereo-ROV sampled different compositions of fish on and off the pipeline.
- Stereo-BRUVs may over-sample fine-scale habitat associations due to bait attraction.
- Stereo-ROV may under-sample fish in open sandy habitats due to possible avoidance behaviours, which are likely magnified in this habitat type
- Stereo-ROV samples fish-habitat associations in greater detail.
- Stereo-ROV represents a more suitable tool for sampling fish on pipeline and fine-scale structures, but caution is needed for surveys in open and sparse environments.

- Will the use of active acoustics (imagining sonar) attached to the ROV aid researchers to determine fish avoidance behaviors likely experienced in open soft sediment habitats?
- Will the use of a AUV mitigate effects of fish avoidance behaviours by eliminating tether vibrations?

Chapter 4

Ecological value of subsea pipelines

- A distinct fish assemblage was observed on pipelines in comparison to the surrounding reef and soft sediment habitat.
- Pipelines were dominated by higher level trophic species, many of which are considered socially and economically important.
- Biomass on pipelines was on average 20 times greater than soft sediment habitats and similar to natural reefs.
- Fish that occupy the pipeline are likely driven by food sources found on the pipeline, in addition to these structures being position in favourable habitat for foraging for some species (e.g. snapper).
- Pipelines exhibit high ecological and socioeconomic value.

- What habitat features and other covariates drive fish associations on subsea pipeline?
- What are the diurnal variations experienced on pipelines?

New Research: Investigate fish associations on subsea pipeline in greater detail, whereby research is **1)** aimed at exploring the influence of different habitat features (e.g. spans and epifaunal growth) on the distribution of fish, **2)** aimed at exploring diurnal variations on and off pipelines, and **3)** aimed at exploring new technologies to survey fish such as AUV with attached imagining sonar which may mitigate associated biases with stereo-ROV operations.

Fig. 5.1 Thesis flow diagram showing the outcomes and new research questions for each chapter, and future directions

5.2. Limitations of research and future directions

The development of new sampling approaches aimed at overcoming the limitations and biases of more traditional sampling techniques can provide new platforms for assessing fish populations. This research used mini-ROVs with an attached stereo video system to assess fish. As with all newly developed or adopted techniques, technical refinements are required to overcome inherent challenges in the field and mitigate any potential sampling biases. In chapter two the stereo-ROV was operated too high in the water column resulting in the under sampling of fish near the reef. This was overcome in the following chapters by piloting the vehicle approximately 20-50 cm above the substrate (i.e. the recommended height for stereo-DOVs (Goetze et al., 2019)) to ensure that the field of view captured fish in close proximity to the pipeline. While the operating height of the stereo-ROV was resolved through the progression of this thesis, there are several limitations that must be acknowledged using the stereo-ROV, some unavoidable while others can be improved in future research. This research also raised some questions prompting recommended future directions.

5.2.1. The use of different ROVs

A potential limitation of this research was the simultaneous use of two different stereo-ROVs in chapters three and four. For logistical reasons and to increase fieldwork efficiency by allowing concurrent surveys along the pipeline and in natural habitats, two ROVs were employed to collect data. It is possible that using two different models introduced some confounding biases into the experimental design. However, because these systems are similar in size and functionality (SeaBotix vLBV300®: 625 mm × 390 mm × 390 mm (l × w × h), ~18 kg; and BlueROV2® Heavy Configuration; 457 mm × 338 mm × 254 mm, ~11 kg), differences reported across habitats more likely reflected habitat variation than system differences. Laidig et al. (2013) found that in similar habitats more fish reacted to an observation size ROV (148 kg) than a manned submersible (2,222 kg). These differences were likely attributable to the size of each system, the presence of a tether with the ROV, orientation of lighting, and noise outputs (Laidig et al., 2013). Further research into potential biases of using different ROV models of a similar size would be beneficial. Measuring the minimum approach distance (MAD) (as described in Goetze et al. (2017)) for different size fish with different ROVs will facilitate a better understanding of any potential biases associated with using different models of a similar size.

5.2.2. Stereo-ROV flight speed

The mean flight speed of the ROV in this research was 0.54 ± 0.04 m/s $^{-1}$, but ranged from 0.3-1.3 m/s $^{-1}$ at a site level. The flight speed reported in other studies varies from 0.1-1.0 m/s^{-1} and variation is often attributable to ocean currents and drag from the tether (Sward et al., 2019). The influence of ROV speed on fish behaviour has yet to be explored. Understanding potential effects of ROV speed may be an important consideration in the mitigation of potential avoidance behaviours, particularly in open habitats where avoidance may be heightened. As the speed of an ROV increases, there is a probability that the noise associated with the thrusters and strumming from the tether will also increase. Using the MAD of fish as a comparative measure among different ROV speeds and across habitats will help facilitate an understanding of how ROV speed across ground may affect fish behaviours. Such experiments should also mount a sound recorder on the ROV to quantify the level of noise associated with different ROVs at different operating speeds and power settings.

5.2.3. USBL flight paths

Another limitation experienced in this thesis was the inaccurate navigational data recorded by the USBLs during stereo-ROV surveys as a result of multipath errors. This issue was most pronounced in shallow reef habitats that were characterised by vertical relief. Similar multipath errors were experienced in steep rocky habitats with high rugosity, but were improved by ensuring there was a clear transmission path between the ROV and the receiver on the vessel (Pacunski et al., 2008). This can be achieved by live boating, but becomes more of a challenge when operating the ROV while anchored, which is often necessary in shallow reef habitats to avoid collisions with exposed or near exposed reefs (e.g. bommies). USBL navigation accuracy in shallow waters is known to be negatively affected by acoustic responses as a result of pings bouncing off the seafloor and the ocean's surface. To standardise transect definitions across habitats, ROV speed was used as a proxy of distance traveled with respect to time. However, technical refinements are necessary for future research in order to use navigation paths recorded by the USBL system, which are more accurate. This may require live boating for sites which were previously anchored. The challenge with live boating in shallow water is that the tether is very close to the surface, and at greater risk of being cut by the boat's propeller.

5.2.4. Anchored versus live boating surveys

Both anchored and live boating was used while carrying out stereo-ROV surveys. It is possible that live boating may have introduced some fish behavioural biases in response to engine noises, as fish have been reported to react to approaching vessels (Olsen, 1990; Xie et al., 2008). Future research should assess the potential effect of engine noise, by comparing anchored and live boating surveys within the same habitat.

5.2.5. Habitat features along the pipeline

Anecdotally, we observed greater abundances and diversity of fish near pipeline spans and other associated structures along the pipeline, such as concrete mattresses and tie downs. We also observed more fish near areas of high epifaunal growth which created greater structural complexity (similar to Bond et al., 2018a, 2018b, 2018c; Love and York, 2005; McLean et al., 2017, 2018, 2020b). A target study to further understand these associations will provide insights into the mechanisms that likely drive fish occupancy on pipelines, which may be of value in planning decommissioning options (Fig. 5.1). Furthermore, this information could be beneficial in enhancing artificial reef designs. By focusing on covariates such as pipeline position, epifaunal growth, distance from reef, associated structures and pipeline diameter, fish-habitat occupancy on these structures can be explored in greater detail.

5.2.6. Diurnal patterns on the pipeline

Very little is known about the diurnal variations experienced on pipelines (although see Bond et al., 2018a; McLean et al., 2017). Lutjanidae species which were encountered on the pipeline are known to display foraging patterns at night where they shift from structure used as refuge by day to surrounding sandy substrate to feed (Harvey et al., 2012; Nagelkerken et al., 2000; Newman and Williams, 1995; Travers et al., 2018). This movement of fish likely contributes to important nutrient transfers on the pipelines via waste excretion (Bond et al., 2018a; Layman et al., 2013; Marnane and Bellwood, 2002; Shantz et al., 2015). Foraging fish returning to the pipeline for shelter are also likely to provide resources for predatory species. Further research should assess diurnal variations by using active acoustics techniques and positioning acoustic cameras on and off the structure to observe shifts across habitats (similar to Becker et al., 2011a, 2011b; Parsons et al., 2017) (Fig. 5.1). Timed cameras (similar to Myers et al., 2016) could also be used in combination with acoustic cameras (i.e. built into one system) to identify the fish displaying these diurnal variations. Alternatively, selected species can be tagged using acoustic telemetry systems, where fine-scale movements can be explored by positioning an array of receivers around the pipeline (Currey et al., 2015).

5.2.7. New technologies to survey pipelines

While this research recommends stereo-ROVs for sampling fish on subsea pipelines, this sampling approach has some inherent biases and limitations. Some of these may be resolved by using untethered AUV (Fig. 5.1). Comparisons have been made between industry footage collected by hybrid AUVs and ROVs, and yielded similar data (Bond et al., 2020). However, the hybrid AUV employed in this study was equipped with a tether (Bond et al., 2020). The use of an untethered AUV may overcome potential behavioural biases relating to tether vibration, live boating and speed variations (Fig. 5.1). A comparison with stereo-ROVs and untethered AUV could separate these effects. Attaching an imaging sonar to both vehicles could also improve our understanding of flight behavioural responses as systems approach, by extending the range of detection of fishes beyond that of the video.

5.3. Conclusion

Subsea pipelines are an integral component of oil and gas operations that form extensive networks in our waters. However, knowledge on their role as fish habitat is very limited. This research suggests that pipelines support a distinct composition of fish and are characterised by high abundance and biomass of both, commercially and recreationally important species. To facilitate decisions on decommissioning options, it is important that appropriate sampling techniques are used. These techniques should be capable of providing accurate and reliable fish estimates that are representative of the structure and not the surrounding habitat, which could lead to over-evaluations. Because these structures are spatially narrow and typically extend into deep-water, considerations are needed on depth restrictions of techniques and the ability to constrain the sampling area. This thesis set out to determine the most appropriate sampling technique for assessing fish communities on and off spatially limited habitats, in particular subsea pipelines. While there are multiple sampling techniques that can achieve fish data on such structures, stereo-ROV represents one of the most appropriate approaches as it has the ability to measure fish *in-situ* at depth with controlled manoeuvrability using a defined sampling unit, all of which are necessary for detailed communication with stakeholders and regulators for facilitating decisions on decommissioning options. However, caution is needed for studies that aim to sample open soft sediment habitats where fish estimates may be underestimated due to ROV avoidance behaviours. With further methodological refinements and research into stereo-ROV applications, this technique promises to be a useful tool for sampling fish, especially on spatially limited habitats.

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Appendix

1.1 Copyright Statement

Chapter 2: Published in the Journal of Experimental Marine Biology and Ecology. As first author, permission is automatically granted to use this article in this thesis.

To Whom It May Concern I, Karl Schramm, contributed to the design of this study, analysed the data, and wrote and edited the following publication:

Schramm, K.D., Harvey, E.S., Goetze, J.S., Travers, M.J., Warnock, B., Saunders, B.J., 2020. A comparison of stereo-BRUV, diver operated and remote stereo-video transects for assessing reef fish assemblages. J. Exp. Mar. Bio. Ecol. 524, 151273.

Karl Schramm

I, as a co-author, endorse that the level of contribution by the candidate stated above is appropriate.

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Karl Schramm

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1.2 Appendix – Chapter 2

Table App. 2.1 All the species sampled between Stereo-DOV, STV, ROV and BRUV, along the ridge, in Geographe Bay, Western Australia.

Species	**Common name**	**DOV**	**STV**	**ROV**	**BRUV**
Acanthaluteres brownii	Spinytail Leatherjacket		X		X
Acanthaluteres granulatus	Leatherjacket spp				X
Acanthaluteres vittiger	Toothbrush Leatherjacket				X
Achoerodus gouldii	Western Blue Groper		X		
Anampses geographicus	Scribbled Wrasse		X		X
Anoplocapros amygdaloides	Western Smooth Boxfish				X
Anoplocapros lenticularis	Whitebarred Boxfish	X			X
Apogon spp	Cardinalfish	X	X	X	
Aracana aurita	Shaw's Cowfish	X	X		
Aulohalaelurus labiosus	Blackspotted Catshark				X
Aulopus purpurissatus	Sergeant Baker	X	X	X	
Austrolabrus maculatus	Blackspotted Wrasse	X	X	X	X
Bodianus frenchii	Foxfish	X	X	X	X
Centroberyx lineatus	Swallowtail	X		X	X
Chaetodon assarius	Western Butterflyfish	X			X
Cheilodactylus gibbosus	Magpie Morwong	X	X		X
Chelmonops curiosus	Western Talma	X	X	X	X
Choerodon rubescens	Baldchin Groper	X	X	X	X
Chromis klunzingeri	Blackhead Puller	X	X	X	X
Chromis weberi	Weber's Puller	X		X	
Chromis westaustralis	West Australian Puller	X			
Chrysophrys auratus	Snapper	X			X
Coris auricularis	Western King Wrasse	X	X	X	X
Cypselurus angusticeps	Narrowhead Flyingfish				X
Dactylophora nigricans	Dusky Morwong	X	X		X
Dasyatis brevicaudata	Smooth Stingray				X
Enoplosus armatus	Old Wife	X	X	X	X
Epinephelides armatus	Breaksea Cod	X	X	X	X
Eupetrichthys angustipes	Snakeskin Wrasse				X
Girella zebra	Zebrafish	X			X
Glaucosoma hebraicum	West Australian Dhufish	X	X	X	X
Gymnothorax prasinus	Green Moray				X
Gymnothorax woodwardi	Woodward's Moray				X
Halichoeres brownfieldi	Brownfield's Wrasse	X	X	X	X
Hypoplectrodes nigroruber	Banded Seaperch	X			X
Kyphosus sp.	Sea Chub	X		X	X
Meuschenia flavolineata	Yellowstriped Leatherjacket	X	X	X	X

Meuschenia freycineti	Sixspine Leatherjacket		X	X	
Meuschenia galii	Bluelined Leatherjacket	X	X	X	X
Meuschenia hippocrepis	Horseshoe Leatherjacket	X	X		X
Meuschenia spp	Leatherjacket	X			
Mustelus antarcticus	Gummy Shark				X
Myliobatis australis	Southern Eagle Ray				X
Neatypus obliquus	Footballer sSweep	X	X	X	X
Nemadactylus valenciennesi	Queen Snapper	X	X	X	X
Neosebastes pandus	Bighead Gurnard Perch				X
Notolabrus parilus	Brownspotted Wrasse	X	X	X	X
Ophthalmolepis lineolatus	Southern Maori Wrasse	X	X	X	X
Orectolobus halei	Gulf Wobbegong	X			X
Othos dentex	Harlequin Fish				X
Parapercis haackei	Wavy Grubfish			X	
Paraplesiops meleagris	Southern Blue Devil	X			X
Parequula melbournensis	Silverbelly			X	X
Paristiopterus gallipavo	Yellowspotted Boarfish				X
Parma mccullochi	Mcculloch's Scalyfin	X	X		X
Parma victoriae	Scalyfin	X	X	X	X
Parupeneus chrysopleuron	Rosy Goatfish	X			X
Parupeneus spilurus	Blacksaddle Goatfish		X	X	
Pempheris spp	Sweeper	X	X	X	X
Petroscirtes breviceps	Shorthead Sabretooth Blenny				X
Psammoperca waigiensis	Sand Bass		X		
Pseudocaranx spp	Trevally	X	X	X	X
Pseudolabrus biserialis	Redband Wrasse	X	X	X	X
Rhabdosargus sarba	Silver Bream				X
Scorpis aequipinnis	Sea Sweep		X		X
Scorpis georgiana	Banded Sweep	X			
Seriola hippos	Samsonfish	X	X	X	X
Sillaginodes punctata	King George Whiting				X
Sillago schomburgkii	Yellowfin Whiting				X
Sphyraena obtusata	Striped Barracuda	X	X	X	X
Tilodon sexfasciatus	Moonlighter	X	X		X
Trachinops brauni	Bluelined Hulafish		X	X	
Trachinops noarlungae	Yellowhead Hulafish	X	X	X	X
Trachurus novazelandiae	Yellowtail Scad				X
Trygonoptera ovalis	Striped Stingaree				X
Trygonoptera personata	Masked Stingaree	X			X
Trygonorrhina dumerilii	Southern Fiddler Ray				X
Upeneichthys vlamingii	Bluespotted Goatfish	X	X	X	X
Urolophus circularis	Circular Stingaree	X			

Table App. 2.2 Cost Analysis for techniques DOV, STV, ROV, BRUV

Technique	Number of systems	Boat costs per day	Possible replicates per day	Replicates for this study	Image analysis time per sample (h)	Salary per hour of analysis	Time for analysis (Total)	Cost of analysis	Total cost
DOV	1	$500.00	36	36	0.5	$50.00	18	$900.00	$1,400.00
STV	1	$500.00	70	36	0.5	$50.00	18	$900.00	$1,400.00
ROV	1	$500.00	70	36	0.5	$50.00	18	$900.00	$1,400.00
BRUV	10	$1,500.00	30	36	2.5	$50.00	90	$4,500.00	$6,000.00

Based on a temperate reef setting (Boat time and image analysis only). DOV replicates based on 2 divers.

Table App. 2.3 Cost Analysis for *Coris auricularis* with techniques DOV

Technique	Boat cost per day	No. of samples required	Analysis time per sample (h)	Days on the boat needed	Cost of boat (Total)	Time for analysis (Total)	Cost of analysis	Total Cost
DOV	$500.00	>100	0.5	3	$1,500.00	50	2500	$4,000.00
BRUV	$1,500.00	13	2.5	1	$1,500.00	32.5	1625	$3,125.00

Boat time and image analysis only. DOV replicates based on 2 divers.

1.3 Appendix – Chapter 3

Table App. 3.1 Presence/absence of species recorded by stereo-BRUVs and stereo-ROV across habitat. P = pipeline, R = reef, SS = soft sediment.

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
Acanthuridae	*Acanthurus grammoptilus*	Ring-tailed surgeonfish	X	X		X	X	
	Acanthurus triostegus	Convict Surgeonfish					X	
	Ctenochaetus striatus	Lined Bristletooth		X			X	
	Naso annulatus	Ringtail unicornfish		X				
	Naso brachycentron	Humpback Unicornfish					X	
	Naso brevirostris	Spotted Unicornfish		X			X	
	Naso caesius	Silverblotched Unicornfish	X					
	Naso fageni	Horseface unicornfish	X	X				
	Naso unicornis	Bluespine Unicornfish		X			X	
	Zebrasoma scopas	Brown Tang					X	
Apogonidae	*Apogonidae spp*	Cardinal fish				X		
	Ostorhinchus angustatus	Broadstripe Cardinalfish				X		
	Ostorhinchus cavitiensis	Whiteline Cardinalfish	X		X			
	Ostorhinchus semilineatus	Blacktip Cardinalfish					X	
Ariidae	*Netuma thalassina*	Giant sea catfish			X			
Balistidae	*Abalistes stellatus*	Starry triggerfish			X	X		
	Pseudobalistes fuscus	Yellowspotted triggerfish	X					
	Sufflamen chrysopterum	Halfmoon triggerfish					X	
Batrachoididae	*Halophryne diemensis*	Banded Frogfish	X					
Blenniidae	*Aspidontus taeniatus*	False Cleanerfish		X			X	

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Ecsenius bicolor	Bicolor combtooth blenny					X	
	Meiacanthus grammistes	Linespot Fangblenny		X			X	
	Petroscirtes breviceps	Shorthead Sabretooth Blenny					X	
	Plagiotremus tapeinosoma	Piano fangblenny	X	X				
Bothidae	*Bothidae spp*	Lefteye flounders			X			
Caesionidae	*Caesio caerulaurea*	Goldband Fusilier		X				
	Caesio cuning	Yellowtail fusilier		X			X	
	Pterocaesio chrysozona	Yellowband fusilier	X	X		X	X	
	Pterocaesio digramma	Doubleline fusilier		X				
	Pterocaesio lativittata	Wide-band Fusilier				X		
	Pterocaesio marri	Bigtail Fusilier		X				
	Pterocaesio tile	Neon Fusilier		X			X	
Carangidae	*Alectis ciliaris*	Pennantfish		X				
	Alepes apercna	Smallmouth Scad			X			
	Atule mate	Yellowtail scad	X	X	X	X		
	Carangoides caeruleopinnatus	Onion trevally			X			
	Carangoides ferdau	Blue Trevally		X				
	Carangoides fulvoguttatus	Gold-spotted trevally	X	X	X	X	X	
	Carangoides gymnostethus	Bludger trevally	X		X			
	Carangoides hedlandensis	Bunmpnose trevally	X		X			
	Gnathanodon speciosus	Golden trevally	X	X	X	X		
	Scomberoides commersonnianus	Giant queenfish	X	X	X			
	Scomberoides lysan	Lesser queenfish	X			X		
	Selaroides leptolepis	Yellowstripe Scad			X	X		X

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Seriolina nigrofasciata	Blackbanded amberjack	X		X			
Carcharhinidae	*Carcharhinus limbatus*	Blacktip Shark			X			
	Carcharhinus melanopterus	Blacktip Reef Shark		X	X			
	Carcharhinus plumbeus	Sandbar Shark	X		X			
	Carcharhinus sorrah	Spot-tail Shark	X		X			
	Galeocerdo cuvier	Tiger Shark	X					
	Loxodon macrorhinus	Sliteye Shark	X		X			
	Negaprion acutidens	Lemon Shark	X					
	Rhizoprionodon acutus	Milk Shark	X		X			
	Triaenodon obesus	Whitetip Reef Shark					X	
Chaetodontidae	*Chaetodon adiergastos*	Philippine Butterflyfish					X	
	Chaetodon assarius	Western butterflyfish				X		
	Chaetodon aureofasciatus	Goldstripe butterflyfish		X			X	
	Chaetodon auriga	Threadfin butterflyfish	X	X		X	X	
	Chaetodon citrinellus	Citron Butterflyfish					X	
	Chaetodon lineolatus	Lined Butterflyfish		X				
	Chaetodon lunula	Racoon Butterflyfish		X			X	
	Chaetodon plebeius	Bluespot butterflyfish		X			X	
	Chaetodon speculum	Ovalspot Butterflyfish		X				
	Chaetodon trifascialis	Chevron Butterflyfish					X	
	Chelmon marginalis	Margined coralfish	X	X	X	X	X	
	Coradion chrysozonus	Orangebaned coralfish	X		X	X		X
	Heniochus acuminatus	Longfin bannerfish		X		X	X	
	Parachaetodon ocellatus	Ocellate butteflyfish	X		X			X
Chanidae	*Chanos chanos*	Milkfish			X			

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
Cheilodactylidae	*Cheilodactylus gibbosus*	Magpie Morwong					X	
Cirrhitidae	*Cirrhitichthys aprinus*	Blotched Hawkfish				X		
	Cirrhitichthys oxycephalus	Spotted Hawkfish	X					
Dasyatidae	*Bathytoshia lata*	Black Stingray			X			
	Maculabatis toshi	Brown Whipray			X			
	Neotrygon australiae	Bluespotted Maskray	X				X	X
	Pastinachus ater	Cowtail Stingray			X			
	Pateobatis jenkinsii	Jenkins' Whipray						X
Echeneidae	*Echeneis naucrates*	Sharksucker	X	X	X		X	
Ephippidae	*Platax batavianus*	Humphead batfish	X	X	X			X
Fistulariidae	*Fistularia commersonii*	Smooth flutemouth		X	X			
Gerreidae	*Gerres oyena*	Blacktip Silverbiddy			X			
Ginglymostomatidae	*Nebrius ferrugineus*	Tawny nurse shark		X			X	
Glaucosomatidae	*Glaucosoma magnificum*	Threadfin pearl perch			X	X		
Gobiidae	*Gobiidae spp*	Goby fish		X			X	X
	Valenciennea muralis	Mural Glidergoby	X					
	Yongeichthys nebulosus	Hairfin Goby						X
Grammistidae	*Diploprion bifasciatum*	Barred soapfish		X		X		
Haemulidae	*Diagramma pictum labiosum*	Painted Sweetlips	X	X	X	X	X	
	Plectorhinchus multivittatus	Manyline Sweetlips		X			X	
Hemigaleidae	*Hemigaleus australiensis*	Weasel Shark	X		X			
Hemiscylliidae	*Chiloscyllium punctatum*	Grey Carpetshark	X					
Holocentridae	*Sargocentron rubrum*	Red Squirrelfish				X		
Kyphosidae	*Kyphosus bigibbus*	Grey drummer		X				
	Kyphosus vaigiensis	Brassy Drummer		X				

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
Labridae	*Anampses caeruleopunctatus*	Diamond Wrasse					X	
	Anampses geographicus	Scribbled Wrasse		X			X	
	Anampses lennardi	Blue and yellow wrasse	X	X	X	X	X	
	Anampses melanurus	Blacktail Wrasse					X	
	Anampses meleagrides	Spotted wrasse		X			X	
	Bodianus axillaris	Coral Pigfish	X	X				
	Cheilinus chlorourus	Floral wrasse		X			X	
	Cheilio inermis	Sharpnose wrasse	X	X		X	X	
	Choerodon cauteroma	Bluespotted tuskfish	X	X	X	X	X	X
	Choerodon cephalotes	Purple tuskfish	X		X	X		X
	Choerodon cyanodus	Blue tuskfish	X	X	X	X	X	X
	Choerodon schoenleinii	Blackspot Tuskfish	X	X	X	X	X	X
	Choerodon vitta	Redstripe tuskfish	X	X	X	X	X	X
	Coris aygula	Redblotched Wrasse		X			X	
	Coris caudimacula	Spot-tail wrasse	X	X		X	X	
	Coris pictoides	Pixy wrasse	X	X		X	X	
	Epibulus insidiator	Slingjaw Wrasse		X			X	
	Gomphosus varius	Birdnose Wrasse		X			X	
	Halichoeres margaritaceus	Pearly Wrasse		X		X		
	Halichoeres melanochir	Orangefin Wrasse		X		X	X	
	Halichoeres nebulosus	Cloud wrasse		X			X	
	Halichoeres prosopeion	Twotone Wrasse					X	
	Hemigymnus fasciatus	Fiveband Wrasse		X			X	
	Hemigymnus melapterus	Thicklip Wrasse		X			X	
	Hologymnosus annulatus	Ring wrasse		X			X	

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Hologymnosus doliatus	Pastel Slender Wrasse	X				X	
	Labroides dimidiatus	Common cleanerfish	X	X		X	X	X
	Leptojulis cyanopleura	Shoulderspot wrasse		X		X	X	
	Macropharyngodon negrosensis	Black leopard wrasse				X		
	Oxycheilinus orientalis	Oriental Maori Wrasse	X					
	Oxycheilinus unifasciatus	Ringtail Maori Wrasse		X				
	Pseudodax moluccanus	Chiseltooth Wrasse					X	
	Stethojulis bandanensis	Redspot wrasse		X		X	X	
	Stethojulis interrupta	Brokenline wrasse				X		
	Stethojulis strigiventer	Threeline wrasse		X			X	
	Thalassoma amblycephalum	Bluehead Wrasse	X			X		
	Thalassoma hardwicke	Sixbar Wrasse		X			X	
	Thalassoma lunare	Moon wrasse	X	X		X	X	
	Thalassoma lutescens	Green Moon Wrasse		X			X	
Leiognathidae	*Aurigequula longispina*	Longspine Ponyfish			X			X
Lethrinidae	*Gymnocranius grandoculis*	Robinson’s sea bream		X	X			
	Lethrinus atkinsoni	Yellow-tailed emperor		X		X	X	
	Lethrinus genivittatus	Longspine emperor	X	X	X	X		
	Lethrinus laticaudis	Grass emperor	X	X	X	X	X	X
	Lethrinus lentjan	Pink ear emperor	X	X		X	X	
	Lethrinus microdon	Smalltooth emperor	X	X				
	Lethrinus nebulosus	Spangled emperor	X	X	X	X	X	
	Lethrinus olivaceus	Longnose Emperor		X	X			
	Lethrinus punctulatus	Bluespotted emperor	X	X	X	X	X	X
	Lethrinus ravus	Drab emperor			X			

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Lethrinus variegatus	Variegated emperor	X	X	X			
	Monotaxis grandoculis	Big-eye bream					X	
Lutjanidae	*Lutjanus carponotatus*	Spanish flag snapper	X	X	X	X	X	X
	Lutjanus fulviflamma	Blackspot Snapper		X			X	
	Lutjanus lemniscatus	Darktail snapper	X	X			X	
	Lutjanus lutjanus	Big eye snapper						X
	Lutjanus monostigma	Onespot Snapper					X	
	Lutjanus quinquelineatus	Five-lined snapper		X		X	X	
	Lutjanus rivulatus	Maori Snapper		X				
	Lutjanus russellii	Moses' snapper		X				
	Lutjanus sebae	Red Emperor		X	X	X		
	Lutjanus vitta	Brownstripe snapper	X		X	X		
	Symphorus nematophorus	Chinamanfish	X	X	X	X		
Malacanthidae	*Hoplolatilus cuniculus*	Green Tilefish						X
Microdesmidae	*Ptereleotris evides*	Arrow Dartgoby					X	
	Ptereleotris microlepis	Greeneye Dartgoby						X
Monacanthidae	*Aluterus scriptus*	Scrawled Leatherjacket	X			X		
	Anacanthus barbatus	Bearded leatherjacket			X			
	Chaetodermis penicilligerus	Tasselled Leatherjacket			X			
	Monacanthus chinensis	Fanbelly leatherjacket	X		X			X
	Paramonacanthus choirocephalus	Pigface leatherjacket	X		X	X		X
	Pseudomonacanthus peroni	Potbelly Leatherjacket			X			
	Thamnaconus modestoides	Modest Leatherjacket			X			
Mullidae	*Mulloidichthys vanicolensis*	Goldstripe Goatfish				X		
	Parupeneus barberinoides	Bicolor goatfish		X	X	X	X	

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Parupeneus chrysopleuron	Rosy goatfish	X					
	Parupeneus cyclostomus	Goldsaddle goatfish		X				
	Parupeneus heptacanthus	Cinnabar goatfish	X	X	X	X		X
	Parupeneus indicus	Yellowspot goatfish	X	X	X	X	X	X
	Parupeneus spilurus	Blackspot goatfish	X	X	X	X	X	
	Upeneus tragula	Bartail goatfish	X		X	X		X
Muraenidae	*Gymnothorax cribroris*	Sieve Moray			X			
	Gymnothorax eurostus	Stout Moray		X				
	Gymnothorax javanicus	giant moray		X				
	Gymnothorax thrysoideus	Grayface moray					X	
	Gymnothorax undulatus	Undulate Moray	X	X	X			
Nemipteridae	*Nemipterus furcosus*	Rosy Threadfin Bream	X		X			
	Pentapodus emeryii	Double whiptail	X	X	X		X	
	Pentapodus porosus	Northwest whiptail	X	X	X	X	X	X
	Pentapodus vitta	Black striped butterfish	X	X	X	X		X
	Scaevius milii	Coral monocle bream	X	X	X			
	Scolopsis affinis	Bridled Monocle Bream	X	X	X		X	
	Scolopsis bilineata	Two-line Monocle Bream		X			X	
	Scolopsis monogramma	Rainbow monocle bream	X	X	X	X	X	X
Ostraciidae	*Ostracion cubicus*	Yellow boxfish	X	X	X	X	X	
	Ostracion meleagris	Black Boxfish		X				
Paralichthyidae	*Pseudorhombus spp*	Flounder fish			X			
Pempherididae	*Pempheris analis*	Bronze Bullseye					X	
	Pempheris oualensis	Oualan Bullseye				X		
	Pempheris schwenkii	Silver Bullseye					X	

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
Pinguipedidae	*Parapercis nebulosa*	Pinkbanded grubfish	X		X			X
	Parapercis spp	Grubfish		X				
Platycephalidae	*Platycephalus endrachtensis*	Northern Sand Flathead			X			
Plotosidae	*Paraplotosus butleri*	Sailfin catfish	X	X		X	X	
Pomacanthidae	*Centropyge tibicen*	Keyhole Angelfish					X	
	Chaetodontoplus duboulayi	Scribbled angelfish	X	X	X	X		X
	Chaetodontoplus personifer	Yellowtail angelfish			X			
	Pomacanthus imperator	Emperor angelfish				X		
	Pomacanthus semicirculatus	Semicircle angelfish	X	X		X	X	
	Pomacanthus sexstriatus	Sixbar angelfish	X	X	X	X	X	
Pomacentridae	*Abudefduf bengalensis*	Bengal sergeant	X	X		X	X	
	Abudefduf septemfasciatus	Banded Sergeant					X	
	Abudefduf sexfasciatus	Scissortail Sergeant		X			X	
	Acanthochromis polyacanthus	Spiny Puller	X		X			
	Amphiprion clarkii	Clark's anemonefish				X		
	Chromis fumea	Smokey puller	X		X	X		X
	Chromis viridis	Blue-green Puller		X			X	
	Chromis weberi	Weber's Puller					X	
	Dascyllus aruanus	Banded Humbug					X	
	Dascyllus reticulatus	Headband humbug		X		X		
	Dascyllus trimaculatus	Threespot humbug				X	X	
	Hemiglyphidodon plagiometopon	Lagoon Damsel		X				
	Neoglyphidodon melas	Black Damsel					X	
	Neoglyphidodon nigroris	Scarface Damsel		X			X	
	Neopomacentrus azysron	Yellowtail Demoiselle		X			X	

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Neopomacentrus cyanomos	Regal Demoiselle		X				
	Neopomacentrus aktites	Western Australian Demoiselle	X	X	X	X	X	X
	Pomacentrus amboinensis	Ambon Damsel					X	
	Pomacentrus coelestis	Neon damsel	X	X		X	X	
	Pomacentrus limosus	Muddy Damsel	X	X		X	X	X
	Pomacentrus milleri	Miller’s damsel	X	X		X	X	
	Pomacentrus moluccensis	Lemon Damsel		X			X	
	Pomacentrus nagasakiensis	Blue-scribbled damsel		X		X		
	Pomacentrus vaiuli	Princess damsel	X	X		X		
	Stegastes fasciolatus	Pacific Gregory					X	
	Stegastes nigricans	Dusky Gregory					X	
	Stegastes obreptus	Western Gregory		X			X	
Pseudochromidae	*Labracinus lineatus*	Lined Dottyback		X				
Pteroidae	*Pterois spp*	Lionfish fish	X					
	Pterois volitans	Red lionfish				X		
Rachycentridae	*Rachycentron canadum*	Cobia			X			
Rhinidae	*Rhynchobatus australiae*	Whitespotted Guitarfish	X	X	X			
Scaridae	*Chlorurus microrhinos*	Steephead parrotfish					X	
	Chlorurus sordidus	Bullethead Parrotfish		X			X	
	Hipposcarus harid	Candelamoa parrotfish		X				
	Scarus forsteni	Whitespot Parrotfish		X			X	
	Scarus frenatus	Sixband Parrotfish		X				
	Scarus ghobban	Blue-barred parrotfish	X	X		X	X	
	Scarus oviceps	Darkcap Parrotfish	X					
	Scarus prasiognathos	Greencheek Parrotfish		X			X	

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Scarus psittacus	Palenose Parrotfish		X			X	
	Scarus rivulatus	Surf Parrotfish		X			X	
	Scarus schlegeli	Schlegel's Parrotfish		X		X	X	
Scombridae	*Scomberomorus spp*	Mackerel	X	X	X	X		
Serranidae	*Cephalopholis boenak*	Brown-banded rockcod	X			X	X	
	Cephalopholis miniata	Coral Rockcod		X			X	
	Epinephelus bilobatus	Frostback rockcod	X	X	X	X	X	
	Epinephelus coioides	Goldspotted Rockcod	X	X	X	X	X	
	Epinephelus fasciatus	Blacktip grouper		X		X	X	
	Epinephelus malabaricus	Blackspotted Rockcod		X				
	Epinephelus merra	Birdwire Rockcod				X		
	Epinephelus multinotatus	Rankin cod			X	X		
	Epinephelus polyphekadion	Camouflage Grouper		X				
	Epinephelus quoyanus	Longfin rockcod	X	X		X		
	Epinephelus rivulatus	Halfmoon grouper	X	X	X		X	
	Plectropomus spp	Coral trout	X	X	X	X	X	
Siganidae	*Siganus argenteus*	Forktail rabbitfish				X		
	Siganus canaliculatus	White-spotted spinefoot	X	X	X	X	X	
	Siganus doliatus	Barred rabbitfish	X	X			X	
	Siganus fuscescens	Black rabbitfish	X	X	X	X		X
	Siganus punctatus	Spotted Rabbitfish					X	
	Siganus trispilos	Threespot Rabbitfish					X	
Sillaginidae	*Sillago analis*	Goldenline Whiting			X			
	Sillago sihama	Northern Whiting			X			
Sphyraenidae	*Sphyraena jello*	Pickhandle barracuda	X	X		X		

Family	Taxa	Common name	Stereo-BRUVs			Stereo-ROV		
			P	R	SS	P	R	SS
	Sphyraena obtusata	Yellowtail barracuda		X	X		X	
	Sphyraena qenie	Blackfin barracuda		X				
Synodontidae	*Saurida undosquamis*	Brushtooth lizardfish			X			
	Synodus variegatus	Variegated Lizardfish					X	
Terapontidae	*Pelates quadrilineatus*	Fourline Striped Grunter			X			
Tetraodontidae	*Arothron hispidus*	Stars-and-stripes Puffer		X		X		
	Arothron immaculatus	Yellow-eye Puffer		X				
	Arothron manilensis	Narrowlined Puffer					X	
	Arothron stellatus	Starry Puffer					X	
	Canthigaster valentini	Blacksaddle toby	X			X		
	Lagocephalus lunaris	Lunartail puffer	X		X			
	Lagocephalus sceleratus	Northwest Blowie	X		X			
	Torquigener pallimaculatus	Rusty-spotted toadfish			X			
	Torquigener whitleyi	Whitley’s Toadfish					X	
Triakidae	*Hemitriakis falcata*	Sicklefin Houndshark	X					
	Mustelus ravidus	Australian grey smooth-hound	X					
Zanclidae	*Zanclus cornutus*	Moorish Idol		X			X	

Table App. 3.2 Total number of species recorded by stereo-BRUVs and stereo-ROV in soft sediment habitat, showing the feeding guild and common habitat association of each species.

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
Abalistes stellatus	Invertivore	Sandy, muddy and silty substrate	3		Kuiter and Tonozuka, 2001; Randall, 1995
Acanthochromis polyacanthus	Planktivore	Coral reefs/outer reef slopes	2		Allen, 1991; Lieske and Myers, 1994
Alepes apercna	Invertivore	Inshore waters	16		Hoese et al., 2006; Smith-Vaniz et al., 1999
Anacanthus barbatus	-	Sandy and weedy areas of reefs/muddy substrate with sponges	2		Lieske and Myers, 1994
Anampses lennardi	Invertivore	Silty reefs	3		Bray, 2018; Lieske and Myers, 1994; Nguyen and Nguyen, 2006
Atule mate	Invertivore	Inshore waters	63		Allen and Erdmann, 2012; Fischer et al., 1990; Mundy, 2005; Smith-Vaniz, 1999
Aurigequula longispina	Generalist Carnivores	Inshore waters	538	28	Bray, 2018; Fischer et al., 1990
Bathytoshia lata	Generalist Carnivores	Sandy substrate, sometimes near coral reefs	1		Last et al., 2016; Michael, 1993
Bothidae spp	Generalist Carnivores	Sandy and muddy substrate	2		Nelson, 1994
Carangoides caeruleopinnatus	Generalist Carnivores	Sandy substrate near reef	2		Allen and Erdmann, 2012; Salini et al., 1994
Carangoides fulvoguttatus	Generalist Carnivores	Rocky and coral reefs/offshore banks	2		Fischer et al., 1990; Sommer et al., 1996
Carangoides gymnostethus	Generalist Carnivores	Offshore reefs	2		Fischer et al., 1990; Kuiter and Tonozuka, 2001; Lieske and Myers, 1994
Carangoides hedlandensis	Generalist Carnivores	Inshore waters	1		Fischer et al., 1990; Paxton et al., 1989
Carcharhinus limbatus	Generalist Carnivores	Inshore/offshore waters	3		Compagno, 1984; Myers, 1999
Carcharhinus melanopterus	Generalist Carnivores	Coral reefs/reef flats	1		Compagno, 1984; Last and Stevens, 1994; Mundy, 2005
Carcharhinus plumbeus	Generalist Carnivores	Inshore/offshore waters	3		Compagno, 1984; Compagno et al., 1989

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
Carcharhinus sorrah	Generalist Carnivores	Inshore waters near reefs	8		Carpenter et al., 1997; Compagno, 1984; Last and Stevens, 1994
Chaetodermis penicilligerus	Invertivore	Weedy areas of coastal reefs/trawling grounds	2		Lieske and Myers, 1994; Nelson, 1994
Chaetodontoplus duboulayi	Invertivore	Coastal reefs /sandy and rubble substrate with reef outcrops, sponges and seawhips	8	5	Masuda and Allen, 1993; Randall et al., 1990
Chaetodontoplus personifer	Invertivore	Sandy or rubble substrate with coral, sponges and seawhips	1		Allen and Swainston, 1988; Masuda and Allen, 1993
Chanos chanos	Omnivore	Offshore waters	1		Allen et al., 2002; Hiatt, 1947; Schuster, 1960; Seegers et al., 2003
Chelmon marginalis	Invertivore	Coastal coral and rocky reefs	11		Anderson et al., 1981; Lieske and Myers, 1994
Choerodon cauteroma	Invertivore	Sandy and weedy areas near coral reefs	26	9	Allen, 1997; Randall et al., 1990
Choerodon cephalotes	Invertivore	Coral reefs/seagrass beds	31	8	Westneat, 2001; Randall et al., 1990
Choerodon cyanodus	Invertivore	Sandy and rubble substrate of reefs flats	3	1	Lieske and Myers, 1994; Randall et al., 1990
Choerodon schoenleinii	Invertivore	Sandy or weedy areas near reefs	2	3	Lieske and Myers, 1994; Randall et al., 1990
Choerodon vitta	Invertivore	Sandy or weedy areas	13	11	Allen and Swainston, 1988; Randall, 1990
Chromis fumea	Planktivore	Coral and rocky reefs	13	112	Allen, 1991; Jan, 1997; Lieske and Myers, 1994; Wantiez, 1994
Coradion chrysozonus	Invertivore	Reef drop offs with rich invertebrates/sparse corals	3	1	Lieske and Myers, 1994; Myers, 1991
Diagramma pictum labiosum	Generalist Carnivores	Sandy and muddy substrate, around rock outcrops	4		Allen and Erdmann, 2012; Kuiter and Tonozuka, 2001; Sommer et al., 1996
Echeneis naucrates	Generalist Carnivores	Inshore waters	5		Cervigón et al., 1992; Smith, 1997
Epinephelus bilobatus	Generalist Carnivores	Coral reefs/rocky bottoms/scattered coral patches	7		Craig et al., 2011; Kulbicki et al. 2005
Epinephelus coioides	Generalist Carnivores	Coastal reefs over muddy and rubble substrate	4		Kailola et al., 1993; Kulbicki et al., 2005; Parrish, 1987

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
Epinephelus multinotatus	Generalist Carnivores	Coral reefs/inshore/offshore waters	3		Heemstra and Randall, 1993; Kailola et al., 1993; Ramm et al., 1991; Randall and Heemstra, 1991
Epinephelus rivulatus	Generalist Carnivores	Coral reefs/algae flats/seagrass beds	1		Craig et al., 2011; Sommer et al., 1996
Fistularia commersonii	Generalist Carnivores	Coastal reefs/sandy substrate near reefs	2		Golani, 2000; Watson and Sandknop, 1996
Gerres oyena	Invertivore	Sandy substrate near reefs	12		Allen and Erdmann, 2012; Hajisamae et al., 2004; Masuda and Allen, 1993
Glaucosoma magnificum	Invertivore	Trawling grounds/near reefs	6		McKay, 1997
Gnathanodon speciosus	Generalist Carnivores	Coastal reefs	19		Allen and Erdmann, 2012; Lieske and Myers, 1994
Gobiidae spp	Invertivore	Coral reefs/sandy and rubble substrate		2	Nelson, 1994; Swainston, 2010
Gymnocranius grandoculis	Generalist Carnivores	Offshore reefs/trawling grounds	2		Kuiter and Tonozuka, 2001; Sommer et al., 1996
Gymnothorax cribroris	Generalist Carnivores	Coral reef amoong crevies	1		Nguyen and Nguyen, 2006; Swainston, 2010
Gymnothorax undulatus	Generalist Carnivores	Reef flats among rocks and rubble	1		Mundy, 2005; Thollot, 1996
Hemigaleus australiensis	Invertivore	Inshore/offshore waters	7		Hoese et al., 2006; Salini et al., 1992; Taylor and Bennet, 2008
Hoplolatilus cuniculus	Invertivore	Sandy, muddy and rubble substrate		3	Dooley, 1978; Randall and Dooley, 1974
Labroides dimidiatus	Invertivore	Coral reefs		4	Kuiter and Tonozuka, 2001; Myers, 1991; Westneat, 2001
Lagocephalus lunaris	Invertivore	Sandy substrate on coastal reefs	11		Carpenter et al., 1997
Lagocephalus sceleratus	Invertivore	Offshore reefs/sandy substrate	5		Kapiris et al., 2014; Kulbicki et al., 2005; May and Maxwell, 1986; Mohsin et al., 1986
Lethrinus genivittatus	Generalist Carnivores	Outer reef slopes/seagrass beds/sandy substrate	67		Broad, 2003; Carpenter and Allen, 1989; Nguyen and Nguyen, 2006
Lethrinus laticaudis	Generalist Carnivores	Coral reefs/seagrass/sandy substrate	35	5	Carpenter and Allen, 1989; Nguyen and Nguyen, 2006
Lethrinus nebulosus	Generalist Carnivores	Coral reefs/seagrass/sandy substrate	2		Broad, G., 2003; Carpenter and Allen, 1989; Gell and Whittington,

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
					2002; Nguyen and Nguyen, 2006; Walker, 1978
Lethrinus olivaceus	Generalist Carnivores	Sandy substrate/reef slopes	1		Carpenter and Allen, 1989; Sommer et al., 1996; Allen, 2009
Lethrinus punctulatus	Generalist Carnivores	Macroalgae areas/sandy substrates/seagrass beds	122	7	Evans et al., 2014; Allen, 2009
Lethrinus ravus	Generalist Carnivores	On or near reefs	8		Carpenter and Randall, 2003; Hoese et al., 2006
Lethrinus variegatus	Invertivore	Sandy and weedy areas near coral reef	5		Allen and Erdmann, 2012; Sommer et al., 1996
Loxodon macrorhinus	Generalist Carnivores	Inshore/offshore waters	1		Compagno, 1984; Last and Stevens, 1994
Lutjanus carponotatus	Generalist Carnivores	Coral reefs/outer reef slopes	3	1	Allen and Erdmann, 2012; Connell, 1998
Lutjanus lutjanus	Generalist Carnivores	Offshore reefs/trawling grounds		230	Hoese et al., 2006; Kuiter and Tonozuka, 2001; Sommer et al., 1996
Lutjanus sebae	Generalist Carnivores	Around coral/rocky reefs/sandy substrate	4		Allen, 1985; Anderson, 1986 ; Salini et al., 1994
Lutjanus vitta	Generalist Carnivores	Coral reefs/sandy substrate with coral outcrops, sponges and seawhips	7		Allen, 1985; Hoese et al., 2006
Maculabatis toshi	Generalist Carnivores	Sandy substrate	1		Last and Compagno, 1999; White et al., 2006
Monacanthus chinensis	Omnivore	Coastal reefs/offshore reefs/weedy areas/muddy or sility substrate	10	1	Conacher et al., 1979; Kuiter and Tonozuka, 2001; May and Maxwell, 1986
Nemipterus furcosus	Generalist Carnivores	Sandy and muddy substrate	25		Kuiter and Tonozuka, 2001; Russell, 1990
Neopomacentrus aktites	Planktivore	Coral and rocky reef with crevices	1	5	Allen et al., 2017
Neotrygon australiae	Generalist Carnivores	Near rocky and coral reefs in inshore waters/sandy substrate		1	Last et al., 2016; Swainston, 2010
Netuma thalassina	Generalist Carnivores	Inshore waters	1		Hoese et al., 2006; Rainboth, 1996
Ostorhinchus cavitiensis	Invertivore	Inshore waters	1		Hoese et al., 2006

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
Ostracion cubicus	Omnivore	Coastal reefs	1		Cornic, 1987; Letourneur et al., 2004
Parachaetodon ocellatus	Invertivore	Sandy and silty substrate of coastal reefs/open muddy areas	2	1	Allen, 2006; Kuiter and Tonozuka, 2001
Paramonacanthus choirocephalus	Invertivore	Coastal reefs/Sandy and muddy substrate	18	1	Allen and Erdmann, 2012; Kuiter and Tonozuka, 2001
Parapercis nebulosa	Invertivore	Sandy, silty and rubble substrate	37	5	Allen and Swainston, 1988; Randall, 2001
Parupeneus barberinoides	Invertivore	Sandy and rubble substrate near reefs	1		Randall, 2004
Parupeneus heptacanthus	Invertivore	Sandy, muddy and rubble substrate/coastal reefs/seagrass	63	7	Letourneur et al., 2004; Nguyen and Nguyen, 2006
Parupeneus indicus	Generalist Carnivores	Sandy and silty substrate of coral reefs	8	3	Jenkins, 2019; Lieske and Myers, 1994; Randall, 2004
Parupeneus spilurus	Invertivore	Coastal reefs	4		Jenkins, 2019; Kuiter, 1993
Pastinachus ater	Generalist Carnivores	Sandy and sility substrate/reef flats	1		Semeniuk et al., 2005
Pateobatis jenkinsii	Generalist Carnivores	Sandy substrate		1	Hoese et al., 2006; Last and Compagno, 1999
Pelates quadrilineatus	Generalist Carnivores	Inshore waters/seagrass beds	13		Kuiter and Tonozuka, 2001; Paxton et al., 1989
Pentapodus emeryii	Generalist Carnivores	Silty costal reefs	1		Broad, 2003; Russell, 1990
Pentapodus porosus	Generalist Carnivores	Near reefs in offshore waters	314	47	Russell, 1990, 2001
Pentapodus vitta	Generalist Carnivores	Seagrass beds/reefs	4	4	Lieske and Myers, 1994; Russell, 1990
Platax batavianus	Generalist Carnivores	Open substrate with sparse reef	3	1	Lieske and Myers, 1994
Platycephalus endrachtensis	Generalist Carnivores	Sandy substrate	4		Allen and Swainston, 1988; Salini et al., 1998
Plectropomus spp	Piscivores	Coral reefs	2		Kailola et al., 1993; Kuiter and Tonozuka, 2001
Pomacanthus sexstriatus	Omnivore	Inshore/offshore reefs	3		Lieske and Myers, 1994; Jekins, 2019

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
Pomacentrus limosus	Omnivore	Rocky outcrops with sandy and silty substrate		2	Allen, 1992; Evans et al., 2014
Pseudomonacanthus peroni	-	Trawling grounds/rocky reefs	4		Allen, 1997; Randall et al., 1997
Pseudorhombus spp	Generalist Carnivores	Sandy and muddy substrate	1		Kuiter and Tonozuka, 2001; Lin et al., 1999
Ptereleotris microlepis	Planktivore	Costal reefs over sandy and rubble substrate		3	Bacchet et al., 2006; Lieske and Myers, 1994
Rachycentron canadum	Generalist Carnivores	Coral reefs/mud and gravel bottoms	1		Collette, 1999; Fisher et al., 1990; Nguyen and Nguyen, 2006
Rhizoprionodon acutus	Generalist Carnivores	Sandy substrate	5		Compagno, 1984; Salini et al., 1994
Rhynchobatus australiae	Generalist Carnivores	Sandy substrate near coral reefs	5		Compagno and Last, 1999; Last et al., 2016
Saurida undosquamis	Generalist Carnivores	Sandy and mudy substrate	5		Heemstra, 1995; Fischer et al., 1990; Yamada et al., 1995
Scaevius milii	Generalist Carnivores	Sandy and muddy substrate near reefs	1		Hoese et al., 2006; Russell, 1990
Scolopsis affinis	Generalist Carnivores	Sandy and muddy substrate near reefs	12		Jenkins, 2019; Kuiter and Tonozuka, 2001
Scolopsis monogramma	Generalist Carnivores	Sandy and silty substrates near reefs	9	3	Kuiter and Tonozuka, 2001; Russell, 1990
Scomberoides commersonnianus	Generalist Carnivores	Coastal waters near reefs	5		Allen and Erdmann, 2012; Fischer et al., 1990; Kuiter and Tonozuka, 2001;Sommer et al., 1996
Scomberomorus spp	Generalist Carnivores	Pelagic/near and over reefs	19		Kuiter and Tonozuka, 2001
Selaroides leptolepis	Generalist Carnivores	Inshore waters	142	74	Paxton et al., 1989; Yamashita et al., 1987
Seriolina nigrofasciata	Generalist Carnivores	Offshore reefs	12		Kuiter and Tonozuka, 2001; Sommer et al., 1996; Randall, 1995; Yamada et al., 1995
Siganus canaliculatus	Herbivore	Inshore algae reefs/seagrass flats	17		Kuiter and Tonozuka, 2001; Soh, 1976; Woodland, 1990
Siganus fuscescens	Herbivore	Algae and seagrass flats/coastal reefs	85	2	Lieske and Myers, 1994; Yamada et al., 1995
Sillago analis	Invertivore	Silty and sandy substrate	6		Jenkins, 2019

Taxa	Feeding guild	Habitat	Stereo-BRUVs	Stereo-ROV	References
Sillago sihama	Invertivore	Sandy substrate	1		Allen et al., 2002; Chacko, 1949; McKay, 1992
Sphyraena obtusata	Piscivores	Rocky reefs/seagrass beds/sandy and weedy areas	14		Blaber et al., 1990, May and Maxwell, 1986; Senou, 2001
Symphorus nematophorus	Piscivores	Coral reefs/outer reef and slopes/sandy substrate near reef	1		Allen, 1985; Jenkins, 2019; Nguyen and Nguyen, 2006;
Thamnaconus modestoides	-	Trawling grounds	1		Hutchins, 1986
Torquigener pallimaculatus	Invertivore	Sandy and muddy substrate	16		Kuiter and Tonozuka, 2001
Upeneus tragula	Invertivore	Sandy substrate	82	16	Carpenter et al., 1997; Kulbicki et al., 2005; Jenkins, 2019
Yongeichthys nebulosus	Invertivore	Silty and muddy substrate near reefs		1	Broad, 2003

1.4 Appendix – Chapter 4

Table App. 4.1 Feeding guild of taxa

Taxa	Feeding guild	Reference
Abalistes stellatus	Invertivore	Jenkins, 2019; Kuiter and Tonozuka, 2001; Randall, 1995
Abudefduf bengalensis	Omnivore	Lieske and Myers, 1994
Abudefduf septemfasciatus	Omnivore	Hiatt and Strasburg, 1960, Lieske and Myers, 1994
Abudefduf sexfasciatus	Omnivore	Sano et al., 1984
Acanthurus grammoptilus	Herbivore	Randall, 2001
Acanthurus triostegus	Herbivore	Dominici-Arosemena and Wolff, 2006
Aluterus scriptus	Omnivore	Dominici-Arosemena and Wolff, 2006, 2005; Randall, 1967
Amphiprion clarkii	Omnivore	Miyagawa, 1989; Moe, 1992; Sano et al., 1984
Anampses caeruleopunctatus	Invertivore	Myers, 1991
Anampses geographicus	Invertivore	Sano et al., 1984
Anampses lennardi	Invertivore	Lieske and Myers, 1994; Nelson, 1994
Anampses melanurus	Invertivore	Lieske and Myers, 1994
Anampses meleagrides	Invertivore	Jenkins, 2019; Nelson, 1994
Apogonidae spp	Invertivore	Nelson, 1994
Arothron hispidus	Omnivore	Dominici-Arosemena and Wolff, 2006; Thollot, 1996
Arothron manilensis	Invertivore	Myers, 1991
Arothron mappa	Omnivore	Myers, 1991
Arothron stellatus	Invertivore	Kulbicki et al., 2005
Aspidontus taeniatus	Generalist Carnivores	Kuiter and Tonozuka, 2001; Myers, 1991
Atule mate	Invertivore	Fischer et al., 1990; Yamashita et al., 1987
Aurigequula longispina	Generalist Carnivores	Blaber, 1980; Fischer et al., 1990; Masuda and Allen, 1993
Blenniidae spp	Omnivore	Nelson, 1994
Caesio caerulaurea	Planktivore	Jenkins, 2019
Caesio cuning	Planktivore	Carpenter, 1987
Canthigaster valentini	Omnivore	Myers, 1991
Carangoides fulvoguttatus	Generalist Carnivores	Fischer et al., 1990; Gell and Whittington, 2002
Centropyge tibicen	Herbivore	Masuda and Allen, 1993; Myers, 1991
Cephalopholis boenak	Generalist Carnivores	Blaber et al., 1990, Craig et al., 2011
Cephalopholis miniata	Generalist Carnivores	Shpigel and Fishelson, 1989
Chaetodon adiergastos	Invertivore	Pyle, 2001
Chaetodon assarius	Omnivore	Steene, 1978
Chaetodon aureofasciatus	Corallivore	Pratchett, 2005
Chaetodon auriga	Omnivore	Myers, 1991
Chaetodon citrinellus	Omnivore	Harmelin-Vivien, 1989; Myers, 1991
Chaetodon lunula	Omnivore	Harmelin-Vivien, 1989; Heemstra, 1986

Taxa	**Feeding guild**	**Reference**
Chaetodon plebeius	Corallivore	Pratchett, 2005; Sano et al., 1984; Steene, 1978
Chaetodon trifascialis	Corallivore	Steene, 1978, Sano et al., 1984
Chaetodontoplus duboulayi	Invertivore	Randall et al., 1990
Cheilinus chlorourus	Invertivore	Sano et al., 1984
Cheilio inermis	Invertivore	Myers, 1999
Cheilodactylus gibbosus	Omnivore	Allen and Heemstra, 1976
Chelmon marginalis	Invertivore	Anderson et al., 1981; Lieske and Myers, 1994
Chlorurus microrhinos	Herbivore	Bacchet et al., 2005
Chlorurus sordidus	Herbivore	Sommer et al., 1996
Choerodon cauteroma	Invertivore	Randall, 1990
Choerodon cephalotes	Invertivore	Randall, 1990
Choerodon cyanodus	Invertivore	Randall, 1990
Choerodon schoenleinii	Invertivore	Randall et al., 1990
Choerodon vitta	Invertivore	Randall, 1990
Chromis fumea	Planktivore	Jan, 1997; Wantiez, 1994
Chromis viridis	Planktivore	Emslie et al., 2019, Sano et al., 1984
Chromis weberi	Planktivore	Emslie et al., 2019
Cirrhitichthys aprinus	Generalist Carnivores	Hiatt and Strasburg, 1960
Coradion chrysozonus	Invertivore	Lieske and Myers, 1994
Coris aygula	Invertivore	Sano et al., 1984; Westneat, 2001
Coris caudimacula	Invertivore	Randall, 1999
Coris pictoides	Invertivore	Jenkins, 2019
Ctenochaetus striatus	Omnivore	Axe, 1990; Sano et al., 1984; Sluka and Miller, 2001
Dascyllus aruanus	Omnivore	Jenkins, 2019; Sano et al., 1984
Dascyllus reticulatus	Herbivore	Hobson and Chess, 1978
Dascyllus trimaculatus	Omnivore	Allen, 1991
Diagramma pictum labiosum	Generalist Carnivores	Sommer et al., 1996
Diploprion bifasciatum	Piscivores	Jenkins, 2019; Paxton et al., 1989
Echeneis naucrates	Generalist Carnivores	Smith, 1997
Ecsenius bicolor	Herbivore	Jenkins, 2019
Epibulus insidiator	Generalist Carnivores	Lieske and Myers, 1994
Epinephelus bilobatus	Generalist Carnivores	Craig et al., 2011; Kulbicki et al., 2005
Epinephelus coioides	Generalist Carnivores	Kilbicki et al., 2005; Randall and Heemstra, 1991
Epinephelus fasciatus	Generalist Carnivores	Harmelin-Vivien and Bouchon, 1976; Morgans, 1982; Randall and Ben-Tuvia, 1983
Epinephelus malabaricus	Generalist Carnivores	Lieske and Myers, 1994
Epinephelus merra	Generalist Carnivores	Thollot, 1996
Epinephelus multinotatus	Generalist Carnivores	Kailola et al., 1993; Parrish, 1987
Epinephelus quoyanus	Generalist Carnivores	Craig et al., 2011; Heemstra and Randall, 1993
Epinephelus rivulatus	Generalist Carnivores	Sommer et al., 1996
Glaucosoma magnificum	Invertivore	McKay, 1997

Taxa	**Feeding guild**	**Reference**
Gnathanodon speciosus	Generalist Carnivores	Allen and Erdmann, 2012; Lieske and Myers, 1994
Gobiidae spp	Invertivore	Swainston, 2010
Gomphosus varius	Generalist Carnivores	Randall et al., 1990
Gymnothorax thrysoideus	Generalist Carnivores	Bacchet et al., 2005, Swainston, 2010
Halichoeres margaritaceus	Generalist Carnivores	Myers, 1991
Halichoeres melanochir	Invertivore	Sano et al., 1984
Halichoeres nebulosus	Invertivore	Sano et al., 1984
Halichoeres prosopeion	Invertivore	Myers, 1999; Randall et al., 1997
Hemigymnus fasciatus	Invertivore	Sano et al., 1984; Westneat, 2001
Hemigymnus melapterus	Invertivore	Myers, 1999; Westneat, 2001
Heniochus acuminatus	Planktivore	Jenkins. 2019; Masuda and Allen, 1993
Heniochus monoceros	Invertivore	Allen, 1985; Anderson and Hafiz, 1987
Hologymnosus annulatus	Generalist Carnivores	Randall et al., 1990
Hologymnosus doliatus	Generalist Carnivores	Cornic, 1987, Myers, 1991
Hoplolatilus cuniculus	Invertivore	Dooley, 1978
Labroides dimidiatus	Invertivore	Westneat, 2001, Kuiter and Tonozuka, 2001
Leptojulis cyanopleura	Planktivore	Randall, 1996
Lethrinus atkinsoni	Generalist Carnivores	Carpenter, 1997, Kilbicki et al., 2005
Lethrinus genivittatus	Generalist Carnivores	Carpenter and Allen, 1989; Kilbicki et al., 2005; Sano et al., 1984
Lethrinus laticaudis	Generalist Carnivores	Carpenter and Allen, 1989; Salini et al., 1994
Lethrinus lentjan	Generalist Carnivores	Carpenter and Allen, 1989, Kilbicki et al., 2005, Salini et al., 1994
Lethrinus nebulosus	Generalist Carnivores	Salini et al., 1994; Walker, 1978
Lethrinus punctulatus	Generalist Carnivores	Allen, 2009
Lutjanus carponotatus	Generalist Carnivores	Connell, 1998
Lutjanus fulviflamma	Generalist Carnivores	Sommer et al., 1996
Lutjanus lemniscatus	Generalist Carnivores	Allen, 1985
Lutjanus lutjanus	Generalist Carnivores	Allen, 1985; Sommer et al., 1996
Lutjanus monostigma	Generalist Carnivores	Myers, 1999
Lutjanus quinquelineatus	Generalist Carnivores	Allen, 1984
Lutjanus sebae	Generalist Carnivores	Allen, 1985
Lutjanus vitta	Generalist Carnivores	Allen, 1985; Salini et al., 1994
Macropharyngodon negrosensis	Invertivore	Jenkins, 2019
Meiacanthus grammistes	Generalist Carnivores	Myers, 1999
Monacanthus chinensis	Omnivore	Bell et al., 1978; Conacher et al., 1979
Monotaxis grandoculis	Invertivore	Carpenter and Allen, 1989; Kulbicki, et al., 2005
Mulloidichthys vanicolensis	Invertivore	Honebrink, 1990
Naso brachycentron	Herbivore	Sommer et al., 1996
Naso brevirostris	Omnivore	Choat et al., 2002; Randall, 1985
Naso lituratus	Herbivore	Sluka and Miller, 2001
Naso unicornis	Herbivore	Choat et al., 2002; Jenkins, 2019; Tinker, 1978

Taxa	Feeding guild	Reference
Nebrius ferrugineus	Generalist Carnivores	Compagno, 1984; Cortés, 1999; Myers, 1999
Nelusetta ayraud	Generalist Carnivores	Lindholm, 1984
Nemipterus furcosus	Generalist Carnivores	Salini et al., 1994, Russell, 1990
Neoglyphidodon melas	Corallivore	Broad, 2003; Jenkins, 2019; Myer, 1991
Neoglyphidodon nigroris	Omnivore	Allen, 1991
Neopomacentrus azysron	Planktivore	Hammer et al., 1988
Neopomacentrus aktites	Planktivore	Allen et al., 2017
Neotrygon australiae	Generalist Carnivores	Swainston, 2010
Ostorhinchus angustatus	Invertivore	Myers, 1991
Ostorhinchus semilineatus	Invertivore	Horinouchi and Sano, 2000
Ostracion cubicus	Omnivore	Cornic, 1987; Myers, 1991, 1999
Oxymonacanthus longirostris	Corallivore	Barlow, 1987
Parachaetodon ocellatus	Invertivore	Allen, 2006; Kuiter and Tonozuka, 2001
Paramonacanthus choirocephalus	Invertivore	Allen and Erdmann, 2012; Kuiter and Tonozuka, 2001
Parapercis nebulosa	Invertivore	Allen and Swainston, 1988
Paraplotosus butleri	Invertivore	Allen, 1998
Parupeneus barberinoides	Invertivore	Myers, 1991
Parupeneus heptacanthus	Invertivore	Kilbicki et al., 2005
Parupeneus indicus	Generalist Carnivores	Jenkins, 2019; Randall, 2004
Parupeneus spilurus	Invertivore	Sano et al., 1984, Jenkins, 2019
Pateobatis jenkinsii	Generalist Carnivores	Last and Compagno, 1999
Pempheris analis	Planktivore	Randall et al., 1990
Pempheris oualensis	Generalist Carnivores	Hiatt and Strasburg, 1960
Pempheris schwenkii	Planktivore	Randall et al., 1990
Pentapodus emeryii	Generalist Carnivores	Jenkins, 2019; Russell, 1990
Pentapodus porosus	Generalist Carnivores	Russell, 1990
Pentapodus vitta	Generalist Carnivores	Russell, 1990
Petroscirtes breviceps	Omnivore	Sano et al., 1984
Plagiotremus tapeinosoma	Generalist Carnivores	Jenkins, 2019; Sano et al., 1984
Platax batavianus	Generalist Carnivores	Lieske and Myers, 1994
Plectorhinchus multivittatus	Invertivore	Jenkins, 2019
Plectorhinchus polytaenia	Invertivore	Allen and Erdmann, 2012; Lieske and Myers, 1994
Plectropomus spp	Piscivores	Kailola et al., 1993; Kingsford, 1992
Pomacanthus imperator	Invertivore	Anderson and Hafiz, 1987
Pomacanthus semicirculatus	Omnivore	Sommer et al., 1996
Pomacanthus sexstriatus	Omnivore	Allen et al., 1998, Jenkins, 2019
Pomacentrus amboinensis	Omnivore	Allen, 1991; Sano et al., 1984
Pomacentrus coelestis	Omnivore	Hobson and Chess, 1978; Jenkins, 2019
Pomacentrus limosus	Omnivore	Evans et al., 2014
Pomacentrus milleri	Omnivore	Allen, 1991
Pomacentrus moluccensis	Omnivore	Allen, 1991,
Pomacentrus nagasakiensis	Omnivore	Allen,1975, 1991

Taxa	Feeding guild	Reference
Pomacentrus vaiuli	Omnivore	Jenkins, 2019, Myers, 1991
Psammoperca datnioides	Generalist Carnivores	Jenkins, 2019
Pseudodax moluccanus	Omnivore	Westneat, 2001
Ptereleotris evides	Planktivore	Randall and Hoese, 1985; Sano et al., 1984
Ptereleotris microlepis	Planktivore	Bacchet et al., 2005
Pterocaesio chrysozona	Planktivore	Carpenter, 1987,1988; Bellwood, 1988
Pterocaesio lativittata	Planktivore	Carpenter, 1988
Pterocaesio tile	Planktivore	Bellwood, 1988; Carpenter, 1988
Pterois volitans	Generalist Carnivores	Myer, 1991; Sano et al., 1984
Sargocentron rubrum	Generalist Carnivores	Göthel, 1992; Randall et al., 1990
Scarus forsteni	Herbivore	Bacchet et al., 2005
Scarus ghobban	Herbivore	Humann and Deloach, 1993
Scarus prasiognathos	Herbivore	Bruce and Randall, 1984
Scarus psittacus	Herbivore	Bruce and Randall, 1984
Scarus rivulatus	Herbivore	Schroeder, 1980
Scarus schlegeli	Herbivore	Bacchet et al., 2005
Scolopsis affinis	Generalist Carnivores	Jenkinsn 2019; Mequila and Campos, 2007
Scolopsis bilineata	Generalist Carnivores	Russell, 1990
Scolopsis monogramma	Generalist Carnivores	Jenkins, 2019; Russell, 1990; Salini et al., 1994
Scomberoides lysan	Generalist Carnivores	Fischer et al., 1990
Scomberomorus spp	Generalist Carnivores	Bachok et al., 2004; Jenkins, 2019
Selaroides leptolepis	Generalist Carnivores	Paxton et al., 1989; Yamashita et al., 1987
Siganus argenteus	Herbivore	Sommer et al., 1996
Siganus canaliculatus	Herbivore	Cuihong et al., 2014; Soh, 1976
Siganus doliatus	Herbivore	Bennett and Bellwood, 2011; Cheal et al., 2010
Siganus fuscescens	Herbivore	Lieske and Myers, 1994; Pillans et al., 2004
Siganus lineatus	Herbivore	Fox et al., 2009; Thollot, 1996
Siganus punctatus	Herbivore	Rhodes et al., 2017; Woodland, 1997
Siganus trispilos	Herbivore	Woodland and Allen, 1977
Sphyraena jello	Piscivores	Bachok et al., 2004; Jenkins, 2019
Sphyraena obtusata	Piscivores	Kilbicki et al., 2005; Thollot, 1996
Stegastes fasciolatus	Herbivore	Jenkins, 2019
Stegastes nigricans	Herbivore	Hata and Kato, 2004
Stegastes obreptus	Herbivore	Allen and Emery, 1985; Hata and Kato, 2004
Stethojulis bandanensis	Invertivore	Sano et al., 1984
Stethojulis interrupta	Invertivore	Randall, 2000
Stethojulis strigiventer	Invertivore	Gell and Whittington, 2002; Randall, 2000, Sano et al., 1984
Sufflamen chrysopterum	Invertivore	Myers, 1991
Symphorus nematophorus	Piscivores	Allen, 1985; Kilbicki et al., 2005
Synodus variegatus	Generalist Carnivores	Bacchet et al., 2005

Taxa	Feeding guild	Reference
Thalassoma amblycephalum	Planktivore	Jenkins, 2019
Thalassoma hardwicke	Generalist Carnivores	Jenkins, 2019; Myers, 1991; Randall et al., 1990
Thalassoma lunare	Invertivore	Westneat, 2001
Thalassoma lutescens	Invertivore	Sano et al., 1984
Torquigener whitleyi	Invertivore	Sainsbury et al., 1985
Triaenodon obesus	Generalist Carnivores	Compagno, 1984
Upeneus tragula	Invertivore	Jenkins, 2019; Kilbicki et al., 2005
Yongeichthys nebulosus	Invertivore	Allen, 1997
Zanclus cornutus	Invertivore	Anderson and Hafiz, 1987; Dominici-Arosemena and Wolff, 2006; Hobson, 1975
Zebrasoma scopas	Herbivore	Guiasu and Winterbottom, 1998

Table App. 4.2 The value of commercial fish ($AUD/kg) based on the mean wet weight market value for commercial species for 2017/2018 (Gaughan et al., 2019). C = Commercial Fisheries; MC = Minor Commercial Fisheries; R = Recreational Fisheries; AQM = Aquarium Fisheries; AQC = Aquaculture Fisheries; AF = Artisanal Fisheries.

Row Labels	Common name	Catch value ($AUD/kg)	Target Status
Carangoides fulvoguttatus	Yellowspotted trevally	1.70	C/R
Cephalopholis boenak	Brownbarred Rockcod	7.15	C/R/AQM
Cephalopholis miniata	Coral rockcod	9.84	C/R
Choerodon cauteroma	Bluespotted tuskfish	6.80	C/R
Choerodon cephalotes	Purple tuskfish	6.80	C/R
Choerodon cyanodus	Blue tuskfish	6.80	C/R/AQM
Choerodon schoenleinii	Blackspot tuskfish	6.80	C/R/AQM
Diagramma pictum labiosum	Painted sweetlips	4.95	C/R
Epinephelus bilobatus	Frostback rockcod	5.85	C/R
Epinephelus coioides	Goldspotted rockcod	6.77	C/R/AQC
Epinephelus fasciatus	Blacktip rockcod	5.47	C/R
Epinephelus malabaricus	Blackspotted rockcod	6.99	C/R
Epinephelus merra	Birdwire rockcod	9.84	C/R/SF
Epinephelus multinotatus	Rankin cod	8.08	C/R
Epinephelus rivulatus	Chinaman rockcod	5.47	-
Glaucosoma magnificum	Threadfin pearl perch	7.03	C/R
Gnathanodon speciosus	Golden trevally	3.08	C/R/AQM
Lethrinus atkinsoni	Yellowtail emperor	4.42	C
Lethrinus laticaudis	Grass emperor	6.76	C/R
Lethrinus lentjan	Redspot emperor	4.95	C/R
Lethrinus nebulosus	Spangled emperor	6.00	C/R
Lethrinus punctulatus	Bluespotted emperor	4.28	C/R
Lutjanus carponotatus	Stripey snapper	3.82	C/R
Lutjanus lemniscatus	Darktail snapper	5.39	C/R
Lutjanus quinquelineatus	Fiveline snapper	3.82	C/R
Lutjanus sebae	Red emperor	11.31	C/R/AQM
Lutjanus vitta	Brownstripe snapper	3.82	C/R
Monotaxis grandoculis	Bigeye seabream	5.22	C/R/AF
Nelusetta ayraud	Chinaman-leatherjacket	4.35	C/R
Plectropomus spp	Coral trout	14.91	C/R/AQC
Psammoperca datnioides	Black sand bass	7.86	C/R
Scolopsis monogramma	Rainbow Monocle bream	1.89	C
Scomberoides lysan	Doublespotted queenfish	5.10	MC/R
Scomberomorus spp	Mackerel	9.54	C/R
Siganus fuscescens	Black rabbitfish	3.96	C/AQC
Siganus lineatus	Goldlined rabbitfish	3.96	C
Sphyraena jello	Pickheadle barracuda	3.99	C/R
Sphyraena obtusata	Obtuse barracuda	3.99	R
Symphorus nematophorus	Chinamanfish	5.48	C/R