RIGS-TO-REEFS ECOLOGY: OFFSHORE OIL AND GAS PLATFORMS AS NOVEL ECOSYSTEMS

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The Wandoo B platform off the coast of Dampier, Western Australia.

THE UNIVERSITY OF WESTERN AUSTRALIA

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The following approvals were obtained prior to commencing the relevant work described in this thesis: AU-COM2012-170, AU-COM2018-426, PA2018-00036-1, PA2018-00091-1, PA2018-00091-2, PA2018-00079, DPAW 01-000049-4, DPAW 01-000049-7, DPAW 01-000049-8, CMR-17-000526, CMR-16-000426, CMR-18-000550, and Fisheries Exemption Numbers 2853 and 3172.

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ABSTRACT

There are thousands of oil and gas platforms (offshore platforms) situated offshore of coastlines around the world. Shortly after installation, these platforms become habitats for a variety of marine organisms, and over their ~30-40 year life spans, they can develop into highly complex artificial reefs. In many locations, these platforms also provide protection from fishing, acting as de facto marine protected areas. When offshore platforms reach the end of their productive lives, they are decommissioned, a process which, in most cases, involves the complete removal of the platform from the marine environment. However, this process also results in the destruction of the long-established marine community. In some regions, Rigs-to-Reefs programs provide options for *in situ* decommissioning, ensuring that the artificial reefs created by the infrastructure are preserved. However, the ecology of many active offshore platforms, particularly outside of major oil and gas producing regions, is poorly understood.

In Chapter 2, I reviewed the literature on the ecology of oil and gas platforms globally to determine whether restoration ecology principles, and specifically the concept of novel ecosystems, is applicable to offshore platforms. I found that the ecosystems created by offshore platforms are consistent with the concept of novel ecosystems, and therefore novel ecosystems management principles can be applied to offshore platforms. In this chapter I provide a method for recognising, classifying, and managing the ecosystems created by offshore platforms, using existing decommissioning decision analysis models already implemented by industry stakeholders.

The empirical components of my thesis are based on fieldwork at an active oil platform in northwest Australia and two natural “control” habitats within the region. Stereo-BRUVS, both midwater and seabed, were used because they are non-destructive and standardised method for documenting the diversity, abundance and biomass of both demersal and pelagic species. In total, 1,125 BRUVS were deployed, recording 35,070 animals from 358 taxa. I used this dataset to assess the ecological role of offshore platforms, and demonstrated the usefulness of BRUVS for documenting rare species and behaviours.

In Chapter 3, I assessed the Wandoo oil platform within the novel ecosystems framework. I compared the fish assemblages at Wandoo with two natural sites: one resembling the habitat found in the oilfield pre-installation, and the other being a natural reef. Both species assemblages, demersal and pelagic, and the benthic habitat around the Wandoo platform more closely resemble a natural reef than the site pre-installation. This chapter demonstrates the ecological importance of the Wandoo platform within a region largely depauperate of hard substrate, and the role platforms play in increasing regional diversity and providing protection from destructive fishing activities.

In Chapter 4, I document the first known wild observation of putative decapod mimicry by a cuttlefish *Sepia* cf. *smithi*. This cuttlefish was observed at the Wandoo platform, displaying 'crustacean-like aggressive mimicry' while approaching the stereo-BRUVS bait bag. Records such as this are important in understanding how animals behave in their natural environments, and show that stereo-BRUVS are an effective method of studying animal behaviour *in situ*. Industry-focused research enables us to expand our knowledge of the typically understudied offshore ecosystems around offshore structures, and observe new behaviours and interactions.

In Chapter 5 I documented the abundance of threatened elasmobranchs in Australia, with a focus on the Wandoo field. Endangered elasmobranchs are highly vulnerable to fishing pressure and climate change, and knowledge of where they are found is critical to effectively manage their populations. Abundance of these species around Wandoo was among the highest across the nine regions studied. This finding is significant because Wandoo excludes fishing activity, acting as a refuge for these species in a region exposed to significant fishing pressure.

The presence of offshore platforms can result in the emergence of novel ecosystems characterised by unique species assemblages and ecosystem services. The marine community around the Wandoo platform has shifted from its historical state to resemble a reef-type community, and a novel ecosystem has emerged. This platform also potentially acts as a refuge for threatened elasmobranchs in a region of high fishing pressure, further underlining the importance of recognising the ecological value of offshore platforms when making decommissioning decisions. The findings presented here will help to inform decommissioning of the Wandoo platform, and more generally Australia’s future decommissioning policies.

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I would firstly like to thank Jessica Meeuwig for her endless patience and guidance as my supervisor. Jessica took a chance on me, finding a path for me to enrol as a PhD candidate where others had said it was impossible. Over the four years since our first meeting, Jessica has been a mentor and a friend. Through the trials and tribulations associated with both PhD candidature and life in general, she was there to offer encouragement, support, and a sense of perspective. I have watched and been inspired by Jessica's tireless efforts to save the oceans and to encourage representation in science, no matter who stands in her way. I am eternally grateful for everything I have learned from her on this journey.

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My colleagues and friends in the Marine Futures Lab have been a constant source of support, motivation, entertainment and inspiration over the past four years. Tom Tothill was a key part of this journey in everything from fieldwork, to video analysis, to insightful conversations on offshore platform ecology. I am particularly grateful to everyone who joined me on the less than glamorous expeditions to Wandoo: David Tickler, Louis Masarei, Vyvyan Summers, Lincoln Hood, and Jack McElhinney. Thank you to the past and present members of the MFL family, particularly Chris Thompson, Adam Jolly, Nikki De Campe, Alex McLennan, Claire Raphael, Hanna Jabour Christ, Naima Andrea López, Jem Turner, Rachel White, Gabriel Vianna, Kristina Heidrich, James Hehre, and Shona Murray.

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Student contribution to work:

I conceived the study with input from JJM and I wrote the first draft of the manuscript. I revised and submitted the manuscript with input from all co-authors.

Co-author signatures and dates:

Prof. Jessica Meeuwig	A/Prof Jan Hemmi	Prof. Richard Hobbs
Date: 30/11/2020	Date: 30/11/2020	Date: 30/11/2020

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I developed the idea with input from JJM. I conducted all six field expeditions and processed around 75% of the imagery obtained during the expeditions, with the remainder processed by fellow lab members. I completed the data analysis and drafted the manuscript. I revised the manuscript with input from co-authors.

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I observed the novel behaviour during image processing and developed the idea with input from JJM. I conducted all six field expeditions and processed around 75% of the imagery obtained during the expeditions, with the remainder processed by fellow lab members. I completed the data analysis and drafted the manuscript. I revised and submitted the manuscript with input from JJM.

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I conceived the study with input from JJM. I conducted all six expeditions at the Wandoo locations, and processed around 75% of the imagery obtained from these locations. The rest of the expeditions and associated image processing were completed by fellow lab members. I wrote the first draft, and revised and submitted the manuscript with input from JJM.

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Student contribution to work:

I conceived this publication along with TT. The concept for the publication was developed further by all authors, and I wrote the first draft of the manuscript along with TT. The manuscript was revised and submitted with input from all authors.

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Mr Thomas Tothill

Date: 30/11/2020

Student signature:

Date: 11/12/2020

I, Professor Jessica Meeuwig, certify that the student's statements regarding their contribution to each of the works listed above are correct.

Coordinating supervisor signature:

Date: 11/12/2020

STATEMENT OF CANDIDATE CONTRIBUTIONS

This Dissertation contains a General Introduction (Chapter 1), four data chapters (Chapters 2-5) each of which is in the form of a manuscript that is about to be submitted (Chapters 3 and 5), or published (Chapters 2 and 4), and a General Discussion (Chapter 6).

I developed the ideas and hypotheses that underpin this Dissertation with input from my supervisors, Prof. Jessica Meeuwig, A/Prof. Jan Hemmi, and Prof. Richard Hobbs. My supervisors revised the manuscripts with input from other colleagues who co-authored the specific chapters.

Chapter 2 was conceived when I first started reading the literature on offshore platforms. I noticed that the novel ecosystems concept had not, at that stage in 2017, been mentioned at all in the literature on offshore platform ecology. I developed the concept for this chapter through discussions with Prof. Jessica Meeuwig and Prof. Richard Hobbs. I drafted the manuscript and all of my supervisors provided valuable input in revising and improving the manuscript.

The data collected from the Wandoo field and adjacent natural habitats comprised the major field component of my dissertation, and involved six field expeditions over three years. I created the survey designs for these expeditions with input from Prof. Jessica Meeuwig. I conducted all expeditions with assistance from various members of the MFL. I conducted most of the image analysis, supported by the MFL due to the volume of imagery obtained during the expeditions. These data formed the basis of chapters 3-5 of this dissertation. The Wandoo expeditions were funded by Vermilion Oil and Gas Australia.

The databases used to analyse the abundance of threatened species throughout tropical Australia (Chapter 5) were mainly generated by the Great West Ozzie Transect (GWOT). This sampling programme was conducted by the Marine Futures Lab (MFL) from the Kimberley in the north to the Recherche Archipelago in the south and commenced in 2013. The Marine Futures Lab databases used in Chapter 5 also contain data from previous surveys conducted by the MFL. These surveys were funded by a combination of sources, including the Ian Potter Foundation, TeachGreen, Woodside Energy and the Clough Foundation.

I conceived of the idea for the manuscript presented in Chapter 3, with input from Prof. Meeuwig. I conducted the analyses and wrote the first draft of the manuscript with input from Prof. Meeuwig. Prof. Meeuwig and Prof. Richard Hobbs provided input and revision of the manuscript. I developed the concepts for chapters 4 and 5 with input from Prof. Jessica Meeuwig. I conducted the analyses and drafted the manuscripts, with input from Prof. Meeuwig throughout.

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CHAPTER 1 GENERAL INTRODUCTION

1.1 Offshore platforms

Background

Oil and natural gas together account for 60% of the fuel consumed worldwide (British Petroleum P.L.C., 2020). Offshore oil and gas fields contribute a significant portion of global energy production, with 30% of oil and 27% of gas produced offshore (Planete Energies, 2015; US Energy Information Administration, 2016). Offshore energy production began in the Gulf of Mexico in the 1940s: Ship Shoal Block 32, a converted World War II navy barge, was installed in waters off the Louisiana coast in 1947, becoming the first platform to be installed out of sight of land (Aagard and Besse, 1973; Beu, 1988). This milestone led to 70 years of technological and engineering advances, with modern-day platforms weighing hundreds of thousands of tonnes, and able to withstand severe environmental conditions including tropical cyclones (Dragani and Kotenev, 2013; Elsayed et al., 2016; Sheng and Hong, 2020). The world's deepest platform, Perdido, is installed in waters 2,450 m deep in the Gulf of Mexico, underlining how far offshore production has progressed in a relatively short period (Lohr and Smith, 2010). There are currently over 12,000 offshore oil and gas platforms (hereafter offshore platforms) installed in the continental shelf waters of 53 countries, varying greatly in size and water depth (Ars and Rios, 2017; Parente et al., 2006). Most oil fields have a production life of around 15-30 years, while deeper fields may have lifespans of less than ten years due to higher extractive costs (Planete Energies, 2015).

Decommissioning

The end of life process for offshore platforms is referred to as decommissioning. Offshore platforms generally reach the end of their productive lives when extraction is no longer profitable, even though the platform itself may still be fit-for-purpose. Decommissioning is a regulated process which involves shutting down production, plugging wells, and cleaning, capping, and possibly removing subsea pipelines and the platform itself (Hakam and Thornton, 2000). Decommissioning regulations vary greatly between regions: countries and regions such as the Australia and the North Sea prescribe complete removal of offshore platforms (Chandler et al., 2017; Ounanian et al., 2019). In contrast, the Gulf of Mexico, California, Brunei, and Malaysia all allow for *in situ* decommissioning options (Fowler et al., 2015; Pietri et al., 2011; Reggio Jr., 1987). The various regulations and decommissioning processes around the world were comprehensively assessed in two recent reviews (Bull and Love, 2019; Sommer et al., 2019).

Decommissioning of offshore platforms typically occurs under one of five scenarios: (1) complete removal, whereby the entire platform structure is removed for onshore disposal (Fig. 1.1); (2) leave in place, which involves removing the superstructure and placing navigational aids on the above-surface shaft or jacket structure; (3) topping, where the structure is cut below the waterline and the top portion is either removed or placed next to the remaining structure; (4) horizontal reefing, which involves cutting the platform at the seabed and laying the structure horizontally; and (5) tow-and-place, where the platform is removed from the seabed and “reefed” at another location (Dauterive, 2000; Sommer et al., 2019). Other suggested alternatives include converting the existing platform for use as hotels, wind/wave power generators; mariculture farms, or research stations (Schroeder and Love, 2004; Sommer et al., 2019; Zawawi et al., 2012). Decommissioning is regulated under international law, specifically the 1996 Protocol to the London Dumping Convention, which prohibits the dumping, abandonment, or toppling of offshore platforms for the sole purpose of disposal (Elizabeth, 1996). However, *in situ* decommissioning is not specifically prohibited by this Protocol, which states that dumping does not include placement of the platform for purposes other than disposal (Elizabeth, 1996; Techera and Chandler, 2015).

Rigs-to-Reefs

The primary reason for decommissioning offshore platforms *in situ* is for their use as designated artificial reefs, under programs typical referred to as “Rigs-to-Reefs” (RTR). The first structure to be “reefed”, prior to an official RTR program, was in 1979 when a 2,000 tonne subsea production system was towed from Louisiana to Florida to create an artificial reef (Kaiser, 2006). RTR was soon signed into federal law under the 1984 National Fishing Enhancement Act, with the primary intention of improving offshore fishing in the Gulf of Mexico (Reggio Jr., 1987). Over 500 platforms have since been reefed under state RTR programs in the Gulf of Mexico, which is a relatively small portion, around 11%, of the total number of platforms that have been installed in the region (Bull and Love, 2019). Brunei was not far behind the US in terms of RTR, establishing its own program in 1988 (Bull and Love, 2019). To date seven platforms have been reefed in Brunei, however with around 150 platforms having been installed at least 20 years ago in this region, there is scope for significant expansion of the Brunei RTR program (Bull and Love, 2019; Lyons et al., 2015). Malaysia also has many

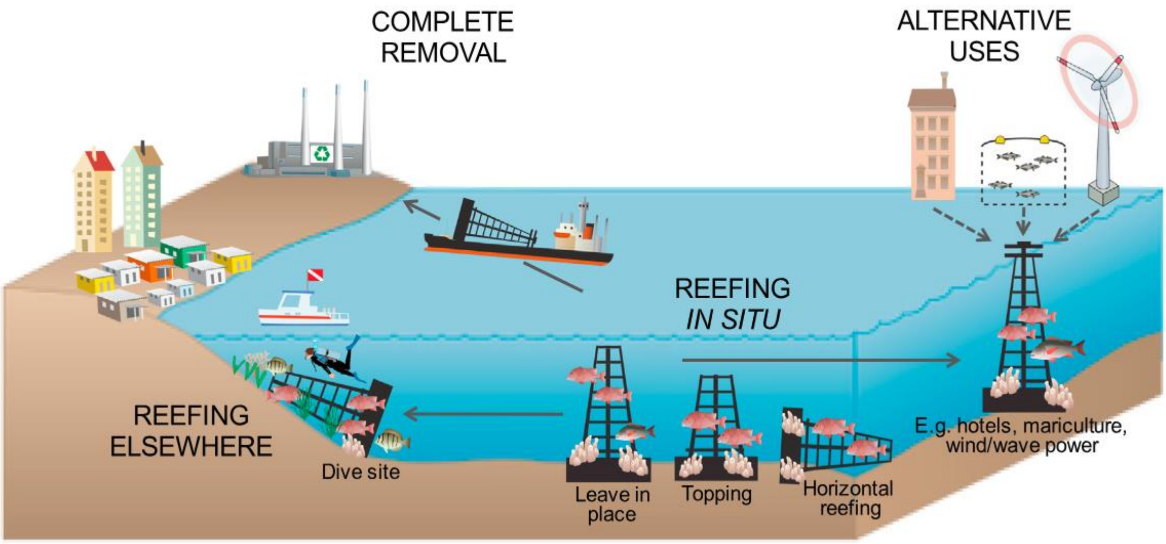

Figure 1.1 Potential options for the decommissioning of offshore platforms, including complete removal, various reefing options, and alternative uses of the existing infrastructure. Figure from Sommer et al. (2019).

potential RTR candidates: there are hundreds of platforms in this region, around half of which are older than 25 years (Zawawi et al., 2012). However, as of 2015, the only offshore platform to be reefed in Malaysia was the Baram-8 platform, which collapsed in a storm in 1975 before being salvaged and reefed in a different location in 2004 (Lyons et al., 2015). Southeast Asia as a region may benefit from RTR programs, with 1,800 platforms of which almost half have been in place for over 20 years (Ars and Rios, 2017). Many Southeast Asian countries already have established artificial reef programs to enhance fisheries and tourism, and the shortage of decommissioning yards in the region would make onshore disposal difficult (Lyons et al., 2015).

The North Sea and California are both regions where RTR programs have been discussed, or even legislated, but not implemented. In the North Sea, Greenpeace's protest over the offshore disposal of the Brent Spar in 1995 effectively excluded RTR from the region (Jørgensen, 2012). The legacy of the Brent Spar has shaped RTR policy not just in the North Sea, but in other regions around the world (Lyons et al., 2015; Salcido, 2005; Zawawi et al., 2012). In California, RTR was legislated in 2010, after three previous unsuccessful attempts (Bull and Love, 2019; Pietri et al., 2011; Schroeder and Love, 2004). There was considerable division among stakeholders over the impacts of RTR in California, and no platforms have yet been reefed in the region (Manago and Williamson, 1997; Ounanian et al., 2019). Scientific research played a key role in the successful legislation of the California RTR program (Macreadie et al., 2012; Pietri et al., 2011), and Australia is taking a similar approach to its decommissioning policy. In Australia, offshore platforms must be completely removed at the time of decommissioning, despite scientific evidence of the environmental benefits of RTR (Techera and Chandler, 2015). However, Australia's position has recently come under review, and this review process is to be based on independent scientific research, through the National Decommissioning Research Initiative (NDRI) as well as ongoing decommissioning research projects (Offshore Resources Branch, 2018).

Ecology of offshore platforms

Offshore platforms have been converted into artificial reefs for decades, and these structures also function as artificial reefs during their productive lives (Shinn, 1974). When a platform is installed, it provides bare, hard substrate that is available for colonisation by sessile organisms including sponges, corals, mussels and hydroids (Forteath et al., 1982; Todd et al., 2020). The new habitat provided by offshore platforms can be transformed into complex reef-type habitat within a few years, and supports a range of marine fauna including invertebrates, fish, and marine megafauna (Driessen, 1986; Love et al., 2003; McLean et al., 2017; Todd et al., 2016). Offshore platforms are some of the most productive marine habitats globally, with higher biomass and secondary production than some pristine Pacific coral reefs (Claisse et al., 2014; Friedlander et al., 2014). Similarly to pristine reefs, much of the biomass around platforms is made up of top predators such as groupers, jacks and sharks, as described in Gabon (Friedlander et al., 2014).

Offshore platforms are physically complex structures and create habitat from the seafloor to the surface. High habitat complexity is associated with higher abundance and diversity of fishes through the provision of refuge opportunity and reduced predation pressure (Claisse et al., 2014; Lingo and Szedlmayer, 2006). Habitat complexity also influences reproduction and recruitment, with some juvenile fishes preferentially selecting more complex habitat (Sayer et al., 2005; Todd et al., 2018). Artificial reefs, including offshore platforms, create an “ecological halo” of elevated abundance and diversity in the area surrounding the structure, to a distance of around 15-34 m (Reeds et al., 2018; Scarcella et al., 2011; Stanley and Wilson, 1996). In California, offshore platforms have been found to support large populations of Critically Endangered bocaccio rockfish *Sebastes paucispinis* (IUCN, 2020): populations at eight offshore platforms support an estimated 430,000 juvenile bocaccio (Love et al., 2006). Juvenile recruitment was also higher at platforms than in natural habitats (Love et al., 2006). In the Gulf of Mexico, 246 fish species have been recorded at offshore platforms (Cowan Jr. and Rose, 2016). The great barracuda *Sphyraena barracuda* was not known as a sport fishing species in Louisiana prior to the presence of offshore platforms (Dugas et al., 1979). In northwest Australia, Fowler and Booth (2012) found that artificial structures could support full populations of the red-belted anthias *Pseudanthias rubrizonatus*, from newly recruited juveniles to mature adults. Offshore infrastructure in northwest Australia supports a diverse range of both pelagic and reef-dependent species, and plays a particular important role for commercial fish species such as goldband snapper *Pristipomoides multidens*, saddletail snapper *Lutjanus malabaricus*, and mangrove jack *Lutjanus argentimaculatus* (Bond et al., 2018b; Pradella et al., 2014).

Offshore platforms generally exclude fishing activity, particularly commercial fishing. This exclusion can either be through legislation, as is the case in Australia and Ghana, or through the presence of infrastructure acting as physical obstacles to longlining and seabed trawling (Chalfin, 2018; Commonwealth of Australia, 2010; de Groot, 1982;

Fabi et al., 2004; McLean et al., 2019). The exclusion of fishing effectively means that offshore platforms and the waters surrounding them function as *de facto* marine protected areas (MPAs), providing a refuge from fishing and potentially helping to rebuild populations of overfished species (Friedlander et al., 2014; Fujii and Jamieson, 2016; Love et al., 2006).

Offshore platforms may play important roles for marine megafauna. Platforms have been shown to act as fish aggregating devices (FADs), attracting small pelagic fishes and providing enhanced foraging opportunity for large predators, including transient species that may not be resident at the platforms (Franks, 2000; Scarcella et al., 2011). Large predators including bull sharks *Carcharhinus leucas*, tiger sharks *Galeocerdo cuvier*, great hammerheads *Sphyrna mokarran*, and porbeagles *Lamna nasus* have all been reported around offshore platform various regions, while white sharks *Carcharodon carcharias* have been reported near platforms in the Adriatic Sea (De Maddalena, 2000; Franks, 2000; Haugen and Papastamatiou, 2019; Reynolds et al., 2018). Other large marine megafauna observed at offshore platforms include whale sharks *Rhincodon typus*, basking sharks *Cetorhinus maximus*, oceanic manta rays *Mobula birostris*, minke whales *Balaenoptera acutorostrata*, and various seal and porpoise species (Bernstein et al., 2010; McLean et al., 2019; Robinson et al., 2013; Todd et al., 2016).

1.2 Novel Ecosystems

A novel ecosystem is one which has been altered by human activity and where restoration is not feasible or would result in the loss of ecosystem value (Hobbs et al., 2013). The term "novel ecosystem" was introduced in 1997 (Chapin and Starfield, 1997), but the most comprehensive definition of the concept was developed by Hobbs et al. (2013) (Box 1). Novel ecosystems can emerge through both direct and indirect human activity, including species introductions, land-use changes, and climate change (Kennedy et al., 2013). A key assertion about novel ecosystems is that ecosystems that have been altered are not necessarily 'degraded', but may just provide different ecosystem services from what was present before (Hobbs, 2016). A frequently used example of a novel ecosystem is the Mt Sutro forest in San Francisco (Venton, 2013).

The native vegetation in this area has been almost entirely replaced by non-native species, predominantly Australian eucalyptus, creating a cloud forest. Due to the perceived fire risk posed by eucalyptus, it was proposed that the ecosystem be restored through the removal of the eucalyptus and planting of native vegetation. However, the cloud forest is argued to be less prone to fire due to its fog-trapping qualities. Furthermore, the Mt Sutro forest is the largest urban forest in San Francisco and is highly valued by the community for recreation, with significant public campaigns to save the forest (Venton, 2013). These factors represent important environmental and social considerations preventing this novel ecosystem from being restored to its historical state.

BOX 1: NOVEL ECOSYSTEMS DEFINITION (Hobbs et al., 2013)

"A NOVEL ECOSYSTEM IS A SYSTEM OF ABIOTIC, BIOTIC AND SOCIAL COMPONENTS THAT, BY VIRTUE OF HUMAN INFLUENCE, DIFFER FROM THOSE THAT PREVAILED HISTORICALLY, HAVING A TENDENCY TO SELF-ORGANIZE AND MANIFEST NOVEL QUALITIES WITHOUT INTENSIVE HUMAN MANAGEMENT. NOVEL ECOSYSTEMS ARE DISTINGUISHED FROM HYBRID ECOSYSTEMS BY PRACTICAL LIMITATIONS (A COMBINATION OF ECOLOGICAL, ENVIRONMENTAL AND SOCIAL THRESHOLDS) ON THE RECOVERY OF HISTORICAL QUALITIES."

There have been a handful of studies applying the novel ecosystem concept to marine ecosystems, including regime shifts on coral reefs and altering fish assemblages due to warming oceans (Graham et al., 2014; Harborne and Mumby, 2011). Various anthropogenic impacts in the oceans facilitate the emergence of novel ecosystems. Climate change-related impacts include ocean acidification, changes in temperature and oxygen content, and altered ocean circulation (Doney et al., 2012). These broad-scale impacts drive novelty in marine systems concurrently with regional impacts, including illegal, unreported and unregulated (IUU) fishing, aquaculture, point source pollution, and coastal engineering (Perring and Ellis, 2013).

Offshore platforms appear to be ideal candidates for the application of the novel ecosystem concept to marine systems, with ecosystem-level shifts occurring through what is effectively the creation of large artificial reefs. However, the novel ecosystem concept has only recently begun to gain traction in the field of offshore platform ecology. Schläppy and Hobbs (2019) developed a framework for classifying altered marine ecosystems, including offshore platforms, as novel, hybrid, or designed ecosystems. Sommer et al. (2019) suggested that the ecosystem-level shifts that occur due to the presence of offshore platforms present qualities consistent with the novel ecosystem concept. However, there has not yet been a quantitative assessment of an offshore platform within the framework of novel ecosystems.

1.3 Methods for Studying Platform Fish Communities

Offshore platforms present unique challenges for ecological sampling. Many platforms are located in waters too deep to be effectively surveyed by SCUBA divers, and access to the waters surrounding platforms is restricted in some regions. Many offshore platforms are also located significant distances from land, or in areas prone to adverse environmental conditions. A suite of sampling methods has been used to study the marine communities associated with offshore platforms. SCUBA diver observations of fish distribution on offshore platforms occurred as far back as the 1970s and UVC surveys have been conducted by divers on platforms around the world, including Brunei, the Gulf of Mexico, and California (Bull and Kendall, 1994; Chou et al., 1992; Meyer-Gutbrod et al., 2019; Shinn, 1974). A major disadvantage of this method is that the presence of divers can influence the species composition and density of fishes (Bohnsack and Bannerot, 1986; Sale and Douglas, 1981). Divers are also limited in the depth at which they can operate, and in many cases can only survey the shallower sections of a platform. These constraints can be overcome through combining diver surveys with ROV or submersible surveys of the deeper areas (Love et al., 1994). Manned submersibles allow for visual observations at significantly deeper depths than SCUBA divers, and have been used in the Gulf of Mexico (Shinn and Wicklund, 1989), as well as in a seven-year long survey in California (Love et al., 2019). Fish catch data have been used to sample offshore platform-associated communities, either experimentally in the form of trammel net traps and longline surveys (Ajemian et al., 2015; Fabi et al., 2004; Scarcella et al., 2011), or using data from local fishing activity

(Fujii, 2015). Fishing is also a component of tagging studies, with both acoustic telemetry and mark-recapture tagging used at offshore platforms (Everett et al., 2020; Jørgensen et al., 2002; Love et al., 1994). Hydroacoustic surveys have been used to quantify fish abundance in the North Sea and the Gulf of Mexico, while ROVs have been used both in targeted surveys, and the use of industry ROV archives (McLean et al., 2017, 2019; Soldal et al., 2002; Stanley and Wilson, 2000; Todd et al., 2018). Platform-based observations of marine megafauna have been documented both in the form of opportunistic sightings (Haugen and Papastamatiou, 2019; Robinson et al., 2013) and designed monitoring programs (Todd et al., 2016).

Baited remote underwater video systems (BRUVS) have been used more recently in a handful of studies on offshore infrastructure, in order to survey communities at various depths in the water column as well as on the seabed (Barker and Cowan Jr., 2018; Bond et al., 2018b; Schramm et al., 2020). Stereo-BRUVS are a well-established method for non-destructively sampling marine communities (Cappo et al., 2006) that can be used to sample both the seabed and the water column, (Letessier et al., 2013; Whitmarsh et al., 2017). Stereo-BRUVS are cost-effective, can be deployed across large spatial scales, and are used to study abundance, biomass, diversity and distribution of marine fauna (Cappo et al., 2006; Letessier et al., 2015b). Stereo-BRUVS are also useful for studying less abundant animals such as migratory or endangered species (Letessier et al., 2015a; Thompson et al., 2019), and have recorded a suite of unique animal behaviours (Barley et al., 2016; Birt et al., 2019).

1.4 Australia's Northwest Shelf

Australia's tropical marine region is vast, ranging across over 50 degrees of longitude from the Cocos-Keeling Islands remote territory in the west to the Great Barrier Reef in the East. This region encompasses diverse ecosystems, including seagrass beds, mangroves, coral reefs, seamounts, and estuaries (Lough, 2008). These ecosystems are used for a range of activities, including commercial and recreational fishing, oil and gas exploration and production, aquaculture, tourism, and recreation. Many of these ecosystems are only are partially protected from extractive activities under multiple-use MPAs, with the notable tropical MPAs being the Ningaloo Marine Park, Great

Kimberley Marine Park, and Great Barrier Reef Marine Park (Department of Parks and Wildlife, 2020; Parks Australia, 2020).

This dissertation focuses on the Northwest Shelf (NWS) region along Australia's tropical northwest coast. The NWS spans almost 2,500 km, from North West Cape in the south to Melville Island in the north, and is rich in natural resources (Wilson, 2013). This geographic province is comprised of four major sedimentary basins, from north to south: Bonaparte, Browse, Offshore Canning, and Carnarvon (Purcell and Purcell, 1988). The NWS has estimated oil reserves of 2.6 billion barrels, but the most dominant natural resource is gas (Longley et al., 2002). The NWS is a world-class gas province, with estimated reserves of 152 trillion cubic feet, making up 84% by barrels of oil equivalent of the oil and gas reserves on the NWS (Longley et al., 2002). Exploration drilling commenced in 1953, and today there are around 60 production facilities and thousands of kilometres of subsea pipelines on the NWS (Bond et al., 2018b; Geoscience Australia, 2009; Longley et al., 2002). The production facilities are diverse in their design and include: semi-submersible platforms; floating production, storage, and offloading vessels (FPSOs); monopods; conventional steel jacket structures; and concrete gravity structures (CGS). The NWS facilities also include the Shell Prelude and Ichthys Explorer, the world's largest FPSO and semi-submersible platform respectively (Gust et al., 2019; Marshall and Grose, 2014). The infrastructure on the NWS is generally sparsely distributed over a distance of some 2,000 km, from offshore of Exmouth in the south to the Timor Sea in the north. However, approximately two thirds of the production facilities on the NWS are found in the Carnarvon Basin (Geoscience Australia, 2009).

The NWS has a diverse range of marine habitats and is one of the world's biodiversity hotspots (Roberts et al., 2002; Wilson, 2013). These habitats include mangroves, coral reefs, offshore shoals, submarine canyons, and macrobenthos communities (Commonwealth of Australia, 2012; Fromont et al., 2016; Wilson, 2013). Some areas of the NWS are globally recognised areas of ecological importance, including the Ashmore Reef Ramsar Wetland and the Ningaloo Coast World Heritage Area (Anon., 2018). The high diversity and productivity of the NWS are driven by multiple factors: it is mostly shallow, with 40% of the area being less than 200 m deep; it is a sink for tropical species from the Indo-West Pacific via the Indonesian Throughflow current; and large internal tides encourage mixing of waters across depths, bringing nutrients into surface waters (Anon., 2018; Holloway, 2001; Richards et al., 2015; Wilson, 2013). The NWS is inhabited by globally significant populations of various marine organisms (Anon., 2018). The Pilbara region of the NWS is a hotspot for sponges, while the southern Kimberley and northern Pilbara are hotspots for threatened elasmobranchs, including sawfishes *Pristis* spp. and northern river sharks *Glyphis garricki* (Fromont et al., 2006; Morgan et al., 2011). Large sharks are abundant along most of the NWS coast (Letessier et al., 2019). Whale sharks *Rhincodon typus* and reef mantas *Mobula alfredi* aggregate along the Ningaloo Reef, and humpback whales *Megaptera novaeangliae* migrate southwards along the NWS from June to October each year (Commonwealth of Australia, 2012; Wilson et al., 2003). However, apart from known aggregations at a limited number of locations, research into the ecology of much of the NWS is lacking (Wilson, 2013).

Research associated with oil and gas production has provided insight into particularly understudied parts of the NWS. By 1985, the only knowledge of inshore fish fauna in the Dampier region was from environmental impact studies carried out by Woodside Petroleum and Dampier Salt (Blaber et al., 1985). Over the past decade there have been several ecological studies conducted on the oil and gas infrastructure of the NWS. These studies have recorded various threatened marine species including green sawfish *Pristis zijsron*, whale sharks, grey nurse sharks *Carcharias taurus*, and oceanic mantas *Mobula birostris* (Bond et al., 2018a; McLean et al., 2019). Novel behavioural records have also been reported, including pufferfish nests at mesophotic depths previously only observed in Japan (Bond et al., 2020), and the first wild record of pre-copulatory behaviour in leopard sharks *Stegostoma tigrinum* (Birt et al., 2019). Offshore platforms on the NWS are inhabited by diverse fish communities, including reef-dependent and pelagic species, and have been shown to support full age-structured populations of the red-belted anthias *Pseudanthias rubrizonatus* (Fowler and Booth, 2012; Pradella et al., 2014). Both platforms and pipelines also provide habitat for commercially important fish species, including saddletail snapper, goldband snapper and mangrove jack (Bond et al., 2018b; McLean et al., 2017, 2019; Pradella et al., 2014). Bond et al. (2018b) also found that diversity and abundance of fishes were both significantly higher on pipelines than in adjacent natural habitats, with biomass of commercial species 7.5 higher on the pipelines.

1.5 Aims of Research

Australia's NWS is a marine biodiversity hotspot that is also populated with various types of offshore infrastructure. There is growing evidence that these structures are important habitats for threatened marine megafauna and commercially important fish species. Research into the offshore platforms of the NWS not only increases our knowledge of the ecosystems created by offshore platforms, but also provides insight into an understudied biodiversity hotspot.

Australia's decommissioning legislation is currently under review; however little is known about the marine communities associated with the many offshore platforms in Australian waters (Taylor 2020). It is critical that the ecological roles played by Australia's offshore platforms are understood before decommissioning decisions are made, as potentially valuable ecosystems could be lost. Other regions around the world have recognised the importance of the ecosystems created by offshore platforms, and have implemented successful RTR programs to retain these ecosystems. Australia can learn from both successful and failed RTR programs, and use scientific research to make evidence-based decommissioning decisions.

The goals of this PhD dissertation are firstly to determine whether the presence of offshore platforms, both on the NWS and globally, results in the emergence of novel ecosystems; and secondly to expand our understanding of the marine communities associated with offshore platforms, and the role these platforms play in a regional context.

The key questions that my PhD will explore are:

- Can offshore platforms around the world be classified as novel ecosystems? This question is explored in Chapter 2:

  van Elden S, Meeuwig JJ, Hobbs RJ, Hemmi JM. 2019. Offshore Oil and Gas Platforms as Novel Ecosystems: A Global Perspective. Frontiers in Marine Science 6: Article 548.

- Has a novel ecosystem emerged in the Wandoo field on Australia's NWS? This question is explored in Chapter 3:

  van Elden S, Meeuwig JJ, Hobbs RJ. 2020. Offshore platforms as novel ecosystems: a case study from Australia's Northwest Shelf. Global Change Biology. In prep.

- How does ecological research around offshore platforms provide insight into animal behaviour? This question is explored in Chapter 4:

  van Elden S, Meeuwig JJ. 2020. Wild observation of putative dynamic decapod mimicry by a cuttlefish (Sepia cf. smithi). Marine Biodiversity 50:93

- Do offshore platforms create refuges for threatened elasmobranchs? This question is explored in Chapter 5:

  van Elden S, Meeuwig JJ. 2020. Elevated abundance of threatened elasmobranchs at an offshore oil field in Australia. Conservation Biology. Submitted.

1.6 Approach to thesis field studies and statistical analysis

The overall approach to this thesis is empirical, *sensu* Peters (1991), and is comparative rather than experimental. The installation of the Wandoo infrastructure represents a natural experiment (Barley and Meeuwig, 2017), as the potential emergence of a novel ecosystem was not the intent at the time of installation. This natural experiment provides an opportunity to test hypotheses with respect to offshore platforms as novel ecosystems. The Wandoo infrastructure allows for these hypotheses to be tested at ecologically relevant spatial and temporal scales that would not be feasible in a controlled experiment. True replicates of the Wandoo field do not exist, making this an “n=1 experiment”, however such unreplicated natural experiments provide unique opportunities to test hypotheses at the ecosystem scale (Barley and Meeuwig, 2017).

Field studies

The data underpinning this thesis were collected during six field surveys in the area of the Wandoo field, carried out every austral autumn and spring for three years. The Wandoo field is located 75 km northwest of Dampier, Western Australia, in waters approximately 50 m deep. The infrastructure in the Wandoo field has been in place for over 25 years, and includes: a catenary anchored leg mooring (CALM) buoy, secured by six moorings around the pipeline end manifold (PLEM); Wandoo A, an unmanned monopod platform consisting of production infrastructure with a helideck supported by a 2.5 m diameter shaft; and Wandoo B, a concrete gravity structure (CGS) with a caisson measuring 114 m long by 69 m wide, and four shafts 11 m in diameter supporting the superstructure 18 m above the surface (Fig. 1.2). Commercial and recreational vessel access is restricted in the Wandoo field, with no unauthorised entry into the 500 m petroleum safety zone surrounding all infrastructure (Commonwealth of Australia, 2010). In addition to the Wandoo field, two natural sites were also studied to provide “controls” for the platform. The first natural site, Control Sand (CS), was selected as a proxy for the Wandoo field prior to the installation of infrastructure. The region was extensively trawled in the 1970s and 1980s(Sainsbury et al., 1993), simplifying habitat from characterised by macrobenthos to ones characterised by sand. This site is located 15 km northeast of the Wandoo field, in similar water depths of approximately 50 m. The CS site is key to understanding whether a novel ecosystem has emerged in the Wandoo field, as novel ecosystems have been biotic and abiotic components different from those that prevailed historically (Hobbs et al., 2013). Comparisons between the Wandoo field and the CS site allow for assessment of the broad, ecosystem-level changes that may have occurred in the Wandoo field over the last 25 years. The second site, Control Reef (CR), was selected to compare the ecology of the artificial reef (the Wandoo infrastructure) with a comparable natural reef. This rocky reef is located approximately 15 km west of the Wandoo field, has a similar seabed footprint to the Wandoo infrastructure, and rises to around 35 m from the 50 m deep seabed. The CR site is a proxy for Wandoo under a "topping" decommissioning scenario, where the midwater portions of the structure would be removed (Dauterive, 2000). Surveys were carried out a minimum of 50 m away from any infrastructure at the Wandoo site so as to avoid collision and or entanglement with the infrastructure. To ensure consistency in data collection, surveys at the CR site were carried out a minimum of 50 m away from the reef structure.

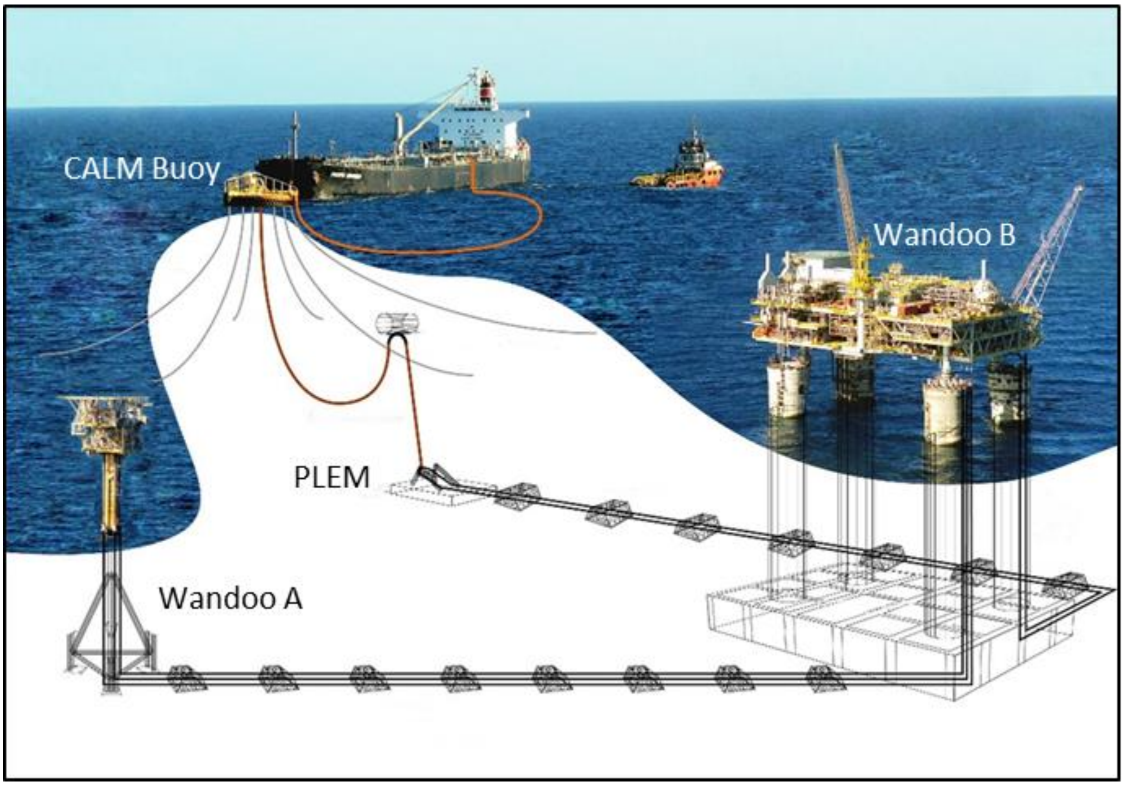

Figure 1.2 Wandoo oil field schematic adapted from Vermilion Oil and Gas Australia (2014). The infrastructure at the Wandoo field includes the unmanned monopod Wandoo A, the concrete gravity structure Wandoo B, the pipeline end manifold (PLEM), and the catenary anchored leg mooring (CALM) Buoy. Not to scale.

I chose seabed and mid-water stereo-BRUVS as the sampling method chosen for these field surveys. Stereo-BRUVS consist of two GoPro cameras mounted on a horizontal basebar, converging to a common focal point at an angle of four degrees per camera. Seabed stereo-BRUVS are baited with ~800 g of pilchards *Sardinops* sp., placed in a bag made of plastic coated wire or galvanised steel mesh. The bait bag is attached to a bait arm extending 1.5 m in front of the horizontal base bar. Seabed stereo-BRUVS are deployed on the seabed according to established field protocols (Langlois et al., 2018), and record the animals that enter the field of view over a period of one hour. Mid-water stereo-BRUVS are baited with 1 kg of crushed pilchards placed in a perforated PVC canister. This canister is fixed to the end of a bait arm extending 1.5 m from the horizontal basebar. Mid-water stereo-BRUVS are suspended ten metres below the surface and record for two hours, due to pelagic taxa generally being more sparsely distributed than demersal taxa. Mid-water stereo-BRUVS are deployed according to established field protocols (Bouchet et al., 2018).

Prior to fieldwork, stereo-BRUVS are calibrated in an enclosed swimming pool using the CAL software (SeaGIS Pty Ltd, 2020), following established calibration protocols (Harvey and Shortis, 1998). Collected videos are converted to AVI format using Xilisoft Video Converter Ultimate (Xilisoft Corporation, 2016) before being imported into the EventMeasure software package (SeaGIS Pty Ltd, 2020) for processing. A slow hand clap is recorded in the shared field of view of each stereo-BRUVS rig prior to deployment. This hand clap is used to synchronise the left and right cameras videos in the lab prior to processing. Processing commences either once the seabed stereo-BRUVS have settled on the substrate or once the mid-water stereo-BRUVS have stabilised at the set depth of 10 m. All animals are identified to the lowest possible taxonomic level. Relative abundance is estimated as the maximum number of individuals of a given species in a single frame (MaxN; Cappo et al., 2006).

Stereo-BRUVS provide a variety of data. Abundance is determined using the conservative abundance metric MaxN, which is the maximum number of individuals of a given taxa recorded in a single video frame (Cappo et al., 2006). Diversity is determined by the identification of all animals to the lowest possible taxonomic level. Not all taxa can be readily identified to species, particularly in mid-water environments where many species have counter-shaded and/or reflective colouration (Santana-Garcon et al., 2014). The stereo camera design of stereo-BRUVS allows for animals to be measured, and the weight of animals can be calculated using known length-weight relationships. The data derived from stereo-BRUVS surveys is the basis for Chapters 3-5 of this dissertation, as well as Appendix 1.

The potential effects of environmental variables and human impact on the abundance and biomass of marine fauna were also analysed. A database of physical, chemical, biological, and anthropogenic variables was compiled. Travel time variables were based on human accessibility calculations undertaken by Maire et al. (2016). Distance to market and population were calculated using the LandScan 2016 database (Dobson et al., 2000), and distances to marine features were calculated using bathymetry data following Yesson et al. (2020). Environmental data were derived from the following datasets:

- Geoscience Australia (GA) 250 m bathymetry (Whiteway, 2009);
- GA Australian submarine canyons (Huang et al., 2014);
- CSIRO Atlas of Regional Seas (CARS) (Ridgway et al., 2002); and
- Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS) (IMOS, 2020)

Statistical analysis

In this dissertation, I use a variety of statistical methods to analyse the diverse data in the thesis, depending on the nature of the data itself and the sub-hypothesis being tested. The core analyses are generally permutational, whether applied to univariate or multivariate data. Permutation-based statistical methods were chosen because they are robust to heterogeneity in the data while still maintaining statistical power (Anderson, 2017). The nature of the field surveys lend themselves to testing at the levels of year and season, and therefore most analyses are also based on permutational ANOVA. I did not use repeated measures ANOVA as the samples across the surveys were selected randomly within the stratified sampling design and were thus independent of those samples in the previous surveys (Zar, 1999). I did not use a Bonferroni correction following the advice of Armstrong (2014) as (1) I had a restricted number of planned comparisons and (2) I was more concerned about a type II error than a type I error, i.e. that a difference existed but none was detected; in other words, where a novel ecosystem had emerged but I failed to detect it. The univariate measures of abundance, biomass and fork length were $\log_{10}$ transformed to stabilise variance (Zar, 1999), and Euclidean distance matrices were calculated prior to application of PERMANOVAs (Anderson, 2017). For multivariate analyses, abundance and biomass data were log(x+1) transformed to increase the influence of rare taxa and reduce the influence of common taxa, and Bray-Curtis resemblance matrix were calculated. Multivariate analyses were visualised using canonical analysis of principal coordinates (Anderson and Willis, 2003). When analysing environmental variables, a Pearson's correlation was run to identify highly correlated independent variables with a correlation coefficient >0.6 (Havlicek and Peterson, 1976). Analyses included only one of any highly correlated variables in a given test. A distance-based linear model (DistLM) was used to determine the relationship between these variables and the assemblage data. All analyses were completed using the Primer 7 software package with the PERMANOVA+ add-on (Anderson, 2017). In some cases such as for the Wilcoxon Signed Rank tests, and the Chi-square contingency tests where the response variable was counts, the analyses were calculated by hand in Microsoft Excel (Microsoft Corporation, 2013).

1.7 Additional Information

During my PhD I compared the data I obtained from the Wandoo platform using stereo-BRUVS, with data obtained from ROV surveys by Thomas Tothill for his Master's Thesis (Tothill, 2019). This led to the development of a standardised method for using a combination of ROV and stereo-BRUVS to more effectively sample the three-dimensional habitat created by offshore platforms. This work is presented here as Appendix 1:

van Elden S, Tothill T, Meeuwig JJ. 2020. Strategies for obtaining ecological data to enhance decommissioning assessments. The APPEA Journal 60.2. 559–562

1.8 Summary

There are thousands of offshore platforms installed throughout the world's oceans, which contribute a large percentage of global energy for consumption. These platforms operate for decades before they are decommissioned, which in most cases involves the complete removal of the platform from the marine environment. However, over the course of their productive lives, offshore platforms function as artificial reefs establishing complex marine communities and acting as aggregation sites for pelagic megafauna. Some regions have recognised the value of these artificial reef structures and allow for *in situ* decommissioning in the form of RTR programs. The emergence of new marine communities around offshore platforms is congruent with the novel ecosystem concept, which recognises the potential ecological value of ecosystems altered by human activity. However, the novel ecosystem concept has only been applied to a handful of marine ecosystems and only been used twice to describe offshore platforms.

Australia's NWS is not only rich in oil and gas resources, but is also a marine biodiversity hotspot with globally significant populations of several marine species (Anon., 2018). The offshore platforms on the NWS are distributed across an area containing internationally recognised ecosystems and megafauna aggregation sites (Venables et al., 2016; Purcell and Purcell, 1988). The ecology of much of the NWS, particularly offshore, remains poorly studied. Offshore platforms represent a unique opportunity for expanding our knowledge of this diverse and productive region. Research on these platforms so far has reported endangered species, unique behaviours, and important habitats for commercial fish species. However, these potentially important ecosystems could be lost due to decommissioning without fully being understood.

This dissertation will focus on the application of the novel ecosystem concept to offshore platforms, both globally and on the NWS. Offshore platforms represent unique marine ecosystems, and I will assess offshore platform-associated communities within the context of communities in natural habitats, to determine the regional role these platforms play on the NWS. Offshore platforms provide us with the opportunity to study remote, otherwise undervalued areas of the ocean. Recent reports have found that these remote areas are ideal locations for discovering unique animal behaviours (Birt et al., 2019; Bond et al., 2020; Haugen and Papastamatiou, 2019). If Australia is to formulate a decommissioning policy that is in the best interest of the environment, it is critical that we first understand the potentially crucial ecological roles played by offshore platforms.

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CHAPTER 2 OFFSHORE OIL AND GAS PLATFORMS AS NOVEL ECOSYSTEMS: A GLOBAL PERSPECTIVE

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2.1 ABSTRACT

Offshore oil and gas platforms are found on continental shelves throughout the world's oceans. Over the course of their decades-long life-spans, these platforms become ecologically important artificial reefs, supporting a variety of marine life. When offshore platforms are no longer active they are decommissioned, which usually requires the removal of the entire platform from the marine environment, destroying the artificial reef that has been created and potentially resulting in the loss of important ecosystem services. While some countries allow for these platforms to be converted into artificial reefs under Rigs-to-Reefs programs, they face significant resistance from various stakeholders. The presence of offshore platforms and the associated marine life alters the ecosystem from that which existed prior to the installation of the platform, and there may be factors which make restoration of the ecosystem unfeasible or even detrimental to the environment. In these cases, a novel ecosystem has emerged with potentially significant ecological value. In restoration ecology, ecosystems altered in this way can be classified and managed using the novel ecosystems concept, which recognizes the value of the new ecosystem functions and services and allows for the ecosystem to be managed in its novel state, instead of being restored. Offshore platforms can be assessed under the novel ecosystems concept using existing decommissioning decision analysis models as a base. With thousands of platforms to be decommissioned around the world in coming decades, the novel ecosystems concept provides a mechanism for recognizing the ecological role played by offshore platforms.

2.2 Introduction

Since 1947, when Ship Shoal Block 32 in the Gulf of Mexico became the world's first offshore oil drilling platform (Aagard and Besse, 1973), the offshore energy industry expanded rapidly to currently number over 12,000 offshore installations globally (Ars and Rios, 2017). Offshore platforms are situated on the continental shelves of 53 countries, making offshore oil and gas production a major global industry (Parente et al., 2006). Significant advances in engineering over the last 70 years have not only increased the number of rigs, but also the environmental conditions which they can withstand: offshore platforms are now larger and found in deeper waters, further from shore. These technological advances have implications for decommissioning, which occurs when hydrocarbon production ceases or the lease ends and the platform is shut down. The decommissioning process now takes longer, requires more specialized equipment and, by extension, has become more costly (Kaiser and Liu, 2014).

A 2016 study by the IHS Markit forecast the global decommissioning of over 600 offshore structures between 2017 and 2021, with a further 2,000 projects by 2040, resulting in a total cost between 2010 and 2040 of US $210 billion (IHS Markit, 2016). In countries where total removal is the legal requirement, decommissioning involves the plugging of wells, cleaning, capping and possibly removal of pipelines, removal of production equipment and removal of the structure (Hakam and Thornton, 2000). In United Kingdom waters alone, decommissioning expenditure is forecast to amount to £17 billion between 2017 and 2025 (Oil and Gas UK, 2017). Even a nation with comparatively low oil and gas production, such as Australia (0.9% of global production), has a future decommissioning liability of US $21 billion over the next 50 years (NERA, 2016). The process of decommissioning is far from straightforward in many cases, and is often complicated by the process of transferability, whereby an existing platform is sold to a company which can continue production at lower profit margins (Parente et al., 2006).

From a biological viewpoint, increasing evidence suggests that offshore oil and gas platforms provide significant ecosystem services while active. The installation of these platforms creates hard substrate in open waters which is colonized by a variety of sessile organisms and results in the formation of artificial reefs (Shinn, 1974; Scarborough-Bull, 1989). Because they may exclude commercial fishing, particularly trawling, and in some cases recreational fishing, these platforms can also act as important refuges for a variety of taxa (Frumkes, 2002; Claisse et al., 2014). The potential ecological value of offshore platforms raises the question of whether there may be alternatives to the standard decommissioning process that might have important positive ecological outcomes, and ecological factors are more recently being included in decommissioning assessments (Fowler et al., 2014; Henrion et al., 2015; Sommer et al., 2019). The successes of various Rigs-to-Reefs projects, particularly in the Gulf of Mexico, have demonstrated that these structures can be effectively repurposed as artificial reefs (Frumkes, 2002; Kaiser and Pulsipher, 2005; Sammarco et al., 2014). However, to date only a few countries around the world have successfully implemented Rigs-to-Reefs programs (summarized in Bull and Love, 2019).

Evaluating offshore platforms as novel ecosystems would provide a mechanism for considering the ecological importance of these platforms in the decommissioning process. Novel ecosystems is a relatively recent ecological concept, brought into focus by Hobbs et al. (2006), where human activity has altered ecosystems to a point where restoration may not be feasible. In a world that is increasingly being altered by human activity, the concept of novel ecosystems recognizes that in some cases, ecosystems changed from their historical state by human intervention may not feasibly be able to be restored (Hobbs et al., 2006). With many case studies throughout a variety of ecosystems around the world (Hobbs et al., 2013b), novel ecosystems provide an approach for recognizing value in altered ecosystems, rather than implementing restoration for restoration’s sake. In the cases of both active and decommissioned platforms, it is possible that the concept of novel ecosystems can be applied as a way to describe the ecosystems created by the presence of the platforms. The aim of this review is to evaluate the ecological role of offshore oil and gas platforms, and to assess these platforms against the criteria of the novel ecosystems concept.

2.3 Decommissioning

Decommissioning, the end of life stage for offshore infrastructure, is a process which is regulated internationally, regionally and nationally. The 1996 Protocol to the London Dumping Convention (London Protocol) aimed to protect the marine environment from all sources of pollution, and regulates against the dumping of "... platforms or other man-made structures at sea; and any abandonment or toppling at site of platforms or other man-made structures at sea, for the sole purpose of deliberate disposal." (Elizabeth, 1996). However, the London Protocol does not expressly prohibit decommissioning of structures *in situ* (Techera and Chandler, 2015), stating that dumping does not include "placement of matter for a purpose other than disposal thereof, provided that such placement is not contrary to the aims of this Protocol (Elizabeth, 1996)." There are four alternatives to complete removal: (1) leave wholly in place with appropriate navigational aids; (2) partial removal, usually of the superstructure); (3) tow-and-place by moving the structure to a new location; and (4) toppling by laying the structure on its side (Schroeder and Love, 2004; Macreadie et al., 2011; Fowler et al., 2014).

Decommissioning regulations and options in various countries and regions have been reported on and assessed extensively in the literature. While decommissioning in the North Sea and the United States (US) has been well studied (e.g., Reggio, 1987; Löfstedt and Renn, 1997; Dauterive, 2000; Cripps and Aabel, 2002; Schroeder and Love, 2004; Kaiser and Pulsipher, 2005; Jørgensen, 2012; Claisse et al., 2015), there has been more recent focus on decommissioning policy in relatively "new" oil and gas producing regions, such as south-east Asia (Zawawi et al., 2012; Al-Ghuribi et al., 2016; Fam et al., 2018; Laister and Jagerroos, 2018), Australia (Fowler et al., 2015; Techera and Chandler, 2015; Chandler et al., 2017), and Brazil (Barros et al., 2017; Mimmi et al., 2017). Two recent reviews (Bull and Love, 2019; Sommer et al., 2019) provide comprehensive assessments of the literature on the decommissioning process, options, and regulations around the world. These two reviews complement each other by focusing on somewhat different aspects of decommissioning. Sommer et al. (2019) focuses on the ecosystem functions and services provided by platforms, and suggests a more ecosystemsbased approach to decommissioning. Bull and Love (2019) provides the most in-depth review to date of the literature on offshore oil and gas platforms, including platform installation, decommissioning, relevant legislation, and platform ecology. While this review is mainly focused on the United States, it does briefly review Rigs-to-Reefs programs in other regions around the world.

2.4 Rigs-to-Reefs

Rigs-to-Reefs is a potential decommissioning outcome for offshore oil and gas structures whereby obsolete infrastructure is re-purposed as artificial reefs instead of being brought back to shore for disposal (Kaiser and Pulsipher, 2005). The first examples of Rigs-to-Reefs occurred in the 1980s, when platforms were removed from production in Louisiana and transported to Florida where they were repurposed as artificial reefs (Kaiser, 2006; Jørgensen, 2009). By April 2018, approximately 532 offshore platforms have been re-purposed as artificial reefs in the Gulf of Mexico, mostly in Louisiana and Texas (Ajemian et al., 2015; Bureau of Safety and Environmental Enforcement, 2018). This represents just over 11% of the total number of platforms decommissioned in the Gulf of Mexico (Bull and Love, 2019).

Offshore oil and gas platforms are spatially complex structures and their value as artificial reefs has been discussed in numerous studies (Shinn, 1974; Dugas et al., 1979; Bohnsack and Sutherland, 1985; Guerin et al., 2007). Offshore platforms have not only been shown to have a higher fish biomass than sandy bottom areas but even natural reefs (Claisse et al., 2014). This results in offshore platforms having an “enhanced fishing zone” of 200–300 m for pelagic species and 1–100 m for demersal species (Bohnsack and Sutherland, 1985). Fishing and diving around offshore rigs, in countries where it is allowed, is a major component of the local tourism industries (Stanley and Wilson, 1989). In Louisiana, recreational fishing is centered around offshore platforms – over 70% of recreational fishing trips into the EEZ are in direct association with offshore platforms, where pelagic fish densities are 20–50 times higher than surrounding areas (Dugas et al., 1979; Reggio, 1987; Dauterive, 2000). As such, sport fishers and recreational divers generally support Rigs-toReefs programs (Frumkes, 2002).

Both active and decommissioned offshore platforms can have a negative impact on commercial trawl fishing, and the prevention of trawling is a common criticism of Rigs-to-Reefs programs (Macdonald, 1994; Hamzah, 2003). The issue of allowing fishing around platforms is one that is still uncertain and needs to be handled carefully. In some cases where platforms have become key habitat for threatened or economically important species, it may be prudent to continue to exclude all fishing from these areas if they are converted into artificial reefs, as they can then be used to bolster populations at surrounding natural reefs where fishing occurs in the same way that marine protected areas (MPAs) do (Mcclanahan and Mangi, 2000).

In sandy, flat-bottom areas with generally limited physical structure, such as the north-west shelf of Australia, the Adriatic Sea and parts of the North Sea, offshore platforms present some of the only obstacles to trawl nets (Rijnsdorp et al., 1998; Wassenberg et al., 2002; Fabi et al., 2004). While the prevention of trawling is detrimental to commercial fisheries, it is ecologically beneficial in offering protection to benthic habitats; in a study to determine the effect of trawling on sponge communities of the north-west shelf of Australia, sponges were caught in 85% of trawls, with a mean catch of 87.2 kg per half-hour (Wassenberg et al., 2002).

Evidence on the success of Rigs-to-Reefs programs and the suitability of oil platforms as artificial reef habitat suggests that these structures can provide significantly more ecological value than other cases of “dumping” (Ajemian et al., 2015). However, it is important to note that just because Rigs-to-Reefs has been successful in a certain area (e.g., the Gulf of Mexico), it does not mean it would automatically be an ecologically beneficial exercise in the North Sea, California or Australia. Every ecosystem is different and needs to be evaluated as such; creating a reef, simply because there is a platform that needs to be decommissioned, is indeed little more than waste disposal (Macdonald, 1994; Salcido, 2005).

A major obstacle in the path of Rigs-to-Reefs legislation is the relative lack of ecological research on offshore structures. For example, despite the presence of over 40 offshore oil and gas installations on the continental shelf of north-west Australia, there has been a limited number of published studies on the ecology of the structures in this region (e.g., Fowler and Booth, 2012; Pradella et al., 2014; McLean et al., 2017, 2018; Bond et al., 2018). Macreadie et al. (2012) concluded that environmental research must be part of the development of Rigs-to-Reefs policy, pointing to the case of California, where a Rigs-to-Reefs bill was vetoed in 2001 based on a lack of evidence that reefed platforms produce net environmental benefits. Macreadie et al. (2012) argue that the subsequent successful passing of a Rigs-to-Reefs bill in 2010 was due in large part to the years of subsequent research by Dr. Milton Love and colleagues (Schroeder and Love, 2002, 2004; Love et al., 2006).

2.5 Ecology of Offshore Platforms

Offshore oil and gas platforms can play important ecological roles for various taxa (Friedlander et al., 2014). They provide substrate for sessile organisms such as sponges and corals and act as a refuge for fish and megafauna such as seals and whales (Forteath et al., 1982; Todd et al., 2016). When a platform is installed, the establishment of a faunal community occurs quickly, with fish appearing within hours (Bohnsack, 1989), and ecological succession results in a complex reef-type habitat within 5–6 years (Driessen, 1986). Offshore platforms can be an important source of habitat not only for fish, but also for sessile invertebrates where hard substrate is limited. Where offshore platforms are isolated from natural reefs, the free-swimming larval stages of invertebrates that settle on offshore platforms would otherwise not likely survive due to a lack of “hospitable” substrate (Driessen, 1986; Thomson et al., 2003; Macreadie et al., 2011). However, the addition of hard substrate means that offshore platforms can also provide habitat for invasive species (Page et al., 2006; Pajuelo et al., 2016).

There is considerable debate as to whether fish associated with artificial structures are actually being produced there for a net gain, or are simply being attracted from nearby natural reefs. Attraction is thought to be detrimental to fish populations, especially those which are targeted by fisheries, as previously sparsely distributed populations become concentrated, making them vulnerable to exploitation (Bohnsack, 1989). However, in the case of offshore platforms, attraction could be beneficial to pelagic species in some regions, where the platforms can act as a temporary refuge from fishing pressure. Macreadie et al. (2011) discuss the importance of habitat limitation as a factor in the attraction vs. production debate; specifically that a habitatlimited fish population would see an increase in regional biomass due to the addition of suitable habitat via artificial structures. Fowler and Booth (2012) found that offshore platforms in northwest Australia could sustain complete size- and age-structured populations of the Serranidae *Pseudanthias rubrizonatus*, with a presumed age range in sampled individuals of 22 days to 5 years. However, production of fish varied among individual platforms. The relative scales of "attraction vs. production" therefore may vary between offshore oil and gas platforms, as biotic and abiotic conditions vary from platform to platform. The presence of larval fish may not be enough to assume production, based on the proximity of other reefs (Bohnsack, 1989; Macreadie et al., 2011). In addition, production is more important in the case of demersal species, which are more dependent on benthic habitat than highly mobile pelagic species (Bohnsack, 1989).

The ecosystem created by offshore platforms means, like natural reefs, they provide economic benefits. In regions where recreational fishing is permitted, these platforms have been highly popular locations for decades (Dugas et al., 1979). "Fishing the rigs" is a major portion of the recreational fishing activity in the Gulf of Mexico, particularly Louisiana, where species caught at the platforms include sharks, billfish, and barracuda (Driessen, 1986). While recreational fishing occurs around offshore platforms, a number of commercial gear types such as trawl and longline are generally excluded from the waters around these structures due to the risk of damage to both fishing gear and subsea infrastructure such as pipelines (de Groot, 1982; Demestre et al., 2008).

In some regions, the exclusion of all vessels, including recreational and commercial fishers, can be legally mandated, and these "exclusion zones" vary in size between countries. In the North Sea, the exclusion from fishing around offshore oil platforms that have been in place for decades, has resulted in a network of *de facto* MPAs (de Groot, 1982; Fujii and Jamieson, 2016). In Australia, the "petroleum safety zones" surrounding offshore platforms extend up to 500 m from the outer edge of any well or structure (Commonwealth of Australia, 2010), while the exclusion zone around a drilling platform in the Jubilee Field in Ghana is five nautical miles (Chalfin, 2018). In 2003, Mexico created an "area of exclusion" of 5,794 km^2 around oil platforms in the Campeche region of the Gulf of Mexico (Quist and Nygren, 2015).

Various studies have described oil platforms around the world as *de facto* MPAs. Because of the exclusion of trawl fishing at all platforms in Gabon, and the exclusion of all types of recreational fishing at some platforms due to security restrictions, Friedlander et al. (2014) concluded that these platforms are functioning as *de facto* MPAs. In California, offshore oil platforms provide a significant refuge for commercially important rockfish species (Frumkes, 2002; Claisse et al., 2014; Fowler et al., 2015). Marine vessels are discouraged from entering the 150 m buffer zone surrounding platforms, meaning that fishing activity is limited, and Schroeder and Love (2002) found that rockfish surrounding an oil platform were larger and greater in density compared with the populations at recreationally and commercially fished sites. In addition, eight offshore oil and gas platforms off southern California supported 430,000 juveniles of the highly overfished and IUCN Critically Endangered Bocaccio rockfish *Sebastes paucispinis*, accounting for 20% of the average annual number of surviving juveniles of this species. In these instances, the refuges provide much higher recruitment and survival rates than natural but fished nursery grounds (Love et al., 2006).

2.6 Novel Ecosystems

Human activities are transforming ecosystems on a global scale (Foley et al., 2005; Mccauley et al., 2015; Laurance and Watson, 2016). Many studies and conservation efforts focus on restoring altered ecosystems to their historical states (Sanchez-Cuervo et al., 2012; Graham and Mcclanahan, 2013), but over the last two decades, the term "novel ecosystems" has emerged as a way of defining ecosystems altered by human activity, where restoration is at best unlikely (Hobbs et al., 2013a). There has been criticism that the concept may exclude restoration and may provide companies a license to trash ecosystems (Aronson et al., 2014; Murcia et al., 2014). However, the novel ecosystem concept is not intended to replace ecological restoration, but is meant to provide a management option for ecosystems where restoration is not feasible or may actually result in the loss of ecosystem value (Hobbs et al., 2014). In some cases, the novel ecosystem may provide ecosystem services that are more beneficial than those provided by the historical state. Backstrom et al. (2018) have suggested that the novel ecosystems concept is most useful in a decision or management context and in terms of meeting social, ecological and economic objectives.

The term novel ecosystems was first used in 1997 (Chapin and Starfield, 1997) but was introduced into terrestrial conservation and restoration ecology fields in 2006 (Hobbs et al., 2006). The concept has more recently been adopted by some marine ecologists, where studies on marine novel ecosystems have generally focused on coral reefs which have been altered by direct human activity, disease, climate change or introduced species (Graham et al., 2013, 2015; Yakob and Mumby, 2013; Hehre and Meeuwig, 2015). However, the concept has not yet gained significant traction amongst marine ecologists. Schläppy and Hobbs (2019) provide a comprehensive decision-making framework for applying the novel ecosystems concept to altered marine ecosystems. This framework creates a mechanism for the novel ecosystems concept to be more widely applied to marine ecosystems in future. While Schläppy and Hobbs only briefly discuss offshore platforms, Sommer et al. (2019) suggest that the ecosystem-level shifts occurring around offshore platforms are “consistent with the science on... novel ecosystems.” However, while drawing parallels between offshore platforms and novel ecosystems, the authors do not explore the concept further, nor do they discuss the application of the concept to some or all offshore platforms.

The degree to which offshore platforms can usefully be considered a novel ecosystem may assist in assessing decommissioning options. Offshore platforms can be broadly assessed in a novel ecosystems context by evaluating these platforms against the criteria outlined in the most recent novel ecosystems definition from Hobbs et al. (2013b):

Criterion 1: *The abiotic, biotic and social components of the system “differ from those that prevailed historically.”* In the case of offshore oil and gas platforms, the abiotic and biotic states of the target ecosystem have clearly been altered due to anthropogenic forcing, specifically due to the installation of a large artificial structure and the associated disturbance of the ecosystem. Examples of this include the growth of cold-water corals on platforms in the North Sea (Gass and Roberts, 2006) and the aggregation of whale sharks around platforms in Qatar (Robinson et al., 2013) both of which are novel qualities not previously present in the historical state of the ecosystem.

Criterion 2: *The ecosystems have a "tendency to self-organize and manifest novel qualities without intensive human management."* In the case of offshore oil and gas platforms, the marine life associated with offshore platforms is not managed in any way, apart from limited maintenance cleaning to remove sessile invertebrates. These ecosystems persist over the lifespan of the platform, with reports of thousands of tons of invertebrate growth on the subsea structures of platforms (Foster and Willan, 1979; Culwell, 1997). Novel qualities manifested by platforms include higher productivity of algae and invertebrates (Chou et al., 1992) and higher fish biomass (Love et al., 2006).

Criterion 3: *Novel ecosystems are prevented from returning to their historical states by practical limitations, in the form of ecological, environmental and social considerations.* In the context of offshore platforms, these considerations can include many of the factors evaluated by stakeholders during the decommissioning process (Table 2.1). However, some considerations may be context specific rather than absolute, and vary among regions. For example, in California where there are relatively few platforms, their role in providing habitat for economically important species such as rockfish makes individual platforms ecologically important, particularly as some platforms produce more of these species than others (Schroeder and Love, 2002). Conversely, in an area such as the Gulf of Mexico with thousands of platforms, the ecological value of an individual platform within a regional context is not necessarily as high and therefore may not be an important ecological consideration (Schroeder and Love, 2004).

Environmental limitations could prevent the removal of offshore platforms, which means that the ecosystem cannot be returned to its historical state. Complete removal decommissioning is a potentially hazardous process both to the environment and personnel, and particularly in regions with harsh weather conditions, decommissioning could be more of a risk than leaving structures in place (Löfstedt and Renn, 1997; OGP Decommissioning Committee, 2012; Ars and Rios, 2017). Additionally, offshore platforms are known as vectors for invasive species, as they are transported long distances at low speed (Page et al., 2006; Pajuelo et al., 2016). The potential transport and spread of the many sponge, algae, coral, and even fish species associated with platforms, could be a factor preventing platform removal, and therefore restoration to historical state.

Perhaps the most significant consideration in the case of offshore platforms is the social aspect. Social factors could prohibit removal of platforms, due to prohibitive costs or platform design making removal unfeasible (Faber et al., 2001; OGP Decommissioning Committee, 2012). The social benefits derived from a platform, in the form of an artificial reef utilized by recreational divers and fishers, could be lost if the platform is removed. Conversely, social opposition to the presence of offshore platforms, as is the case in California (Pietri et al., 2011), or legislation prescribing complete removal, as is the case in Australia (Techera and Chandler, 2015) could lead to the complete removal of platforms, thereby possibly returning the ecosystem to its historical state.

It is important to avoid a blanket classification of all offshore platforms as novel ecosystems. Offshore platforms always result in the creation of habitat, but this does not by default mean that they result in novel ecosystems. For example, a platform placed near a natural reef may not significantly alter the abiotic or biotic components of the ecosystem, and may rather act simply as an “extension” of the existing reef. However, a platform placed in an area with little natural hard substrate significantly alters the abiotic nature of the ecosystem by increasing the hard substrate available, leading to changes in the community of species within the ecosystem, thereby transforming the ecosystem from its historical state.

The novel ecosystems concept can be applied to offshore platforms, so long as it is applied on a case-by-case basis. This is particularly important if the concept is used as part of the decommissioning process, as there may be incentive for energy companies to suggest platforms are novel ecosystems to avoid the costs associated with complete removal. The concept should therefore be applied conservatively and with robust evidence from ecological studies. Various studies have proposed decision analysis frameworks which assess different decommissioning alternatives based on multiple attributes (e.g., Fowler et al., 2014; Bernstein, 2015; Henrion et al., 2015). Some of these attributes can be placed within the novel ecosystems criteria as demonstrated in Table 2.1. Therefore, an assessment can be made of whether an offshore platform is a novel ecosystem simply by using existing decommissioning analysis tools. From an ecological perspective, decommissioning of offshore platforms is an ecological restoration issue. Novel ecosystems provides a tool for recognizing and retaining ecological value created through human activity, as an alternative to ecological restoration. In the same way, Rigs-to-Reefs provides the same tool, as an alternative to complete platform removal.

The decision framework for managing altered marine systems proposed by Schläppy and Hobbs (2019) would be a useful starting point for broadly classifying offshore platforms as novel ecosystems – however, because of the suite of complex, and in some cases contentious, issues surrounding oil and gas platforms, there are more factors that need to be taken into account. In this regard, the decommissioning decision analysis frameworks cited above could be used to assess a platform as a novel ecosystems even if decommissioning isn't yet being considered. For example, using the PLATFORM computer model for decommissioning analysis, Henrion et al. (2015) evaluated the impact of decommissioning options on attributes such as cost, benthic impacts, fish productivity, and water quality, all of which can be considered under novel ecosystems criterion 3 in this review.

Table 2.1 Examples from the literature of practical considerations preventing offshore platform sites from being returned to their historical state.

Practical limitations	Example	References
Ecological considerations	Refuge for endangered and/or economically important species	Love et al., 2006
	Proportion of regional hard substrate provided by the platform	Love et al., 2003
	Attraction of fish from natural habitats, making them more vulnerable to fishing	Cowan and Ingram, 1999
	Risk of environmental contamination during removal	OGP Decommissioning Committee, 2012
	Highly productive ecosystem	Claisse et al., 2014
Environmental considerations	Spread of invasive species during removal/transport	Page et al., 2006
	Environmental damage caused by use of explosives during removal process	Kaiser and Pulsipher, 2003
	Disturbance of shell mounds and remobilization of toxic chemical contaminants	Phillips et al., 2006
	Cost of decommissioning	OGP Decommissioning Committee, 2012
Social considerations	Platform design making removal unfeasible	Parente et al., 2006
	Public support for Rigs-to-Reefs programs	Kaiser and Pulsipher, 2005
	Legal frameworks prescribing complete removal	Techera and Chandler, 2015
	Public opposition to the presence of platforms	Frumkes, 2002
	Obstruction to commercial fishing	Fabi et al., 2004

2.7 Conclusion

Offshore oil and gas platforms play an ecological role for a wide variety of marine life, from corals and sponges (Gass and Roberts, 2006; Friedlander et al., 2014), to fish and sharks (Dugas et al., 1979; Schroeder and Love, 2002; Pradella et al., 2014), to marine megafauna (Robinson et al., 2013; Todd et al., 2016). At the end of their productive life, these platforms are generally removed completely and disposed of onshore, effectively removing the hard substrate and associated marine growth from an ecosystem that has developed over upward of 30–40 years (Driessen, 1986; Ferreira and Suslick, 2001). There is strong opposition to offshore drilling, and the negative perceptions of oil companies and their intentions is a big obstacle in the path of Rigs-to-Reefs programs (Löfstedt and Renn, 1997; Pietri et al., 2011). The costs of decommissioning offshore oil and gas infrastructure over the next 20–30 years run into the tens of billions of US dollars, with thousands of structures set to reach their end-of-life in this period (IHS Markit, 2016; Oil and Gas UK, 2017). In some countries, governments (and therefore taxpayers) cover some of the decommissioning costs; in the North Sea alone, this government expenditure could reach US $6.3 billion (Parente et al., 2006). Conversely, the ecosystems created by these offshore platforms have an intrinsic value in terms of fisheries, tourism, and conservation that cannot be ignored. As such, the ecological cost of decommissioning in the form of the destruction of these ecosystems must be an integral part of the decommissioning debate.

Based on the analysis of the novel ecosystems concept, many offshore oil and gas platforms can be defined as novel ecosystems, depending on a variety of factors. These platforms warrant further study, on a case-by-case basis, within the framework of novel ecosystems. This does not mean that restoration of these ecosystems should no longer be considered, as restoration may be feasible in many cases and therefore should be an option when a particular platform is to be decommissioned. However, classifying suitable offshore platforms as novel ecosystems allows for the recognition of the established, yet underappreciated, ecological value that these platforms provide.

The novel ecosystems concept can contribute to the consideration of decommissioning options using existing decommissioning decision analysis tools. Hobbs et al. (2017) proposed implementing a portfolio of approaches whereby management goals are based on the relative values of ecosystems. This approach recognizes the importance of altered ecosystems, while still allowing for conservation of high-value unaltered ecosystems. Applying this approach to decommissioning would involve identifying ecologically important platforms to be left in place for the ecosystem services they provide, while focusing decommissioning resources and effort on less ecologically valuable platforms.

One of the key arguments against novel ecosystems is that they give companies a "'license to trash' or 'get out of jail' card" (Murcia et al., 2014). This echoes the core opposition to Rigs-to-Reefs; namely that it is simply an excuse for dumping at sea (Macdonald, 1994). This argument, in both cases, ignores the potential ecological value of anthropogenically altered ecosystems. While it is undeniable that companies benefit financially from Rigs-to-Reefs programs, this does not automatically mean that these programs are environmentally detrimental. It should be possible to ensure that any Rigs-to-Reefs policy is robust and comprehensive enough to ensure that any reefing of offshore platforms will benefit the environment.

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2.9 Statements

Author Contributions

SE and JM conceived the study. SE wrote the first draft of the manuscript. All authors contributed to the manuscript revision, read, and approved the submitted version.

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Conflict of Interest Statement

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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CHAPTER 3 OFFSHORE PLATFORMS AS NOVEL ECOSYSTEMS: A CASE STUDY FROM AUSTRALIA'S NORTHWEST SHELF

Target Journal: Global Change Biology

KEYWORDS: OIL AND GAS; OFFSHORE PLATFORMS ECOLOGY; DECOMMISSIONING; STEREO-BRUVS; *DE FACTO* MPAS

3.1 ABSTRACT

The decommissioning of offshore oil and gas platforms typically involves removing some or all of the associated infrastructure and the consequent destruction of the associated marine ecosystem that has developed over decades. There is increasing evidence of the important ecological role played by offshore platforms. Concepts such as novel ecosystems allow stakeholders to consider the ecological role played by each platform in the decommissioning process. This study focused on the Wandoo field in Northwest Australia as a case study for the application of the novel ecosystem concept to the decommissioning of offshore platforms. Stereo-baited remote underwater video systems were used to assess the habitat composition and fish communities at Wandoo, as well as two control sites: a sandy one that resembled the Wandoo site pre-installation, and one characterised by a natural reef as a control for natural hard substrate and vertical relief. We found denser macrobenthos habitat at the Wandoo site than at either of the control sites, which we attributed to the exclusion of seabed trawling around the Wandoo infrastructure. We also found that the demersal and pelagic taxonomic assemblages at Wandoo more closely resemble those at a natural reef than those which would likely have been present historically, but these assemblages are still unique in a regional context. The demersal assemblage is characterised by reef-associated species with higher diversity than those at the other sites, with the pelagic community characterised by species associated with oil platforms globally. These findings suggest that a novel ecosystem has emerged in the Wandoo field. It is likely that many of the novel qualities of this ecosystem would be lost under decommissioning scenarios that involve partial or complete removal. This study provides an example for classifying offshore platforms as novel ecosystems.

3.2 INTRODUCTION

Offshore oil and gas platforms (hereafter offshore platforms) have been a feature of continental shelf waters for over 70 years, with nearly 12,000 of these structures currently installed around the world (Aagard and Besse, 1973; Ars and Rios, 2017). When an offshore platform is no longer economically viable, a decision is made on the fate of the structure through a process referred to as decommissioning. In most cases, decommissioning involves complete removal of the platform from the marine environment for scrapping or recycling on land (Schroeder and Love, 2004). Complete removal is legislated as the default decommissioning method in many countries and regions, including Australia and the North Sea, as well as internationally under the United Nations Convention on the Law of the Sea (UNCLOS) and the 1996 Protocol to the London (Dumping) Convention (Chandler et al., 2017; Elizabeth, 1996; Techera and Chandler, 2015). However, the London Convention does permit *in situ* decommissioning for purposes other than disposal, and some regions have legislated such methods. In the Gulf of Mexico, platforms can be left either wholly or partially in place, or towed to a new location, under a program known as Rigs-to-Reefs (RTR, Reggio, 1987). Offshore platforms have been shown to form highly complex artificial reefs (Shinn, 1974), and RTR programs represent a method for preserving and maintaining these artificial reef communities that are established around offshore platforms over the decades they spend in the ocean, similar to the reefs formed by shipwrecks (Dauterive, 2000; Leewis et al., 2000).

Offshore platforms play various ecological roles, including acting as aggregation sites for marine megafauna (Haugen and Papastamatiou, 2019; Robinson et al., 2013), nurseries for juvenile fishes (Love et al., 2019; Nishimoto et al., 2019), and providing habitat for economically important and overfished species (Bond et al., 2018a; Love et al., 2006). The presence of these offshore platforms creates new habitat, which can have a significant impact on fish production; platforms in California are some of the most productive fish habitats in the world, and platforms in Gabon have higher fish biomass than pristine reefs in the Pacific (Claisse et al., 2014; Friedlander et al., 2014). Fishing is excluded around offshore platforms in many countries, either by law as is the current case in Australia (Commonwealth of Australia, 2010), or by the presence of subsea infrastructure which can damage fishing equipment (de Groot, 1982). The partial or complete exclusion of fishing effectively creates *de facto* marine protected areas (MPAs) around offshore platforms (de Groot, 1982; Friedlander et al., 2014). The exclusion of fishing is particularly important in areas which are overfished, or where hard substrate is limited and infrastructure may be some of the only obstacles to trawling (de Groot, 1982; Fujii and Jamieson, 2016; Love et al., 2006; Schroeder and Love, 2002).

There is an increasing research focus around the world on the potential ecological importance of offshore platforms, and particularly on ensuring that the role of these platforms as ecosystems is considered in the decommissioning process (Bull and Love, 2019; Fowler et al., 2014, 2018; Macreadie et al., 2012). An ecological perspective of offshore platforms allows scientists to apply restoration principles to the decommissioning process, in a similar way to terrestrial restoration of abandoned mine sites (Koch and Hobbs, 2007). The presence of offshore platforms modifies communities and habitats to such an extent that returning the site to its pre-installation state may no longer be feasible or preferable (Sommer et al., 2019) and as such, the benefits of *in situ* decommissioning must be evaluated.

This assertion is congruent with the concept of novel ecosystems, which is intended to complement existing restoration practices by providing a management option for ecosystems where restoration is, at best, unlikely, or could result in lost ecosystem value (Hobbs et al., 2013). Recently there have been attempts to apply restoration management concepts to offshore platforms in terms of: establishing ecological baselines for restoring the ecosystem post-decommissioning (Fortune and Paterson, 2020); the potential for restoration paradigms to shift the discourse surrounding RTR decommissioning (Ounanian et al., 2019); and direct application novel ecosystems criteria to offshore platforms (Schläppy and Hobbs, 2019; van Elden et al., 2019).

There is still a significant knowledge gap around the ecology of these platforms, particularly outside of the major northern hemisphere oil and gas producing regions. In particular, only a limited number of studies exist on the fish and shark communities around offshore infrastructure in Australia (e.g. Bond et al., 2018a, 2018b; Fowler and Booth, 2012; McLean et al., 2019; Pradella et al., 2014). Information on how ecological value is retained under varying decommissioning scenarios is needed at a time when the Australian government is reviewing legislation to potentially allow *in situ* decommissioning options (Offshore Resources Branch, 2018; Taylor, 2020). It is critical that we understand the ecological role platforms play in a regional context before the associated ecosystems are potentially lost due to decommissioning and restoration activity.

The offshore oil and gas producing region of northwest Australia, the Northwest Shelf (NWS), is comprised of over 40 production facilities and over 2,000 km of subsea pipelines (Bond et al., 2018b; Geoscience Australia, 2009). This is not a large number of platforms when compared with other locations around the world. However, the NWS is largely devoid of any significant natural hard substrate, and therefore offshore platforms contribute a significant portion of such habitat regionally, along with its associated fishes. This area was historically characterised by established macrobenthos communities made up of sponges, gorgonians and soft corals on flat, sand inundated pavement (Evans et al., 2014). These macrobenthos communities were largely removed by pair-trawling operations in the 1960s and 1970s (Fromont et al., 2016; Sainsbury et al., 1997). Previous studies on both platforms and pipelines on the NWS have found significant macrobenthos habitat associated with these structures, and abundance and richness of fish was higher on pipelines than on nearby natural habitats (Bond et al., 2018a, 2018b, McLean et al., 2018, 2019). These results suggest that the hard substrate provided by oil and gas infrastructure may modify the habitat and associated communities from their historical state.

We investigate whether the presence of active offshore infrastructure at a site in northwest Australia has resulted in the emergence of a novel ecosystem, characterised by a shift in the structure of marine communities. Demersal and pelagic taxonomic assemblages, as well as macrobenthos communities, were documented around the infrastructure in the Wandoo oil field (Wandoo) over three years and six surveys and in contrast to two control sites: a sandy site (Control Sand); and a natural reef (Control Reef). Baseline (pre-installation) ecological information for the Wandoo site was not collected, as has been the case for many older offshore platforms (Fortune and Paterson, 2020). As such, the Control Sand site acts as a proxy for the historical state of the Wandoo site. The Control Reef site is characterised by a rocky substrate with significant physical relief, and similar in spatial extent to the infrastructure in the Wandoo field. Control Reef provides contrast to the Wandoo site in the form of a comparable natural reef. These two sites allowed us to both assess Wandoo as a novel ecosystem and predict how the marine communities would be altered under two different decommissioning scenarios. Specifically, complete removal may see the Wandoo site revert to a state more similar to the Control Sand site, and partial removal (topping) may lead to something more similar to the Control Reef site. We used baited remote underwater video systems (BRUVS) to determine how taxonomic richness, abundance, biomass, fork length, and community assemblage structure varied between these sites, as well as intra- and inter-annually. We hypothesised that the Wandoo field infrastructure had become an established artificial reef over the 25 years since installation, resulting in the emergence of a novel ecosystem with a unique marine community.

3.3 Materials and Methods

Study sites

The three sites sampled are located in the Northwest Shelf (NWS) region of northwest Australia, approximately 75 km northwest of Dampier, Western Australia (Fig. 3.1). The sites are all situated in waters approximately 50-60 m deep. The Wandoo site (WN) is an active oil field leased by Vermilion Oil and Gas Australia Pty Ltd (Vermilion). This site contains oil production infrastructure including: Wandoo A, an unmanned monopod wellhead platform with a 2.5 m diameter shaft supporting a helideck and production infrastructure; Wandoo B, a concrete gravity structure (CGS) made up of a 114 m long by 69 m wide caisson and four shafts, each 11 m in diameter, supporting the superstructure approximately 18 m above the sea surface; and a catenary anchored leg mooring (CALM) buoy, with six moorings and a Pipeline End Manifold (PLEM) below the buoy (Fig. 3.2). The infrastructure at the Wandoo site is surrounded by a 500 m exclusion zone, within which only authorised vessels are permitted to operate (Commonwealth of Australia, 2010). Two control sites were also sampled: a “historical” flat sand-dominated site, Control Sand (CS) comparable to the Wandoo site prior to infrastructure installation in 1994; and a reef site, Control Reef (CR) that is a natural structure comparable in dimension to the Wandoo infrastructure.

The CS site is situated approximately 15 km northeast of the Wandoo site (Fig. 3.1) and is characterised by little to no physical relief and a dense, silty sand habitat. The CR site is located approximately 15 km west of the Wandoo site (Fig. 3.1) and is characterised by a rocky reef, similar in spatial extent to the infrastructure in the Wandoo field, rising to approximately 20 m below the surface. Unlike the WN site, the CS and CR sites are accessible to commercial and recreational fishing.

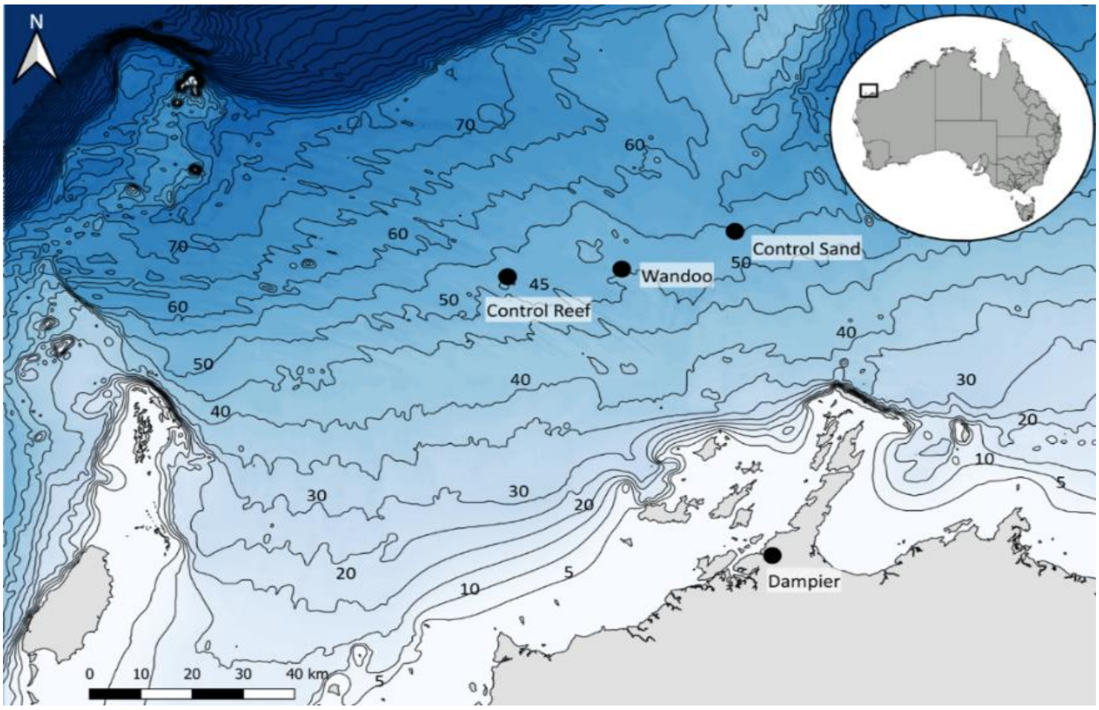

Figure 3.1 Location of the three study sites, Wandoo, Control Reef and Control Sand, approximately 75 km north-west of Dampier, Western Australia

Stereo-baited underwater video systems

Stereo-BRUVS are a non-destructive, cost-effective method for studying marine fauna (Cappo et al., 2006; Letessier et al., 2013, 2015b). They have been used to study abundance, biomass, diversity, distribution and behaviour in animals ranging from fish and sharks, to turtles, moray eels, and marine mammals (Barley et al., 2016; Letessier et al., 2015a; Spaet et al., 2016; Thompson et al., 2019; Whitmarsh et al., 2017). Seabed stereo-BRUVS have been adapted to mid-water environments, making them a useful tool for documenting highly mobile and elusive species (Bouchet et al., 2018; Letessier et al., 2013; Thompson et al., 2019). BRUVS-derived data should be interpreted recognising the potential impact of variable bait plumes (Whitmarsh et al., 2017), the potential higher representation of piscivores, and the relative nature of abundance estimates in contrast with density estimates generated by, for instance, UVC (Langlois et al., 2010). However, despite these constraints, BRUVS can be used to document clear signals in marine communities relative to other methods (Cappo et al., 2006; Lowry et al., 2012).

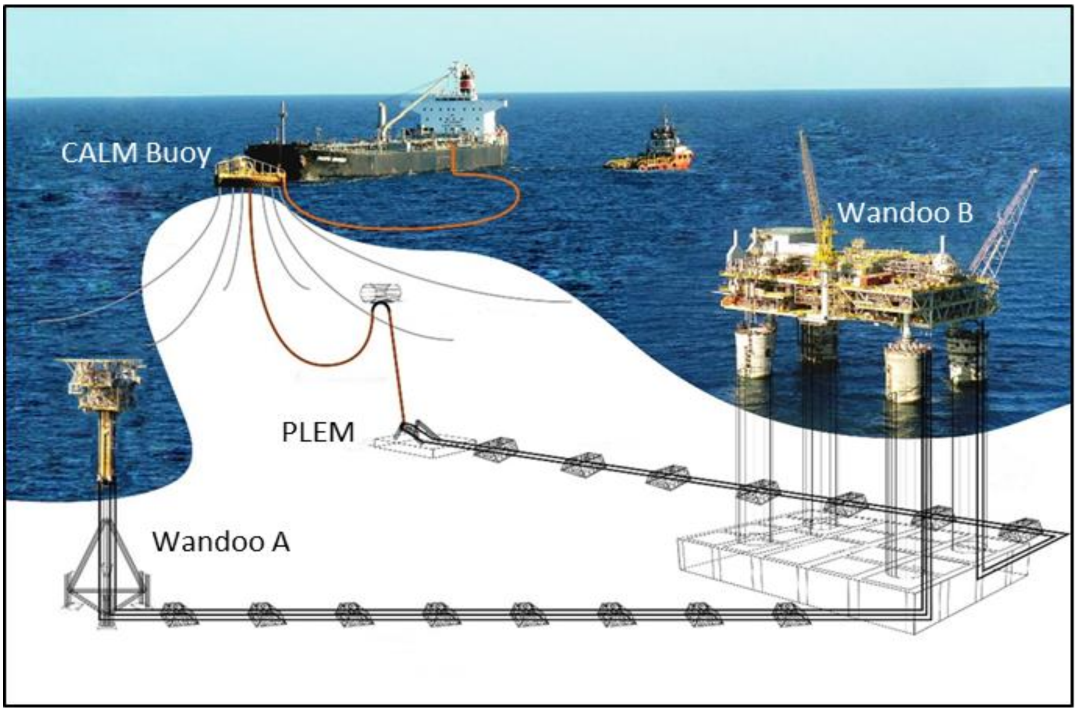

Figure 3.2 Wandoo oil field schematic adapted from Vermilion Oil and Gas Australia (2014). The infrastructure at the Wandoo field includes the unmanned monopod Wandoo A, the concrete gravity structure Wandoo B, the pipeline end manifold (PLEM), and the catenary anchored leg mooring (CALM) Buoy. Not to scale

Seabed stereo-BRUVS consist of two GoPro cameras mounted 80 cm apart on a horizontal base bar, each converging at an angle of four degrees to a common focal point. A galvanised steel mesh bait cage containing 800 g of crushed pilchards is attached to the end of a 1.5 m long bait arm (Supplementary Fig. 3.1a). Seabed stereo-BRUVS are deployed at least 200 m apart for a minimum of 60 minutes.

Mid-water stereo-BRUVS consist of the same horizontal base bar as seabed stereo-BRUVS, mounted on a 1.45 m long steel upright to provide stability, and suspended 10 m below the surface (Supplementary Fig. 3.1b). They are baited with 1 kg of crushed pilchards in a perforated bait canister on a 1.5 m long bait arm, which acts as a rudder to keep the cameras facing down-current for the duration of the deployment. Mid-water stereo-BRUVS are deployed for a minimum of 120 minutes, and in this study, are anchored to prevent entanglement with subsea infrastructure.

Data collection

Sampling was undertaken over three years, from 2017-2019, with twice-yearly expeditions in the austral autumn and spring. Due to the significant tide range and variable weather conditions in the region, surveys were limited to a ten day window over neap tides. In most of the surveys, it was only possible to sample two of the three study sites, and the three sites were therefore not sampled evenly between years and seasons. The WN site was sampled in both autumn and spring in all three years. The CR site was sampled in autumn and spring of 2017, autumn of 2018 and spring of 2019, while the CS site was sampled in autumn and spring of 2018 and autumn of 2019.

A total of 595 seabed stereo-BRUVS and 530 mid-water stereo-BRUVS samples were collected over the three year study period, using a random stratified sampling design. At the WN site, ten “near” sampling zones were established around the infrastructure, with a further four ‘far’ zones established at least 1 km from the infrastructure. Seabed stereo-BRUVS were deployed in the ten zones around the structure, while mid-water stereo-BRUVS were deployed in a subset of five of these zones, as well as the four remote zones, to reflect the highly mobile nature of pelagic fauna. All stereo-BRUVS were deployed a minimum of 50 m away from any infrastructure at the Wandoo site so as to avoid collision and or entanglement between the stereo-BRUVS and the infrastructure. To ensure consistency in data collection, stereo-BRUVS were deployed a minimum of 50 m away from the reef structure at the Control Reef site. All sampling was carried out during daylight hours to minimise the effect of crepuscular animal behaviour. The sampling was conducted under UWA ethics permit RA/3/100/1484.

Data processing and treatment

Prior to each survey, individual stereo-BRUVS were calibrated in an enclosed pool, according to standard protocols, using the CAL software (Harvey and Shortis, 1998; SeaGIS Pty Ltd, 2020). All video samples collected in the field were converted to AVI format using Xilisoft Video Converter Ultimate (Xilisoft Corporation, 2016) and videos were processed using the Eventmeasure software package (SeaGIS Pty Ltd, 2020). Processing commenced either once seabed stereo-BRUVS had settled on the seabed, or when the mid-water stereo-BRUVS had stabilised at 10 m depth following deployment. All animals entering the field of view were identified to the lowest possible taxonomic level, and abundance was estimated using the conservative abundance metric MaxN, which is the maximum number of individuals of a given taxon in a single frame (Cappo et al., 2006). The appropriate length metric (e.g. fork length FL, disc width DW, or carapace length CL) was measured in stereo on as many individuals as possible within the MaxN frame. For seabed stereo-BRUVS, the habitat visible in the field of view was broadly categorised into three groups: sand (bare substrate with no visible macrobenthos or other marine growth); sparse macrobenthos (predominantly bare substrate with less than 50% biotic cover); and dense macrobenthos (the visible substrate was dominated by more than 50% biotic cover).

The video analysis yielded identification, abundance, and length data for each stereo-BRUVS deployment. These data were analysed as taxonomic richness (TR), total abundance (TA) and fork length (FL) respectively. Total biomass (TB) was calculated as the sum of mean weight of a given taxa on a given sample. Weight was calculated based on FL using taxon-specific length weight relationships (LWR) sourced from Fishbase (Froese and Pauly, 2019). Where the LWR was not available for a particular taxon, the LWR based on total length (TL) for that taxon was used, in combination with taxon-specific TL:FL conversions. Where an animal was identified to genus or family, the Bayesian LWR was sourced from Fishbase (Froese et al., 2014). Taxon-specific biomass estimates were calculated by multiplying the abundance of each taxon by the mean weight of that taxon. Marine mammals were excluded from the biomass estimates as they were multiple orders of magnitude heavier than the largest observed fish and heavily skewed the estimates. These four univariate metrics, TR, TA, TB and FL, were analysed separately for each survey in order to ensure like-for-like comparisons between sites. Annual and seasonal variability were also assessed for each site to determine the variability in the demersal communities at each site over time. These analyses were also carried out at the level of survey, comparing annual variability separately for autumn and spring at each site, and seasonal variability (i.e. between spring and autumn) for each year at each site. For seabed stereo-BRUVS, all attributes were reported for each individual rig deployment. For mid-water stereo-BRUVS, all attributes were averaged for each set of five BRUVS deployed in a zone. This method mitigates the potential effect of highly mobile pelagic species being observed on multiple rigs.

The prevalence of each taxa at each site was calculated by determining the percentage of seabed deployments or midwater zones on which the particular taxa was observed of the total for that site. The prevalence data were then used to determine the number of unique demersal and pelagic taxa for each site, by extracting taxa that were only recorded at one site. We did not count taxa which were recorded on only one midwater zone or seabed deployment per site, in order to eliminate chance sightings and possible incorrect identifications. Within the lists of unique taxa, any taxon that was only identified to genus or family was removed if there was a record from that genus or family at another site.

Statistical analyses

The categorised habitat data were analysed using a Chi-square contingency test to determine whether habitat varied significantly by site (Zar, 1999). Variation in the fish assemblage was tested using PERMANOVA as it is robust to data heterogeneity (Anderson, 2017). The linear variables of TA, TB and FL were $\log_{10}$ transformed to stabilise variance (Zar, 1999). For each of these univariate measures, a Euclidean distance resemblance matrix was calculated and a PERMANOVA was applied based on unrestricted permutations (Anderson, 2017) with Site and Survey as fixed factors. Our main hypothesis was whether sites differed in their fish assemblages and the degree to which such differences varied temporally. To first determine whether sites differed, one-way pair-wise PERMANOVAs were applied within each survey period. We also similarly tested for differences between years and between seasons within sites. Repeated measures ANOVA was not used as the sampling through space and time varied randomly within the zones and seasons (Zar, 1999).

The assemblage composition data were treated differently to the univariate metrics. Species composition data were pooled across all surveys in preparation for the multivariate analyses for each sampling method. The data were analysed by survey to ensure like-for-like comparisons between sites. Multivariate analyses were completed on the pelagic and demersal taxonomic assemblage data in terms of abundance and biomass, to understand variations in species composition between sites as well as which variables explained this variation. We log(x+1) transformed the assemblage data and calculated Bray-Curtis resemblance matrices for abundance and biomass of each species. Pairwise PERMANOVAs were applied to determine the differences between the demersal and pelagic species compositions of the three sites, across all surveys, in terms of both abundance and biomass. Canonical analysis of principal coordinates (CAP) was used in order to visualise a constrained ordination of the data on the basis of distance or dissimilarity.

A database of physical, chemical and biological variables was also compiled in order to understand the potential environmental effects on taxonomic assemblages. Distances to marine features were calculated using bathymetry data following Yesson et al. (2020). Environmental data were derived from the following datasets:

- Geoscience Australia (GA) 250 m bathymetry (Whiteway, 2009);
- GA Australian submarine canyons (Huang et al., 2014);
- CSIRO Atlas of Regional Seas (CARS) (Ridgway et al., 2002); and
- Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS) (IMOS, 2020)

A number of anthropogenic variables were also calculated based on human accessibility calculations undertaken by Maire et al. (2016), distance to market and population using the LandScan 2016 database (Dobson et al., 2000). However, the three sites are almost exactly the same distance from the coast, so distance-based variables were similar for all sites, and fishing effort data was not fine-scale enough to separate the three sites. As such, the anthropogenic variables were excluded.

A Pearson's correlation was run to identify highly correlated independent variables with a correlation coefficient >0.6 (Havlicek and Peterson, 1976). Analyses included only one of any highly correlated variables in a given test. A distance-based linear model (DistLM) was used to determine the relationship between these variables and the assemblage data across all surveys. All analyses were completed using the Primer 7 software package with the PERMANOVA + add-on (Anderson et al., 2015).

3.4 Results

In the six surveys across three years, we counted 35,070 individuals from 358 taxa, representing 85 families (Supplementary Tables 3.5 and 3.6). The total biomass of these animals was 42.5 tonnes, excluding marine mammals. Of the 358 taxa, 252 (70%) were unique to the demersal samples, 44 (13%) were unique to the pelagic samples, and 62 (17%) of the taxa were common to both sets of samples. Fork length of demersal taxa ranged from a 2 cm unidentified juvenile to a 260.4 cm wedgefish *Rhynchobatus* sp. Three families accounted for 57% of all demersal animals recorded: jacks (Carangidae; 32%), threadfin breams (Nemipteridae; 14%), and damselfishes (Pomacentridae; 11%), while the most prevalent demersal species was the starry triggerfish *Abalistes stellatus*, occurring on 91% of deployments. Pelagic taxa ranged in fork length from a 0.86 cm juvenile leatherjacket Monacanthidae sp., to a 6.27 m northern minke whale *Balaenoptera acutorostrata*, with the largest fish being a 3.93 m tiger shark *Galeocerdo cuvier*. Two families accounted for 79% of all pelagic animals recorded: herrings (Clupeidae; 40%) and jacks (Carangidae; 39%). The most prevalent pelagic taxon was scads *Decapterus* sp., occurring on 72% of deployments. Threatened species included two Critically Endangered taxa, wedgefishes *Rhynchobatus* sp. and great hammerhead *Sphyrna mokarran*, and two Endangered species, dusky shark *Carcharhinus obscurus* and zebra shark *Stegostoma tigrinum* (Dudgeon et al., 2019; Rigby et al., 2019a, 2019b).

Environment

Observed habitats across the three sites included sand, and macrobenthos which consisted of sponges, sea whips, crinoids, soft corals, and gorgonians. Macrobenthos coverage was both sparse (<50%) and dense (>50%). Habitat differed significantly across the three sites with the WN site characterised by a higher percentage of samples dominated by dense and sparse macrobenthos relative to the other two sites, ($X^2_{(2,\ N = 417)} = 91.1$, $p < 0.001$). Macrobenthos was present on 57% of the deployments at WN, with sand dominating deployments at CR and CS (60% and 99% respectively; Fig. 3.3). The highest percentage of dense macrobenthos also occurred at WN (22%), compared with 15% at CR and none at CS.

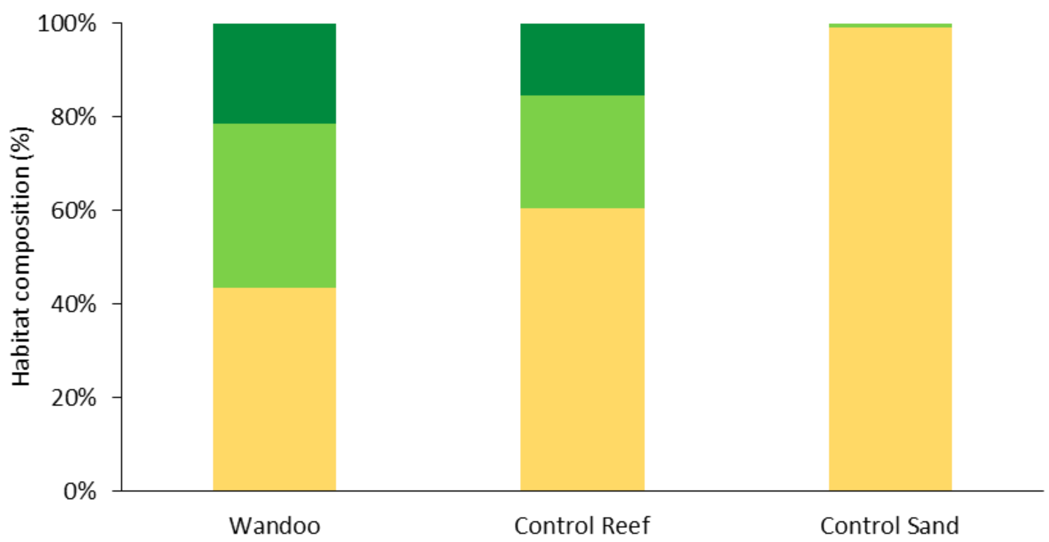

Figure 3.3 Percentage habitat composition for each of the three sites. The habitat types are sand (yellow), sparse macrobenthos (light green) and dense macrobenthos (dark green).

There was limited environmental variability between the sites. As expected based on sampling design, depth was not significantly different between WN and CR (52.5 m ± 0.116 SE and 52.9 m ±0.274 SE respectively; t_{468}=1.87, p=0.06), although CS was significantly but only marginally deeper (54.9 m ±0.129 SE) than the WN and CS (t_{418}=16.8, p< 0.001 and t_{298}=7.87, p< 0.001 respectively). Mean sea surface temperature (SST) in autumn was similar at WN and CR (29.0 °C ± 0.093 SE and 28.8 °C ± 0.148 SE respectively; t_{218}=1.18, p=0.26), but was approximately one degree higher at CS (29.9 °C ± 0.105 SE) than at WN and CR (t_{218}=6.15, p<0.001 and t_{148}=6.19, p<0.001 respectively; Supplementary Table 3.2). Mean SST in spring did not differ significantly between WN and CS (23.9 °C ± 0.057 SE and 23.7 °C ± 0.003 SE respectively; t_{198}=1.30, p=0.21), but was significantly higher at CR (24.1 °C ± 0.086 SE) than at WN and CS (t_{248}=2.08, p=0.038 and t_{148}=2.75, p=0.007 respectively). Mean chlorophyll concentration (Chl-a) in autumn was higher at WN than CR and CS (0.475 mg/m^3 ± 0.024 SE; t_{218}=3.34, p= 0.003 and t_{218}=2.62, p=0.002 respectively), with no difference between the latter two sites (0.355 mg/m^3 ± 0.016 SE and 0.367 mg/m^3 ± 0.031 SE respectively; t_{148}=0.39, p=0.71). In spring, mean Chl-a was significantly higher at CR than both WN and CS (0.436 mg/m^3 ± 0.020 SE; t_{248}=7.84, p<0.001 and t_{148}=4.55, p<0.001 respectively), with no significant difference between WN and CS (0.293 mg/m^3 ± 0.007 SE and 0.307 mg/m^3 ± 0.001 SE respectively; t_{198}=1.21, p=0.22).

Demersal richness, abundance, biomass and fork length

The mean demersal richness was 13.1 ±0.90 SE and ranged between 7.7 and 17.5 taxa per sample. There was significant variation in richness between sites in four of the six surveys (Fig. 3.4a; Table 3.1): richness was higher at WN than CR in Spring 2017; CR and WN were both significantly higher than CS in Autumn 2018; and WN was higher than CS in Spring 2018 and Autumn 2019. There was no variation in richness between years at WN, and the only seasonal variation was in 2018, when richness was higher in autumn than spring (Supplementary Table 3.3). In comparison, there was no annual variation in richness at CR, while at CR richness was higher in 2018 than 2019. There was no seasonal variation at these two sites (Supplementary Table 3.3).

Abundance ranged from 17.9 to 77.4 individuals, with a mean of 43.5 ±4.99 SE. Abundance was consistent between sites in most surveys, only differing in Autumn 2019 when abundance at WN was higher than at CS (Fig. 3.4b; Table 3.1). In terms of annual variability at WN, abundance in autumn was higher in both 2017 and 2018 than in 2019. In spring, abundance was also higher in 2017 than both 2018 and 2019. There was no seasonal variation in abundance at WN in 2017, while abundance was higher in autumn than in spring in both 2018 and 2019 (Supplementary Table 3.3). In comparison, at CR there was also no annual variation in abundance, while at CS abundance was higher in 2018 than 2019. Seasonal variation at both of these sites followed the same trend as WN, with abundance being higher in autumn than spring (Supplementary Table 3.3).

Mean biomass was 44.5 kg ±3.31 SE, and ranged from 28.2 kg to 70.2 kg. Similarly to abundance, biomass was consistent between sites for all surveys except Autumn 2019, when biomass was higher at WN than CS (Fig. 3.4c; Table 3.1). There was no annual or seasonal variation in biomass at WN. In comparison, there was no annual variation in biomass at CR, while biomass was higher in 2018 than 2019 at CS. Both control sites showed significant differences in biomass between seasons, with biomass higher in autumn than spring (Supplementary Table 3.3).

Fork length ranged from 24.8 cm to 38.5 cm, with a mean of 32.6 cm ±1.05 SE. Fork length was higher at WN than CR in Autumn 2017, while in Autumn 2018, WN was higher than both control sites, and CR was higher than CS (Fig. 3.4d; Table 3.1). Fork length was consistent between sites in all other surveys. In terms of annual variability at WN, in both autumn and spring fork length was consistent between 2017 and 2018, and higher in 2019 than both other years. In terms of seasonal variability, fork length was higher in spring than autumn in both 2017 and 2018, with no significant difference in 2019 (Supplementary Table 3.3). In comparison, fork length in autumn was consistent between 2017 and 2018 at CR, but higher in 2019 than 2018 at CS, while in spring fork length was consistent at CR. There was no seasonal variation at CR, while at CS fork length was higher in spring than autumn (Supplementary Table 3.3).

Pelagic richness, abundance, biomass and fork length

Mean pelagic richness was 3.9 ± 0.15 SE, with a range of 1.3 to 8.4 taxa per zone. Richness in the Autumn 2018 survey was significantly higher at WN and CS than at CR, but was consistent between sites in all other surveys (Fig. 3.4e; Table 3.1). There was no annual variation in richness at Wandoo in either autumn or spring, and there was only seasonal variation in 2017, when richness was higher in autumn than spring (Supplementary Table 3.4). In comparison, richness at CR was higher in 2017 than 2018 in autumn and higher in 2019 than 2017 in spring, while there was no annual variation at CS. Richness was also higher in autumn than spring at CR, but there was no seasonal variation at CS (Supplementary Table 3.4).

Abundance ranged from 1.3 to 271 individuals per zone, with a mean of 29.6 ± 4.7 SE. Abundance was significantly higher at WN than CR in both Autumn 2017 and Autumn 2018, and also higher at CS than CR in Autumn 2018. Abundance did not differ significantly in any other survey (Fig. 3.4f; Table 3.1). In terms of annual variability at WN, abundance in autumn was consistent between all years, and consistent in spring between 2017 and 2019, but higher in spring in 2018 than both other years. In terms of seasonal variation, abundance was consistent between seasons in 2018, but higher in autumn than spring in both 2017 and 2019 (Supplementary Table 3.4). In comparison, abundance was higher in autumn in 2017 than 2018 at CR, but did not differ between years at CS. There was also no difference in abundance between years in spring at CR. In terms of seasonal variation, autumn was higher than spring at CR but there was no significant difference between seasons at CS (Supplementary Table 3.4).

Mean biomass was 48.5 kg ± 5.7 SE and ranged from 7.5 g to 429 kg. Biomass was significantly lower at CR than CS in Autumn 2018, but was consistent between sites across all other surveys (Fig. 3.4g; Table 3.1). There was no annual or seasonal variation in biomass at WN. In comparison, biomass was higher in autumn in 2017 than 2018 at CR, but there was no other significant annual or seasonal variation at the control sites (Supplementary Table 3.4).

Fork length ranged from 3.8 to 182 cm, with a mean of 37.5 cm ± 3.3 SE. Fork length was higher in Autumn 2017 at WN than CR, and higher in Autumn 2018 at CS than WN. Fork length was consistent between sites in all other surveys (Fig. 3.4h; Table 3.2). In terms of annual variability at WN, fork length was higher in autumn in 2017 than 2018, and higher in spring in 2019 than both 2017 and 2018, but otherwise did not differ between years. In comparison, there was no annual or seasonal variation in fork length at either control site (Supplementary Table 3.4).

Community assemblages

The composition of the demersal and pelagic taxonomic assemblages were analysed in terms of both abundance (Fig. 3.5) and biomass (Supplementary Fig. 3.2). In all cases, we saw strong separation of assemblages by site, with abundance and biomass at each site characterised by unique species assemblages. Demersal and pelagic taxonomic assemblages were significantly different from each other at all sites, in terms of both abundance and biomass (Table 3.2). The DistLM analysis showed that the three environmental variables, depth, SST and Chl-a, did not explain a sufficient proportion of the variance in the assemblage data and as such, these analyses were excluded.

Demersal abundance (Fig. 3.5a) and biomass (Supplementary Fig. 3.2a) at WN were driven by reef-associated species, namely galloper *Symphorus nematophorus* and spot-cheek emperor *Lethrinus rubrioperculatus*. Both species usually occurred in low abundance but were prevalent across deployments at WN (38% and 40% respectively; Supplementary Table 3.5). Abundance and biomass at CR were driven by different reef-associated species than at WN, namely bluespotted emperor *Lethrinus punctulatus* (the name most commonly used for this unresolved species; Moore et al., 2020) and turrum *Carangoides fulvoguttatus*, both of which occurred in large schools, while biomass was also driven by areolate grouper *Epinephelus areolatus*, a more solitary species. There was some overlap in taxonomic assemblages between WN and CR, driven by bluespotted tuskfish *Choerodon cauteroma*. Abundance at CS was characterised by northwest blowfish *Lagocephalus sceleratus*, a species associated with offshore reefs and sandy habitats, and brushtooth lizardfish *Saurida undosquamis*, a sand or mud bottom associated species. These species occurred in relatively low numbers but were highly prevalent on deployments at this site (51% and 80% respectively; Supplementary Table 3.5) Brushtooth lizardfish also characterised biomass at CS, along with the milk shark *Rhizoprionodon acutus*, also associated with sandy habitats. Habitat associations were sourced from Fishbase (Froese and Pauly, 2019).

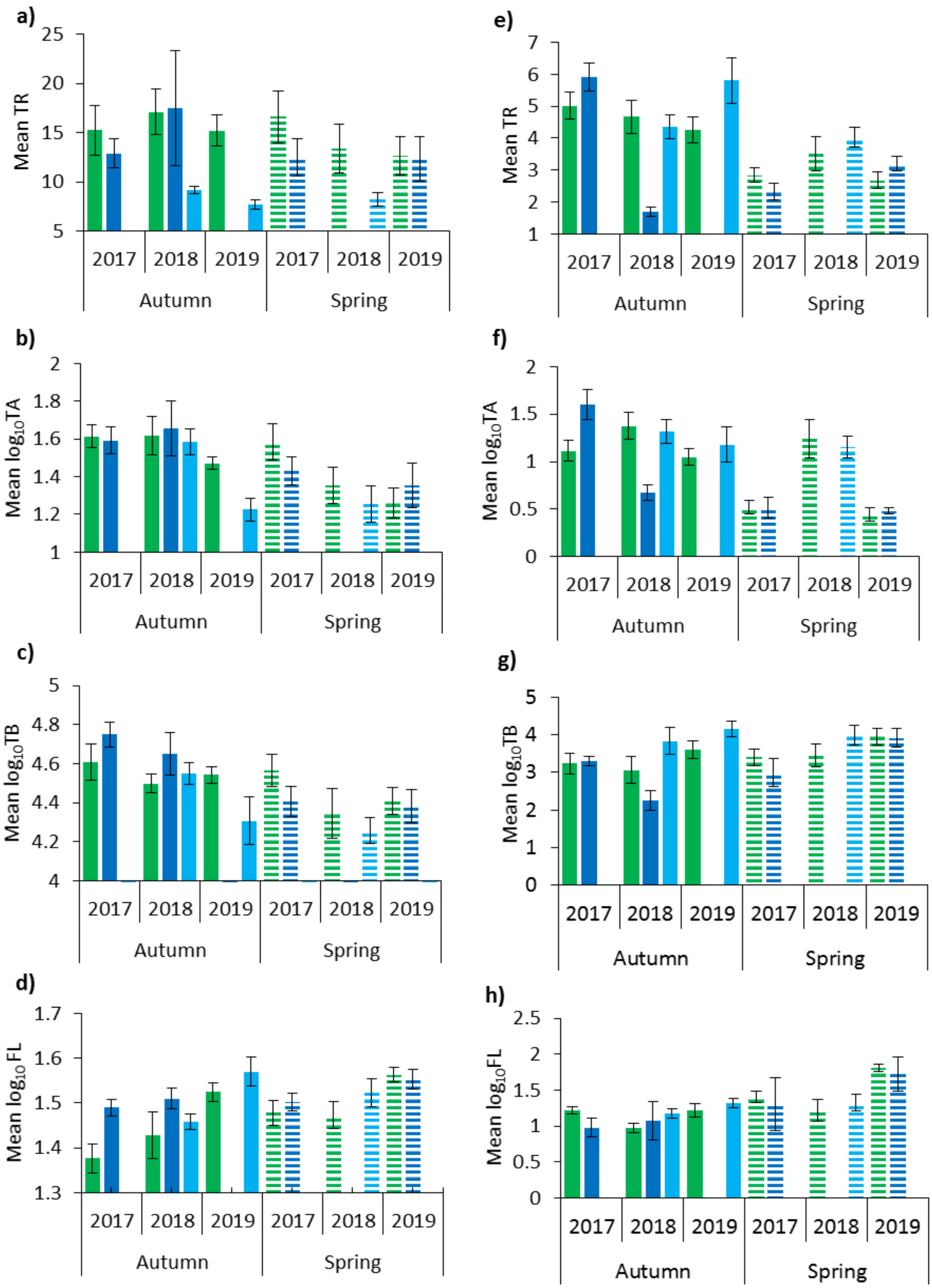

Figure 3.4 Mean values with standard errors (SE) for taxonomic richness (TR), and logged values of total abundance (TA), total biomass (TB) fork length (FL) by survey for demersal (left) and pelagic (right) communities at the three sites: Wandoo (green); Control Reef (dark blue) and Control Sand (light blue).

Table 3.1 Pairwise PERMANOVA tests comparing demersal and pelagic variation between sites for each survey, for taxonomic richness (TR), log total abundance ($log_{10}TA$), log total biomass ($log_{10}TB$) and log fork length ($log_{10}FL$). Degrees of freedom (d.f.) are reported. P-values in bold and with an asterisk are < 0.05, and the number of permutations (perms) are reported in parentheses.

			TR		$log_{10}TA$		$log_{10}TB$		$log_{10}FL$	
Survey x Site	Groups	d.f.	t	p (perms)	t	p (perms)	t	p (perms)	t	p (perms)
Demersal										
Autumn 2017	Control Reef, Wandoo	58	1.94	0.059 (146)	0.88	0.393 (997)	1.91	0.063 (999)	3.29	__*0.004__ (996)
Autumn 2018	Control Reef, Control Sand	59	4.24	__*0.001__ (138)	0.97	0.341 (996)	1.37	0.175 (997)	2.17	__*0.031__ (997)
Autumn 2018	Control Reef, Wandoo	48	0.13	0.885 (168)	0.07	0.936 (994)	1.86	0.063 (995)	3.39	__*0.002__ (997)
Autumn 2018	Control Sand, Wandoo	71	6.99	__*0.001__ (142)	1.31	0.176 (998)	0.34	0.721 (997)	2.27	__*0.024__ (996)
Autumn 2019	Control Sand, Wandoo	62	5.82	__*0.001__ (131)	5.04	__*0.001__ (999)	2.78	__*0.004__ (998)	1.42	0.173 (999)
Spring 2017	Control Reef, Wandoo	71	2.14	__*0.029__ (162)	1.80	0.082 (997)	1.61	0.114 (996)	0.81	0.432 (998)
Spring 2018	Control Sand, Wandoo	66	4.09	__*0.001__ (132)	1.07	0.298 (998)	1.21	0.226 (998)	1.09	0.287 (996)
Spring 2019	Control Reef, Wandoo	76	0.33	0.742 (169)	1.76	0.079 (996)	0.07	0.939 (997)	0.65	0.544 (998)
Pelagic										
Autumn 2017	Control Reef, Wandoo	16	1.43	0.194 (39)	2.59	__*0.026__ (981)	0.20	0.839 (976)	3.14	__*0.014__ (978)
Autumn 2018	Control Reef, Control Sand	12	4.93	__*0.003__ (123)	3.65	__*0.005__ (805)	3.05	__*0.016__ (779)	1.04	0.314 (762)
Autumn 2018	Control Reef, Wandoo	12	4.10	__*0.003__ (375)	3.47	__*0.009__ (768)	1.55	0.159 (775)	0.94	0.39 (775)
Autumn 2018	Control Sand, Wandoo	16	0.48	0.65 (121)	0.31	0.764 (981)	1.52	0.154 (975)	2.21	__*0.032__ (977)
Autumn 2019	Control Sand, Wandoo	9	2.06	0.089 (68)	0.74	0.511 (312)	1.54	0.152 (318)	0.75	0.462 (315)
Spring 2017	Control Reef, Wandoo	16	1.44	0.182 (213)	0.03	0.975 (972)	0.92	0.37 (974)	0.54	0.608 (976)
Spring 2018	Control Sand, Wandoo	16	0.84	0.419 (128)	0.38	0.697 (974)	1.33	0.198 (974)	0.58	0.578 (976)
Spring 2019	Control Reef, Wandoo	16	1.49	0.179 (151)	0.40	0.704 (910)	0.14	0.88 (981)	0.83	0.395 (984)

There were 17 demersal taxa from 11 families observed only at WN, compared with five unique taxa from five families at CR and four taxa from four families at CS (Table 3.4). Many of the demersal species unique to Wandoo are reef-associated species, and Wandoo was the only site where unidentified larval-stage juvenile fishes were present. Two demersal species recorded only at WN were observed on over 10% of deployments, namely the pickhandle barracuda *Sphyraena jello*, and giant sea catfish *Netuma thalassina* (Supplementary Table 3.5).

Pelagic assemblages followed similar patterns in terms of abundance (Fig. 3.5b) and biomass (Supplementary Fig. 3.2b) to those observed in the demersal assemblages. Abundance and biomass at WN were driven by great barracuda *Spyhraena barracuda* and rainbow runner *Elegatis bipinnulata*. Great barracuda were usually solitary, but frequently observed at WN (60% of zones, Supplementary Table 3.6), while rainbow runner was observed less frequently (15% of zones) but in large schools. There was some overlap in abundance between WN and CS, characterised by herrings (Clupeidae spp.) which were observed on 25% of zones at WN and 41% at CS. Abundance and biomass at CS was driven by silky sharks *Carcharhinus falciformis* and live sharksuckers *Echeneis naucrates*, and biomass was also characterised by cobia *Rachycentron canadum*. Abundance at CR was not strongly characterised by any particular species, while biomass was driven by great hammerheads *Sphyrna mokarran*, which was always solitary and only observed on 16% of zones. WN was the only site where any unique pelagic taxa were recorded, with rainbow runner not observed at either of the control sites (Table 3.3).

Table 3.2 Pairwise PERMANOVA results comparing abundance and biomass of the pelagic and demersal taxonomic assemblages between sites: Wandoo (WN); Control Sand (CS); and Control Reef (CR). Degrees of freedom (d.f.) are reported. P-values in bold and with an asterisk are < 0.05, and the number of permutations (perms) are reported in parentheses.

	Abundance			Biomass		
	Groups	t (d.f.)	p(perms)	Groups	t (d.f.)	p(perm)
Demersal	CR, CS	4.46 (218)	***0.001** (998)	CR, CS	4.85 (218)	***0.001** (999)
	CR, WN	3.48 (329)	***0.001** (996)	CR, WN	3.53 (329)	***0.001** (998)
	CS, WN	6.76 (317)	***0.001** (999)	CS, WN	7.53 (317)	***0.001** (997)
Pelagic	CR, CS	1.86 (52)	***0.002** (999)	CR, CS	1.63 (52)	***0.013** (998)
	CR, WN	1.88 (82)	***0.001** (998)	CR, WN	2.20 (82)	***0.001** (999)
	CS, WN	2.30 (72)	***0.001** (999)	CS, WN	2.44 (72)	***0.001** (999)

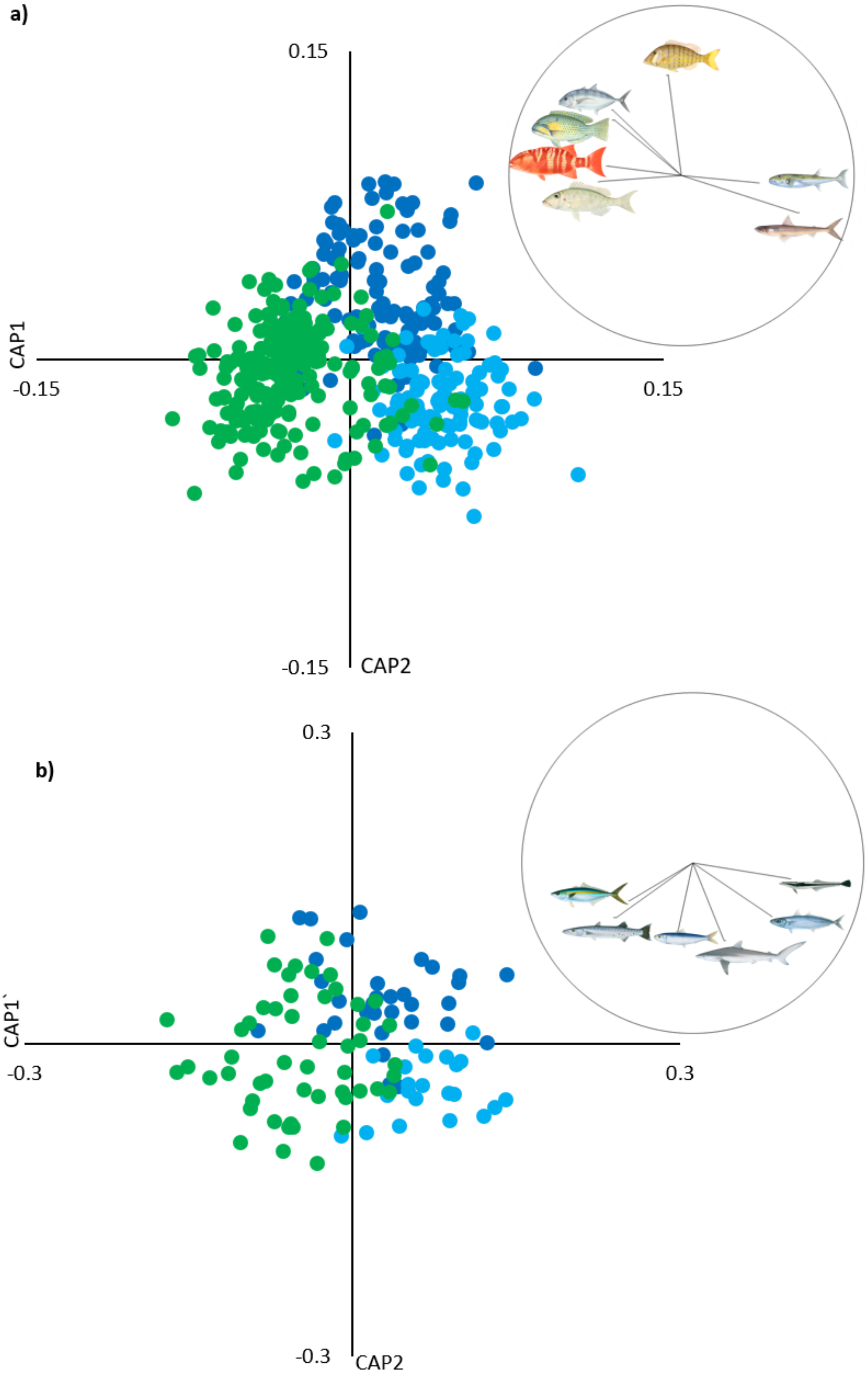

Figure 3.5 Canonical analysis of principal coordinates (CAP) for abundance of (a) demersal and (b) pelagic taxonomic assemblages at Wandoo (green); Control Reef (dark blue) and Control Sand (light blue). Species clockwise from top in (a) are: bluespotted emperor *Lethrinus punctulatus*, northwest blowfish *Lagocephalus sceleratus*, brushtooth lizardfish *Saurida undosquamis*, galloper *Symphorus nematophorus*, spot-cheek emperor *Lethrinus rubrioperculatus*, bluespotted tuskfish *Choerodon cauteroma*, and turrum *Carangoides fulvoguttatus*. Taxa clockwise from top in (b) are: live sharksucker *Echeneis naucrates*, scads *Decapterus* sp., silky shark *Carcharhinus falciformis*, herrings *Clupeidae* sp., great barracuda *Sphyraena barracuda*, and rainbow runner *Elegatis bipinnulata*. Images © R. Swainston/anima.fish

Table 3.3 Abundance, biomass and prevalence of taxa observed at a single site at WN (16 demersal and 1 pelagic species), CR (5 demersal and 0 pelagic species) and CS (4 demersal and 0 pelagic species), based on demersal and pelagic sampling records. Species marked with an asterisk are commonly caught commercially and/or recreationally in the North Coast Bioregion (Rome and Newman, 2010).

	Family	Binomial	Common names	Abundance	Biomass (g)	Prevalence (%)
Demersal						
Wandoo	Apogonidae	*Apogonidae* sp.	cardinalfishes	41	37.65	1.4
	Ariidae	*Netuma thalassina*	giant sea catfish	2	4696.20	12.1
	Blenniidae	*Meiacanthus* sp.	combtooth blennies	2	22.33	0.9
	Carangidae	*Carangoides dinema*	shadow trevally	2	1435.95	0.9
	Carangidae	*Carangoides orthogrammus*	island trevally	1	1308.26	2.3
	Carangidae	*Caranx sexfasciatus*	bigeye trevally	4	12604.88	1.4
	Carangidae	*Caranx tille**	tille trevally	1	4334.75	0.9
	Ginglymostomatidae	*Nebrius ferrugineus*	tawny nurse shark	1	5064.06	2.3
	Juvenile	*Juvenile* sp.	unidentified juvenile	1	0.05	0.9
	Lethrinidae	*Gymnocranius euanus*	paddletail seabream	1	554.72	1.4
	Pinguipedidae	*Parapercis* sp.	grubfishes	1	18.09	2.3
	Pomacentridae	*Pomacentrus nagasakiensis*	blue-scribbled damsel	3	6.68	0.9
	Serranidae	*Cephalopholis sonnerati**	tomato rockcod	2	978.65	0.9
	Serranidae	*Epinephelus chlorostigma**	brownspotted grouper	1	1024.97	0.9
	Serranidae	*Epinephelus malabaricus**	Malabar grouper	1	4097.69	1.9
	Sphyraenidae	*Sphyraena jello*	pickhandle barracuda	2	7637.25	10.7

Table 3.4 (cont.)

	Family	Binomial	Common name	Abundance	Biomass (g)	Prevalence (%)
Demersal						
Control Reef	Chaetodontidae	*Chaetodon auriga*	threadfin butterflyfish	2	445.78	1.7
	Labridae	*Bodianus bilunulatus*	saddleback pigfish	1	439.87	2.6
	Lethrinidae	*Lethrinus atkinsoni**	yellowtail emperor	2	1265.86	2.6
	Monacanthidae	*Eubalichthys caeruleoguttatus*	bluespotted leatherjacket	1	822.73	1.7
	Muraenidae	*Gymnothorax undulatus*	undulated moray	1	0.97	1.7
Control Sand	Carangidae	*Seriola rivoliana**	highfin amberjack	11	3280.25	1.9
	Clupeidae	*Clupeidae* sp.	herrings	334	22240.67	1.9
	Congridae	*Gorgasia* sp.	garden eels	31	2525.02	2.9
	Lutjanidae	*Pristipomoides multidens**	goldband snapper	1	3498.20	1.9
Pelagic						
Wandoo	Carangidae	*Elagatis bipinnulata*	rainbow runner	22	112861.56	15.0

3.5 Discussion

The demersal and pelagic community assemblages in the Wandoo field are distinct from those that would have existed prior to the installation of the infrastructure. The habitat around Wandoo is dominated by macrobenthos, in contrast with the sand-dominated habitat that would have likely prevailed historically (Sainsbury et al., 1993). As a result, the Wandoo demersal assemblage is characterised by reef-associated rather than sand-associated taxa. The pelagic assemblage at Wandoo is different from the other two sites, driven by species associated with offshore platforms on the NWS as well as in other regions around the world (Friedlander et al., 2014; McLean et al., 2019; Reynolds et al., 2018). Overall, the demersal and pelagic assemblages more closely resemble a natural reef than the assemblages that would have existed pre-installation. However, the composition of these assemblages is still unique to Wandoo, suggesting the emergence of a novel ecosystem.

While the focus of our study was on the fish assemblages, we saw clear differences in the habitat at the three sites. The proliferation of macrobenthos at the otherwise flat WN site, in contrast to the barren sand habitat at CS, likely reflects the exclusion of seabed trawling at WN. The WN site also had higher demersal fish richness than the control sites in most surveys which suggests that habitat composition is a driver of diversity in these demersal communities, as has been found elsewhere on the NWS (Keesing, 2019). The Pilbara Offshore meso-scale region, within which the study sites are located, is a biodiversity hotspot for sponges (Fromont et al., 2016). However, as much of the macrobenthos biomass was historically removed by seabed trawling (Sainsbury et al., 1993), most of the habitat in this region has been simplified. The impact of trawling is clear at CS and the area surrounding the reef at CR, with both sites dominated by bare sand. In contrast, WN excludes seabed trawling up to 500 m from the infrastructure, and exhibited similar macrobenthos communities to other oil and gas infrastructure on the NWS (Bond et al., 2018a; McLean et al., 2019).

The demersal community at WN was more diverse and reef-associated than the communities at the control sites. The higher demersal richness at WN is congruent with studies from Brazil, the Persian Gulf and Gabon, which describe offshore platforms as diversity hotspots (Fonseca et al., 2017; Friedlander et al., 2014; Torquato et al., 2017). High diversity is often associated with structural complexity of hard substrate (Friedlander and Parrish, 1998), and this association was observed in ROV surveys of the Wandoo infrastructure (Tothill, 2019). This study sampled areas around the infrastructure with little to no hard substrate, suggesting a large area of influence or “ecological halo” around the Wandoo infrastructure. The species that characterised the demersal taxonomic assemblage at WN, namely galloper and spot-cheek emperor, are both valued as fishing species: galloper is a prized sport fish, while spot-cheek emperor is a food fish targeted by recreational and commercial fishers (Froese and Pauly, 2019; Rome and Newman, 2010). These species occupy different habitats, with galloper inhabiting coral reefs and spot-cheek emperor inhabiting sand/rubble areas (Froese and Pauly, 2019). Spot-cheek emperor was rarely observed at either control site, despite the habitat at CR arguably being more suitable than that found at WN. Fishing activity, which is excluded at WN, may be the reason for the lower prevalence of this species at the control sites.

The similarity in pelagic communities across sites in terms of all four metrics was expected, given the three sites are located relatively close to each other and the highly mobile nature of pelagic species. For example, great barracuda have been shown to travel 12 km in a day and can migrate over 100 km, while silky sharks can travel up to 60 km a day (Bonfil, 2008; O’Toole et al., 2011). While these species are highly mobile, there was still strong distinction in the taxonomic assemblages between the three sites. The two species which characterised the taxonomic assemblage at WN, great barracuda and rainbow runner, are often associated with offshore platforms. Great barracuda is a commonly recorded species around offshore platforms in the Gulf of Mexico (e.g. Reynolds et al. 2018; Wetz et al. 2020), accounted for 33.2% of the biomass at offshore platforms in Gabon (Friedlander et al., 2014), and was recorded in 100% of remotely operated vehicle (ROV) transects at another platform on the NWS (McLean et al., 2019). Rainbow runner have also been recorded around platforms in the Gulf of Mexico, Gabon, and Brunei (Chou et al., 1992; Friedlander et al., 2014; Reynolds et al., 2018). Great hammerheads characterised biomass at CR, which was attributed to the fact that these are large animals and would have a significant effect on biomass even if present in low numbers, especially as there was not a particularly high abundance of any other species. The pelagic taxonomic assemblage at CS was characterised by silky sharks, which were observed around the within minutes of the vessel's arrival to conduct surveys at this site. This behaviour, and the associated high abundance and biomass of this species, were attributed to the frequent commercial fishing activity that occurs at this site. There are commercial line, trap and trawl fisheries operating throughout this area, including CS and, to a lesser extent, CR (WAFIC, 2020). This population of silky sharks is thought to be opportunistically targeting the discards from the commercial fishing vessels as a food source, which would explain their high abundance at a site otherwise scarce in the typical prey of this species, which includes scombrids, carangids, snappers and groupers (Compagno, 1984).

A distinct marine community exists at WN with various taxa not observed at natural habitats. Many of the 17 unique demersal species at WN are reef-associated, but species such as paddletail seabream *Gymnocranius euanus* and blue-scribbled damsel *Pomacentrus nagasakiensis* are found in sandy areas adjacent to reefs (Froese and Pauly, 2019). This suggests that the combination of sand and macrobenthos habitats around WN, itself a *de facto* artificial reef, is a key component of the high diversity and unique assemblage at this site. Reef-associated species tend to have strong site fidelity and post-settlement ranges of less than 50 m (Frederick, 1997). While it is possible that some species recruit to WN from natural sites, and certainly would have done when the platform was first installed, the high number of species unique to WN suggests that fish are being produced at the platform, rather than simply being attracted from natural habitats. Tothill (2019) observed juvenile fishes in the midwater (10-22 m) sections of Wandoo, providing further evidence of fish production. There was only one pelagic species unique to a single site, which may reflect the relatively mobile nature of pelagic animals. Rainbow runner were only observed at the WN site, which could be attributed to the association of this species with offshore platforms around the world (Chou et al., 1992; Reynolds et al., 2018). Offshore platforms can function as fish aggregation devices (FADs), aggregating fish by facilitating foraging and school formation (Dagorn et al., 2000; Haugen and Papastamatiou, 2019). Rainbow runner are thought to primarily aggregate around FADs to prey on small FAD-associated pelagic fishes (Xuefang et al., 2013), and it is possible that the vertical hard structure at WN is providing enhanced foraging opportunity for this species.

The exclusion of fishing around WN has created a *de facto* MPA, as has been reported at other offshore platforms (Friedlander et al., 2014; Fujii and Jamieson, 2016; Love et al., 2006). Seabed trawling on the NWS in the 1970s not only removed much of the macrobenthos habitat, but also resulted in a significant shift in fish composition (Sainsbury et al., 1993). The trawl catch shifted from being dominated by emperors (*Lethrinus* sp.) and snappers (*Lutjanus* sp.), to being dominated by lizardfish (*Saurida* sp.) and threadfin bream (*Nemipterus* sp.), with the abundance of lizardfishes greater by an order of magnitude (Sainsbury et al., 1993; Thresher et al., 1986). This relationship between habitat and species composition was also observed in this study: macrobenthos habitat was present at WN and CR, both of which were characterised by emperors. In contrast, at CS the habitat was almost completely devoid of macrobenthos, and the species composition was characterised by brushtooth lizardfish. Lizardfishes feed on benthic fishes, particularly on juveniles of other species, and are estimated to collectively consume 4×10^7 fishes per day on the NWS (Thresher et al., 1986). Demersal communities dominated by lizardfish, such as CS, would therefore have been significantly impacted by the proliferation of this genus. The *de facto* MPA has also resulted in a large ecological halo around the WN infrastructure. The ecological halo around offshore platforms and artificial reefs is usually around 15-34 m, with abundance and diversity similar to natural habitats beyond this distance (Reeds et al., 2018; Scarcella et al., 2011; Stanley and Wilson, 1996). In contrast, diversity at WN was higher than natural habitats at more than 50 m from the infrastructure. It is likely that WN has a larger ecological halo than those previously reported around offshore platforms, driven by recovery of macrobenthos habitat due to the exclusion of fishing within 500 m of the infrastructure.

Wandoo as a novel ecosystem

The ecosystem in the Wandoo field clearly has novel attributes when compared with natural systems in the region; however this assertion is not, on its own, sufficient to warrant labelling Wandoo a novel ecosystem. Van Elden *et al*. (2019) used the novel ecosystems definition developed by Hobbs *et al* (2013) to establish three criteria for evaluating offshore platforms as novel ecosystems:

1. **The abiotic, biotic and social components of the system differ from those that prevailed historically.** The addition of hard substrate through the installation of

the Wandoo infrastructure altered the abiotic component of the system. It is impossible to quantify the historical baseline of the biotic component, however the findings of this study show that the biotic components of the Wandoo ecosystem, in terms of habitat and marine communities, are distinct from those found at a proxy of their pre-installation historical state, i.e. the Control Sand site. The major social driver of this ecosystem is the exclusion of fishing activity, which has been detrimental to large areas of the NWS. The *de facto* MPA effect of Wandoo has been particularly important in providing a refuge for fishes and allowing macrobenthos communities to recover.

2. **The ecosystems have a tendency to self-organize and manifest novel qualities without intensive human management.** The Wandoo ecosystem, like those found at most other offshore platforms, is an unintended consequence of the installation of the platform and therefore is not subject to any human management. The only management undertaken is cleaning of sections of the subsea structure, but this activity only removes a small portion of the marine growth. The factors that allow this ecosystem to thrive, such as the exclusion of fishing and the provision of hard substrate, are artefacts of the presence of the platform.
3. **Novel ecosystems are prevented from returning to their historical states by practical limitations, in the form of ecological, environmental and social considerations.** Wandoo is due to remain operational for at least a further ten years, which is a significant social consideration as the presence of the infrastructure is central to this ecosystem. When Wandoo is decommissioned, it is possible that complete removal will allow the ecosystem to return to its pre-installation historical state, but the evidence presented here on the unique ecology of Wandoo should provide an ecological consideration against complete removal, thereby preventing a return to the historical state of the site.

Based on these criteria, Wandoo should be classified as a novel ecosystem. The environment and ecology of the site have been altered, a self-organising ecosystem with novel qualities has emerged, and the presence of the platform prevents the ecosystem from returning to its historical state.

Implications for decommissioning

We have used proxies for different decommissioning scenarios, which can provide a broad idea of how the ecosystem might look. We suggest that the Control Sand site is a proxy for complete removal, as this site is already a proxy for the Wandoo site without infrastructure. If the Wandoo infrastructure was completely removed, there would be a significant loss in diversity particularly in terms of reef-associated species. Pelagic species associated with midwater structure, such as great barracuda and rainbow runner, are also likely to be lost. Commercial and recreational fishing activity would likely recommence in the field post-decommissioning, as the petroleum safety zone would no longer be in effect and there would be no significant hard structure to prevent seabed trawling.

Topping, a second decommissioning scenario, would result in partial removal of Wandoo down to around 25 m below the surface. This method has been applied to shallow-water platforms in the U.S. (Ajemian et al., 2015). The reef at the Control Reef site rises to around 30 m below the surface, making this a close approximation to a topped Wandoo. This scenario would also result in the loss of pelagic species associated with structure, but would result in the retention of more of the demersal community than complete removal. There would be some losses: the shallower portions of Wandoo are important for juveniles, exhibit higher richness and abundance than deeper portions and are characterised by small reef fish such as damselfishes (Tothill, 2019). Indeed larval-stage juveniles were absent from the Control Reef site, and abundance of small demersal species such as damselfishes was generally lower than at the Wandoo site. It is likely that even under a topping scenario there would no longer be any exclusion of fishing activity around the remaining part of the platform. Seabed trawling could still occur in the areas surrounding the infrastructure that were previously protected by the petroleum safety zone.

Partial or complete removal of the Wandoo platform will likely have adverse impacts on a number of taxa and alter the ecosystem services provided by the field as a novel ecosystem. Partial removal would be less detrimental in that it would also still afford protection to the macrobenthos from seabed trawling. However there is significant ecological benefit in retaining the midwater sections of the infrastructure, for both pelagic species and juvenile reef-associated species, and leaving the platform standing in place would maintain these benefits. Additional aspects that should also be considered include the value of the field to seabirds, marine megafauna and macrobenthos communities attached to the infrastructure, as well as the potential spillover of fishes from the *de facto* MPA. The exclusion of fishing is a critical component of the large ecological halo present at Wandoo, and post-decommissioning protection, in the form of a no-take MPA, should be considered.

The installation of infrastructure in the Wandoo field has resulted in the emergence of a novel ecosystem with distinct ecological characteristics not found at natural sites in the region. The demersal and pelagic communities more closely resemble reef communities than those present pre-installation, but are still unique from those found at natural habitats in the region. The novel ecosystem at Wandoo also acts as a refuge for these communities, functioning as a *de facto* MPA in a region impacted by historical and current fishing activity. This MPA not only protects fish communities, but has allowed the macrobenthos to recover from the impacts of seabed trawling. Many of the novel characteristics of the Wandoo ecosystem would be lost under decommissioning scenarios that involve partial or complete removal, and the impact of decommissioning on fauna such as seabirds is still unknown. Recognising the Wandoo field as a novel ecosystem provides a mechanism for recognising the ecological role played by the Wandoo infrastructure, and underlines the need to consider the ecological role of each offshore platform prior to decommissioning.

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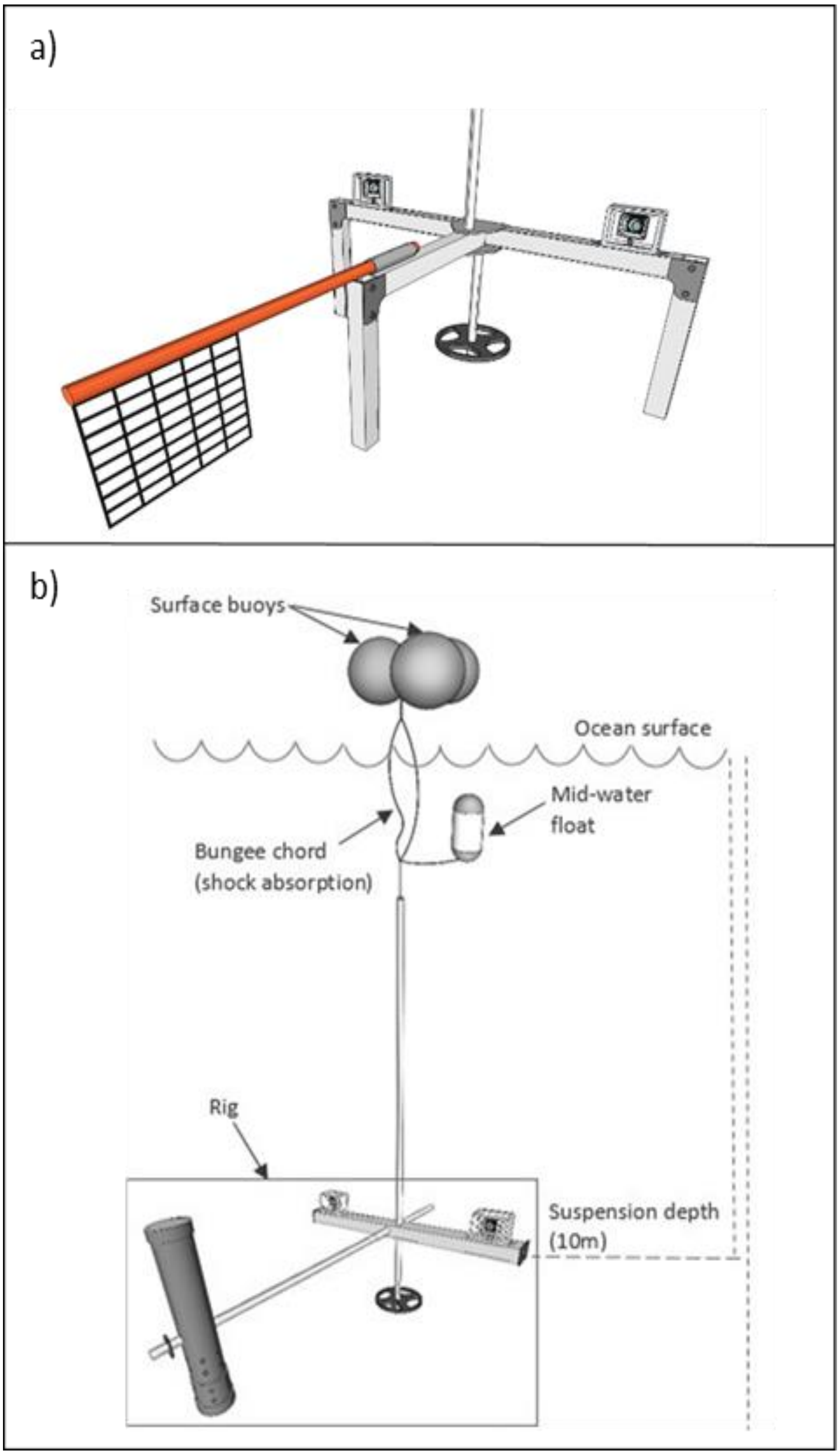
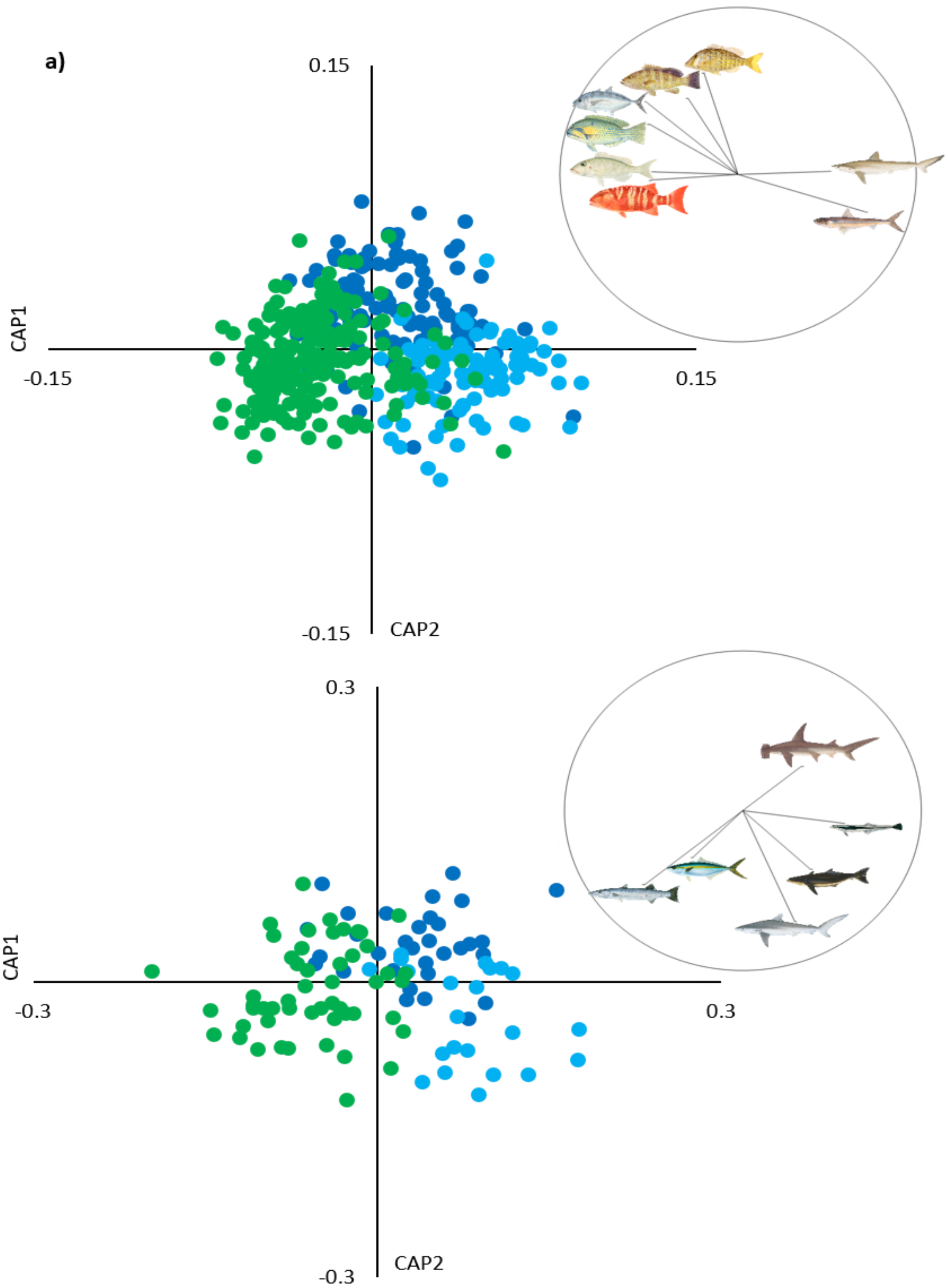
3.7 Supplementary Materials



Supplementary Figure 3.1 Schematics of (a) seabed and (b) mid-water stereo-BRUVS



Supplementary Figure 3.2 Canonical analysis of principal coordinates (CAP) for biomass of (a) demersal and (b) pelagic taxonomic assemblages at Wandoo (green); Control Reef (dark blue) and Control Sand (light blue). Species clockwise from top in (a) are: bluespotted emperor *Lethrinus punctulatus*, milk shark *Rhizoprionodon acutus*, brushtooth lizardfish *Saurida undosquamis*, galloper *Symphorus nematophorus*, spot-cheek emperor *Lethrinus rubrioperculatus*, bluespotted tuskfish *Choerodon cauteroma*, turrum *Carangoides fulvoguttatus*, and areolate grouper *Epinephelus areolatus*. Taxa clockwise from top in (b) are: great hammerhead *Sphyrna mokarran*, live sharksucker *Echeneis naucrates*, cobia *Rachycentron canadum*, silky shark *Carcharhinus falciformis*, rainbow runner *Elegatis bipinnulata*, and great barracuda *Sphyraena barracuda*. Images © R. Swainston/anima.fish

Supplementary Table 3.1 Metrics for each survey, including start and end dates, the number (n) of seabed deployments or midwater strings, sites sampled, number of taxa, taxonomic richness (TR), abundance (TA), biomass (TB) and fork length (FL). The sites sampled in each survey are included: Wandoo (WN); Control Reef (CR); and Control Sand (CS).

BRUVS	Survey	Start date	End date	n	Sites sampled	No. of taxa	TR* ± SE	TA* ± SE	TB* ± SE (kg)	FL ± SE (cm)
Seabed	Autumn 2017	4/05/2017	9/05/2017	60	WN; CR	206	14.3 ± 0.83	47.1 ± 3.81	60.5 ± 4.58	27.6 ± 1.07
	Autumn 2018	19/04/2018	26/04/2018	92	WN; CR; CS	252	13.6 ± 1.24	61.9 ± 5.78	46.9 ± 2.68	29.7 ± 1.09
	Autumn 2019	25/04/2019	30/04/2019	64	WN; CS	157	12.5 ± 0.63	27.9 ± 1.96	40.2 ± 4.14	35.8 ± 1.24
	Spring 2017	28/09/2017	4/10/2017	73	WN; CR	221	14.5 ± 1.34	55.7 ± 11.1	45.7 ± 4.12	32.1 ± 0.88
	Spring 2018	3/09/2018	19/09/2018	68	WN; CS	176	10.5 ± 0.65	31.3 ± 4.4	30.7 ± 3.07	33 ± 1.37
	Spring 2019	7/09/2019	11/09/2019	78	WN; CR	212	12.5 ± 0.84	29.9 ± 2.86	37.2 ± 2.36	37.1 ± 0.87
Midwater	Autumn 2017	1/05/2017	9/05/2017	18	WN; CR	54	5.5 ± 0.16	46.4 ± 6.6	15.2 ± 1.6	15.8 ± 1.1
	Autumn 2018	19/04/2018	26/04/2018	23	WN; CR; CS	83	3.9 ± 0.18	41.7 ± 5.4	38.7 ± 5.3	17.1 ± 1.1
	Autumn 2019	26/04/2019	30/04/2019	11	WN; CS	67	4.8 ± 0.22	35 ± 1.05	49.6 ± 5.7	27.5 ± 1.9
	Spring 2017	28/09/2017	4/10/2017	18	WN; CR	50	2.6 ± 0.09	7.1 ± 1.3	56.7 ± 11.8	48.6 ± 5.3
	Spring 2018	3/09/2018	19/09/2018	18	WN; CS	76	3.8 ± 0.16	41.5 ± 7.8	49.5 ± 5.8	32.7 ± 3
	Spring 2019	7/09/2019	11/09/2019	18	WN; CR	61	2.9 ± 0.1	4.5 ± 0.48	84.4 ± 6.9	84.9 ± 3.5

Supplementary Table 3.2 Environmental data for each survey used in the DistLM analyses, including start and end dates, depth, sea surface temperature (SST), and chlorophyll concentration (Chl-a). Data were derived from: Geoscience Australia 250 m bathymetry (Whiteway, 2009) and Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS) (IMOS, 2020). The sites are: Wandoo (WN); Control Reef (CR); and Control Sand (CS).

				Depth (m)			SST (°C)			Chl-a (mg/m3)		
Survey	Start Date	End Date	Site	Min	Max	Mean ± SD	Min	Max	Mean ± SD	Min	Max	Mean ± SD
Autumn 2017	4/05/2017	9/05/2017	CR	48.1	57.6	53.2 ± 2.02	27.8	27.9	27.9 ± 0.03	0.42	0.47	0.46 ± 0.02
			WN	48.9	55.2	52.4 1.64	27.9	28.0	28 0.02	0.36	0.49	0.4 0.04
Autumn 2018	19/04/2018	26/04/2018	CR	47.3	56.7	53 2.5	30.5	30.6	30.6 0.07	0.15	0.17	0.17 0.01
			CS	53.1	56.7	54.8 0.78	30.5	30.7	30.6 0.1	0.16	0.19	0.19 0.01
			WN	50.3	55.7	52.5 1.39	30.4	30.6	30.5 0.07	0.17	0.19	0.19 0.01
Autumn 2019	25/04/2019	30/04/2019	CS	53.0	55.0	54.1 0.63	28.6	28.6	28.7 0.01	0.69	0.79	0.75 0.04
			WN	50.0	55.0	52.9 1.31	28.4	28.6	28.5 0.04	0.75	0.96	0.89 0.06
Spring 2017	28/09/2017	4/10/2017	CR	43.1	56.6	52.7 2.99	24.9	25.0	25 0.04	0.61	0.66	0.64 0.02
			WN	49.3	57.5	52.4 1.39	24.8	24.9	24.9 0.02	0.38	0.42	0.41 0.01
Spring 2018	3/09/2018	19/09/2018	CS	53.3	57.2	55.3 1.19	23.7	23.8	23.8 0.02	0.30	0.31	0.31 0.01
			WN	50.2	56.7	52.7 1.5	23.5	23.5	23.6 0.02	0.21	0.23	0.23 0.01
Spring 2019	7/09/2019	11/09/2019	CR	43.0	56.0	52.6 2.99	23.2	23.3	23.3 0.03	0.23	0.25	0.24 0.01
			WN	50.0	54.0	52.3 1.17	23.2	23.3	23.3 0.03	0.23	0.34	0.26 0.04

Supplementary Table 3.3 Pairwise PERMANOVA comparing demersal variation between years for autumn and spring at each site for taxonomic richness (TR), log total abundance ($log_{10}TA$), log total biomass ($log_{10}TB$) and log fork length ($log_{10}FL$). Degrees of freedom (d.f.) are reported. P-values in bold and with an asterisk are < 0.05, and the number of permutations (perms) are reported in parentheses.

				TR		$log_{10}TA$		$log_{10}TB$		$log_{10}FL$	
		Groups	d.f.	t	p(perms)	t	p(perms)	t	p(perms)	t	p(perms)
Wandoo	Autumn	2017, 2018	63	0.502	0.601 (161)	0.256	0.805 (999)	1.479	0.133 (995)	0.145	0.875 (992)
		2017, 2019	71	0.679	0.496 (150)	3.725	***0.001** (997)	1.01	0.307 (999)	4.908	***0.001** (997)
		2018, 2019	68	1.24	0.226 (147)	3.312	***0.004** (997)	0.859	0.394 (999)	4.685	***0.001** (998)
	Spring	2017, 2018	66	1.335	0.179 (159)	2.268	***0.020** (996)	1.706	0.089 (995)	0.053	0.958 (998)
		2017, 2019	78	2.014	0.058 (190)	3.53	***0.001** (997)	1.462	0.124 (998)	2.687	***0.009** (997)
		2018, 2019	72	0.633	0.538 (161)	1.206	0.211 (996)	0.367	0.719 (997)	3.07	***0.004** (995)
	Seasonal	2017	69	0.216	0.837 (168)	0.792	0.436 (996)	0.568	0.578 (995)	2.204	***0.021** (997)
		2018	60	2.13	***0.041** (82)	3.72	***0.002** (995)	1.255	0.218 (995)	2.356	***0.019** (997)
		2019	80	1.86	0.053 (163)	3.009	***0.002** (997)	1.718	0.112 (998)	1.265	0.222 (998)
Control Reef	Autumn	2017, 2018	43	1.498	0.149 (160)	0.256	0.805 (999)	1.262	0.224 (996)	0.547	0.631 (999)
	Spring	2017, 2019	69	0.449	0.672 (151)	0.123	0.891 (998)	0.154	0.876 (996)	2.256	***0.024** (997)
	Seasonal	2017	60	0.343	0.767 (131)	2.44	***0.012** (997)	4.344	***0.001** (998)	0.236	0.825 (996)
Control Sand	Autumn	2018, 2019	65	2.58	***0.013** (50)	3.118	***0.003** (996)	2.054	***0.035** (997)	4.392	***0.001** (997)
	Seasonal	2018	77	0.947	0.34 (74)	3.72	***0.001** (995)	3.058	***0.003** (992)	2.143	***0.034** (997)

Supplementary Table 3.4 Pairwise permanova comparing pelagic variation between years for autumn and spring at each site, in terms of taxonomic richness (TR), log total abundance ($\log_{10}$TA), log total biomass ($\log_{10}$TB) and log fork length ($\log_{10}$FL. Degrees of freedom (d.f.) are reported. P-values in bold and with an asterisk are < 0.05, and the number of permutations (perms) are reported in parentheses.

				TR		$\log_{10}$TA		$\log_{10}$TB		$\log_{10}$FL	
		Groups	d.f.	t	p(perms)	t	p(perms)	t	p(perms)	t	p(perms)
Wandoo	Autumn	2017, 2018	16	0.518	0.615 (130)	1.476	0.161 (978)	0.38079	0.689 (982)	2.983	***0.007** (977)
		2017, 2019	14	1.257	0.234 (62)	0.452	0.661 (952)	0.96021	0.349 (958)	0.028	0.98 (951)
		2018, 2019	14	0.592	0.618 (212)	1.843	0.092 (943)	1.1664	0.281 (962)	2.14	0.063 (952)
	Spring	2017, 2018	16	1.167	0.265 (218)	3.397	***0.003** (973)	0.13979	0.892 (974)	1.147	0.256 (975)
		2017, 2019	16	0.452	0.677 (144)	0.741	0.462 (957)	1.8217	0.1 (972)	4.45	***0.001** (973)
		2018, 2019	16	1.398	0.179 (125)	3.718	***0.001** (966)	1.3543	0.192 (971)	3.831	***0.003** (967)
	Seasonal	2017	16	4.49	***0.002** (262)	4.621	***0.001** (975)	0.45032	0.651 (980)	2.064	0.059 (973)
		2018	16	1.534	0.149 (275)	0.548	0.583 (981)	0.81364	0.455 (980)	1.507	0.181 (987)
		2019	14	3.376	0.008 (146)	5.363	***0.001** (942)	1.0773	0.252 (963)	5.734	***0.001** (955)
Control Reef	Autumn	2017, 2018	12	6.83	***0.001** (170)	4.212	***0.002** (787)	4.0458	***0.001** (804)	1.016	0.319 (802)
	Spring	2017, 2019	16	2.483	***0.030** (230)	0.319	0.742 (961)	2.081	0.052 (980)	2.054	0.053 (975)
	Seasonal	2017	16	6.932	***0.001** (393)	5.712	***0.001** (977)	0.75851	0.462 (983)	1.729	0.09 (978)
Control Sand	Autumn	2018, 2019	11	1.959	0.066 (83)	0.627	0.528 (554)	0.58776	0.591 (533)	1.303	0.226 (543)
	Seasonal	2018	16	0.663	0.525 (64)	0.993	0.339 (980)	0.34781	0.722 (983)	1.123	0.279 (973)

Supplementary Table 3.5 Prevalence (%) of demersal species recorded at WN, CR and CS. Prevalence refers to the number of deployments on which a taxon was observed, out of the total number of deployments at that site.

Binomial	WN	CR	CS
Abalistes stellatus	92.1	95.7	84.6
Acanthocybium solandri	0.5	-	-
Acanthurus auranticavus	2.8	6.0	-
Acanthurus blochii	0.9	-	-
Acanthurus grammoptilus	1.9	0.9	-
Acanthurus sp.	3.3	6.0	-
Acanthurus xanthopterus	-	0.9	-
Aetobatus ocellatus	-	0.9	-
Aipysurus laevis	0.9	5.2	1.0
Aipysurus sp.	0.9	0.9	1.0
Alectis ciliaris	0.9	0.9	-
Alectis indica	0.5	-	-
Alepes vari	1.4	3.4	-
Aluterus monoceros	-	-	1.0
Aluterus scriptus	0.5	-	3.8
Amblyeleotris sp.	0.5	-	-
Amblygobius phalaena	0.5	-	-
Apogonidae sp.	1.4	-	-
Apolemichthys trimaculatus	-	0.9	-
Aprion virescens	0.5	-	1.0
Argyrops bleekeri	0.9	-	-
Argyrops spinifer	37.7	24.1	5.8
Arothron sp.	-	0.9	-
Arothron stellatus	0.5	-	-
Aspidontus dussumieri	1.9	1.7	-
Aspidontus taeniatus	-	0.9	-
Asteroidea sp.	0.9	5.2	1.0
Asteroidea sp.	-	0.9	-
Atule mate	1.4	0.9	1.0
Balistidae sp.	0.5	1.7	-
Blenniidae sp.	0.9	0.9	-
Bodianus bilunulatus	-	2.6	-
Bodianus perditio	10.2	16.4	-
Bodianus solatus	0.5	2.6	-
Bodianus sp.	0.9	-	-
Bothus pantherinus	0.5	-	-
Bothus sp.	2.8	-	4.8
Brachyura sp.	0.5	-	1.9
Caesionidae sp.	0.5	-	-
Canthigaster sp.	0.5	-	-
Carangidae sp.	8.4	12.1	27.9
Carangoides armatus	-	-	1.0
Carangoides chrysophrys	32.6	13.8	13.5
Carangoides coeruleopinnatus	69.8	43.1	17.3
Carangoides dinema	0.9	-	-
Carangoides fulvoguttatus	58.6	57.8	4.8
Carangoides gymnostethus	47.4	61.2	50.0
Carangoides oblongus	0.5	-	-
Carangoides orthogrammus	2.3	-	-
Carangoides sp.	7.4	3.4	15.4
Caranx ignobilis	2.8	2.6	2.9
Caranx melampygus	0.5	-	-
Caranx papuensis	1.4	2.6	-
Caranx sexfasciatus	1.4	-	-
Caranx sp.	0.9	-	-
Caranx tille	0.9	-	-
Carcharhinidae sp.	0.5	-	-
Carcharhinus amblyrhynchos	5.1	1.7	-
Carcharhinus amboinensis	1.4	-	1.0
Carcharhinus falciformis	-	-	1.0
Carcharhinus leucas	0.5	0.9	-
Carcharhinus limbatus	0.5	0.9	-
Carcharhinus melanopterus	0.5	-	-
Carcharhinus obscurus	-	-	1.9
Carcharhinus plumbeus	1.4	0.9	3.8
Carcharhinus sorrah	-	-	1.0
Carcharhinus sp.	2.8	6.0	10.6
Caretta caretta	-	0.9	-
Cephalopholis boenak	0.5	-	-
Cephalopholis sonnerati	0.9	-	-
Cephalopholis sp.	1.9	-	1.0
Chaetodon auriga	-	1.7	-
Chaetodon lineolatus	-	0.9	-
Chaetodontidae sp.	-	0.9	-
Chaetodontoplus duboulayi	23.7	15.5	1.0
Chaetodontoplus personifer	24.2	15.5	1.0
Chaetodontoplus sp.	-	1.7	-
Chelmon marginalis	0.5	0.9	-
Chiloscyllium punctatum	1.4	0.9	-
Chlorurus sp.	0.5	-	-
Choerodon cauteroma	47.0	44.8	1.9
Choerodon schoenleinii	0.5	1.7	-
Choerodon vitta	-	0.9	-
Chromis fumea	21.4	11.2	-
Chromis sp.	0.5	0.9	1.0

Binomial	WN	CR	CS
Chromis westaustralis	0.5	-	-
Chrysiptera tricincta	13.0	3.4	-
Cirrhibarbis sp.	0.9	0.9	-
Cirrhilabrus sp.	3.3	1.7	-
Cirrhitidae sp.	0.5	-	-
Clupeidae sp.	-	-	1.9
Congrogadus sp.	-	-	1.0
Coradion altivelis	0.5	0.9	-
Coradion chrysozonus	0.9	2.6	-
Coradion sp.	0.5	-	-
Coris caudimacula	16.7	2.6	2.9
Coris pictoides	2.8	-	-
Coris sp.	-	0.9	-
Crinoidea sp.	0.9	8.6	1.0
Cromileptes altivelis	2.3	0.9	-
Dasyatidae sp.	4.2	0.9	2.9
Decapterus sp.	2.3	1.7	8.7
Diagramma labiosum	18.1	7.8	1.0
Diploprion bifasciatum	2.3	5.2	-
Dischistodus perspicillatus	-	0.9	-
Echeneis naucrates	32.1	38.8	49.0
Echinoidea sp.	-	0.9	2.9
Elapidae sp.	6.0	2.6	1.9
Epinephelus areolatus	26.0	29.3	-
Epinephelus bilobatus	24.7	20.7	-
Epinephelus chlorostigma	0.9	-	-
Epinephelus coioides	1.9	1.7	-
Epinephelus lanceolatus	0.5	-	-
Epinephelus malabaricus	1.9	-	-
Epinephelus multinotatus	11.2	21.6	1.9
Epinephelus polyphekadion	0.5	0.9	-
Epinephelus sp.	7.0	8.6	1.0
Eubalichthys caeruleoguttatus	-	1.7	-
Eviota sp.	0.5	-	-
Feroxodon multistriatus	0.9	3.4	8.7
Fistularia commersonii	0.9	2.6	-
Fistularia sp.	1.9	0.9	-
Galeocerdo cuvier	0.9	0.9	1.0
Gastropoda sp.	0.9	-	-
Glaucostegus typus	0.5	-	-
Gnathanodon speciosus	27.0	16.4	2.9
Gobiidae sp.	3.7	11.2	1.9
Gorgasia sp.	-	-	2.9
Gymnocranius euanus	1.4	-	-
Gymnocranius grandoculis	13.5	5.2	-
Gymnocranius griseus	18.1	6.9	1.0

Binomial	WN	CR	CS
Gymnocranius microdon	3.3	-	-
Gymnocranius sp.	2.8	2.6	-
Gymnothorax flavimarginatus	0.5	-	-
Gymnothorax javanicus	0.5	-	-
Gymnothorax mccoskeri	-	0.9	-
Gymnothorax sp.	1.4	1.7	-
Gymnothorax thyrsoideus	0.5	-	-
Gymnothorax undulatus	-	1.7	-
Haemulidae sp.	1.4	-	-
Hemigaleidae sp.	-	-	1.0
Hemigymnus melapterus	-	0.9	-
Heniochus acuminatus	1.4	2.6	-
Heniochus diphreutes	0.9	0.9	-
Heniochus sp.	0.5	0.9	-
Heteroconger hassi	-	-	1.0
Heteroconger sp.	-	-	1.0
Hydrophis major	0.5	-	-
Hydrophis ocellatus	-	1.7	-
Hydrophis sp.	4.2	0.9	-
Hydrozoa sp.	0.5	0.9	-
Iniistius pavo	0.5	-	3.8
Juvenile sp.	0.9	-	-
Labridae sp.	10.2	1.7	1.0
Labroides dimidiatus	14.0	5.2	1.0
Lagocephalus lunaris	0.5	7.8	10.6
Lagocephalus sceleratus	12.6	32.8	51.0
Lagocephalus sp.	0.5	-	4.8
Leptojulis cyanopleura	3.3	0.9	-
Lethrinidae sp.	0.5	2.6	-
Lethrinus amboinensis	-	0.9	-
Lethrinus atkinsoni	-	2.6	-
Lethrinus erythropterus	-	0.9	-
Lethrinus laticaudis	0.5	-	-
Lethrinus miniatus	9.3	11.2	-
Lethrinus nebulosus	25.6	4.3	-
Lethrinus olivaceus	17.2	10.3	-
Lethrinus punctulatus	30.2	44.0	1.9
Lethrinus rubrioperculatus	40.5	12.1	-
Lethrinus sp.	2.8	3.4	-
Lutjanus argentimaculatus	1.4	1.7	-
Lutjanus carponotatus	0.5	-	-
Lutjanus erythropterus	10.2	6.0	-
Lutjanus johnii	-	0.9	-
Lutjanus lemniscatus	1.9	4.3	-
Lutjanus malabaricus	0.9	0.9	1.9
Lutjanus monostigma	0.5	-	-

Binomial	WN	CR	CS
Lutjanus russellii	-	0.9	-
Lutjanus sebae	41.4	31.0	3.8
Lutjanus sp.	1.9	6.0	1.0
Lutjanus vitta	2.3	1.7	1.0
Megalaspis cordyla	0.9	0.9	-
Meiacanthus sp.	0.9	-	-
Microdesmidae sp.	7.4	3.4	1.0
Monacanthidae sp.	0.5	1.7	1.9
Mullidae sp.	7.4	0.9	1.9
Mulloidichthys flavolineatus	0.5	-	2.9
Muraenidae sp.	0.5	-	-
Naso brevirostris	-	0.9	-
Naso fageni	0.5	-	-
Naso sp.	0.5	3.4	1.0
Natator depressa	0.5	-	-
Nebrius ferrugineus	2.3	-	-
Nemipteridae sp.	0.5	6.0	1.0
Nemipterus furcosus	38.1	52.6	67.3
Nemipterus sp.	17.2	18.1	33.7
Nemipterus sp1	7.0	0.9	21.2
Neotrygon sp.	0.5	-	-
Netuma thalassina	12.1	-	-
Octopus sp.	0.5	0.9	-
Ophichthidae sp.	-	0.9	-
Ophiuroidea sp.	0.5	-	2.9
Octopoda sp.	0.5	0.9	-
Teuthida sp.	-	0.9	-
Ostraciidae sp.	-	0.9	-
Oxycheilinus orientalis	0.5	-	-
Paguridae sp.	7.9	11.2	20.2
Palinuridae sp.	0.5	-	-
Parachaetodon ocellatus	1.9	0.9	-
Paracirrhites sp.	0.5	-	-
Parapercis sp.	2.3	-	-
Parapercis xanthozona	0.5	-	-
Paraplotosus butleri	0.5	-	-
Parupeneus barberinus	3.3	0.9	-
Parupeneus heptacanthus	8.8	5.2	-
Parupeneus indicus	16.3	15.5	-
Parupeneus pleurostigma	-	0.9	-
Parupeneus sp.	0.9	1.7	-
Parupeneus spilurus	-	0.9	-
Pentapodus emeryii	6.0	1.7	-
Pentapodus porosus	4.7	6.0	-
Pentapodus sp.	34.9	17.2	3.8
Pentapodus vitta	4.7	-	1.0
Pinguipedidae sp.	3.3	3.4	-

Binomial	WN	CR	CS
Plagiotremus sp.	0.5	-	-
Platax batavianus	7.4	-	1.0
Platax orbicularis	0.5	-	-
Plectorhinchus caeruleonothus	-	0.9	-
Plectorhinchus flavomaculatus	0.5	2.6	-
Plectorhinchus gibbosus	7.0	6.0	-
Plectorhinchus vittatus	0.5	-	-
Plectropomus areolatus	0.9	-	-
Plectropomus maculatus	18.1	16.4	1.0
Plectropomus sp.	19.1	11.2	-
Polycheata sp.	1.9	-	8.7
Pomacanthidae sp.	-	0.9	-
Pomacanthus imperator	3.7	9.5	-
Pomacanthus semicirculatus	-	5.2	-
Pomacanthus sexstriatus	7.9	4.3	-
Pomacanthus sp.	0.9	-	-
Pomacentridae sp.	9.8	3.4	4.8
Pomacentrus nagasakiensis	0.9	-	-
Pristipomoides multidens	-	-	1.9
Pseudobalistes fuscus	2.8	2.6	-
Pseudobalistes sp.	-	0.9	-
Pseudochromis sp.	1.9	2.6	-
Pseudomonacanthus peroni	-	0.9	-
Ptereleotris sp.	0.5	-	-
Pterocaesio sp.	-	0.9	-
Pterois volitans	-	0.9	-
Rachycentron canadum	2.8	3.4	3.8
Rhina ancylostoma	-	-	1.0
Rhinidae sp.	0.5	-	1.0
Rhinobatidae sp.	0.5	2.6	-
Rhizoprionodon acutus	7.9	31.0	52.9
Rhynchobatus sp.	3.7	11.2	2.9
Rhynchostracion nasus	-	0.9	3.8
Sarda orientalis	0.5	0.9	3.8
Sarda sp.	0.9	-	-
Saurida sp.	-	-	1.0
Saurida undosquamis	11.6	33.6	79.8
Scaridae sp.	13.0	9.5	1.0
Scarus frenatus	0.5	-	-
Scarus ghobban	1.9	4.3	-
Scarus sp.	7.0	1.7	-
Scolopsis monogramma	32.6	15.5	-
Scolopsis sp.	-	2.6	-
Scolopsis taenioptera	-	0.9	-
Scomberoides commersonnianus	1.9	-	1.0

Binomial	WN	CR	CS
Scomberoides lysan	1.4	-	-
Scomberoides sp.	3.7	0.9	-
Scomberomorus commerson	10.2	16.4	8.7
Scomberomorus sp.	9.3	12.9	3.8
Scombridae sp.	3.7	1.7	1.9
Scyphozoa sp.	0.5	-	-
Selar sp.	-	-	1.0
Sepia smithi	0.5	-	-
Sepia sp.	0.5	0.9	2.9
Seriola dumerili	0.5	-	-
Seriola rivoliana	-	-	1.9
Seriolina nigrofasciata	2.8	19.8	38.5
Serranidae sp.	2.3	-	1.0
Siganus punctatus	0.5	2.6	-
Siganus sp.	0.5	7.8	-
Sphyraena barracuda	11.6	0.9	1.9
Sphyraena jello	10.7	-	-
Sphyraena sp.	4.2	0.9	-
Sphyrna mokarran	1.9	0.9	2.9
Stegostoma tigrinum	3.3	0.9	1.9
Suezichthys cyanolaemus	11.2	3.4	4.8
Suezichthys devisi	0.5	-	-
Suezichthys soelae	0.9	0.9	-
Suezichthys sp.	0.5	-	-
Sufflamen fraenatum	32.6	39.7	1.0
Symphorus nematophorus	38.1	14.7	-
Synodontidae sp.	0.9	-	2.9
Synodus sp.	1.9	5.2	-
Synodus variegatus	0.5	-	-
Taeniurops meyeni	1.4	-	-
Tetraodontidae sp.	-	0.9	-
Teuthida sp.	-	-	1.0
Triglidae sp.	-	-	1.0
Upeneus australiae	0.5	-	-
Upeneus luzonius	0.9	-	1.0
Valenciennea sp.	0.9	0.9	-
Zabidius novemaculeatus	-	0.9	-

Supplementary Table 3.6 Prevalence (%) of pelagic species recorded at WN, CR and CS. Prevalence refers to the number of deployments on which a taxon was observed, out of the total number of deployments at that site.

Binomial	WN	CR	CS
Ablennes hians	-	-	5
Acanthocybium solandri	5.8	3	18
Alepes apercna	3.8	3	-
Alepes kleinii	1.9	-	-
Alepes sp.	19	31	-
Alepes vari	1.9	-	-
Aluterus monoceros	27	28	18
Aluterus scriptus	65	47	41
Aluterus sp.	7.7	-	32
Apogonidae sp.	1.9	3	-
Atule mate	40	41	55
Auxis thazard	-	-	5
Balaenoptera acutorostrata	1.9	3	-
Brachyura sp.	1.9	-	-
Cantherhines dumerilii	9.6	6	5
Cantherhines pardalis	1.9	-	-
Carangidae sp.	37	44	50
Carangoides armatus	12	6	36
Carangoides ferdau	-	-	5
Carangoides fulvoguttatus	3.8	-	-
Carangoides gymnostethus	3.8	-	5
Carangoides sp.	9.6	3	27
Caranx ignobilis	1.9	-	-
Caranx sexfasciatus	1.9	6	-
Caranx sp.	3.8	-	-
Carcharhinidae sp.	1.9	-	-
Carcharhinus amblyrhynchos	21	-	-
Carcharhinus amboinensis	9.6	28	50
Carcharhinus brevipinna	3.8	6	-
Carcharhinus falciformis	15	3	50
Carcharhinus galapagensis	1.9	-	-
Carcharhinus leucas	1.9	6	14
Carcharhinus limbatus	21	6	-
Carcharhinus obscurus	40	28	36
Carcharhinus plumbeus	54	47	50
Carcharhinus sorrah	13	6	23
Carcharhinus sp.	58	53	59
Carcharhinus tilstoni	1.9	-	-
Cestum veneris	9.6	-	18
Cheloniidae sp.	5.8	-	5
Clupeidae sp.	25	13	41
Coryphaena equiselis	-	-	5

Binomial	WN	CR	CS
Coryphaena hippurus	1.9	3	5
Decapterus macarellus	-	-	5
Decapterus sp.	69	66	86
Delphinidae sp.	1.9	-	-
Disteira major	1.9	-	-
Echeneidae sp.	-	3	-
Echeneis naucrates	48	63	86
Elagatis bipinnulata	15	-	-
Elapidae sp.	15	9	5
Ephippidae sp.	-	6	-
Eubalichthys caeruleoguttatus	3.8	6	9
Fistularia sp.	9.6	6	18
Galeocerdo cuvier	5.8	9	5
Gnathanodon speciosus	5.8	3	5
Hydrophis ocellatus	1.9	-	5
Hydrophis sp.	1.9	3	9
Hydrozoa sp.	37	13	18
Istiompax indica	3.8	3	18
Istiophoridae sp.	-	3	-
Istiophorus platypterus	7.7	3	-
Juvenile sp.	37	13	18
Katsuwonus pelamis	1.9	-	-
Labroides dimidiatus	-	-	5
Lagocephalus sceleratus	1.9	-	-
Makaira nigricans	3.8	3	23
Megalaspis cordyla	1.9	-	-
Metavelifer multiradiatus	-	-	5
Mobula kuhlii	-	-	5
Mobula sp.	3.8	6	5
Monacanthidae sp.	27	6	14
Natator depressa	1.9	-	-
Naucrates ductor	-	-	5
Nomeidae sp.	1.9	-	5
Platax sp.	-	3	9
Platax teira	1.9	-	-
Priacanthus sp.	1.9	-	9
Psenes sp.	3.8	6	32
Rachycentron canadum	1.9	6	32
Remora remora	-	3	5
Remora sp.	-	-	5
Sardinella sp.	1.9	-	-
Scomberoides lysan	3.8	-	-
Scomberomorus commerson	23	-	-
Scomberomorus sp.	9.6	-	5
Scombridae sp.	3.8	6	-
Scyphozoa sp.	42	25	27
Selar boops	1.9	-	-

Binomial	WN	CR	CS
Selar sp.	1.9	3	-
Sepia sp.	1.9	-	-
Seriola sp.	17	31	-
Seriolina nigrofasciata	31	38	68
Sphyraena barracuda	60	6	23
Sphyraena sp.	-	-	5
Sphyrna mokarran	1.9	16	18
Teuthida sp.	-	3	-

CHAPTER 4 WILD OBSERVATION OF PUTATIVE DYNAMIC DECAPOD MIMICRY BY A CUTTLEFISH (*SEPIA* CF. *SMITHI*)

van Elden S, Meeuwig JJ. 2020. Wild observation of putative dynamic decapod mimicry by a cuttlefish (Sepia cf. smithi). *Marine Biodiversity* 50:93.

KEYWORDS: STEREO-BRUVS . CEPHALOPOD BEHAVIOUR . DECAPOD MIMICRY . NOVEL FIELD OBSERVATION

4.1 ABSTRACT

Stereo baited remote underwater video systems (BRUVS) are widely used to document diversity, abundance, and biomass of marine wildlife and record unusual behaviours. We observed a cuttlefish appearing to mimic decapod morphology and locomotion during a non-targeted BRUVS study on Australia's Northwest Shelf. While the pharaoh cuttlefish Sepia pharaonis (Ehrenberg, 1831) is putatively thought to mimic the appearance of a hermit crab in a laboratory setting, our observation is the first wild record of decapod mimicry by a cuttlefish, tentatively identified as Sepia smithi (Hoyle, 1885). In situ observations increase our understanding of how cuttlefish behave in their natural environment while interacting with other species and provide opportunities to further our understanding of the source and breadth of these mimicry.

4.2 INTRODUCTION

Mimicry at the organism level is a phenomenon whereby a plant or animal (the *mimic*) uses various signal emissions such as sound, colour, shape, or scent to plagiarise something living or non-living (the *model*), in order to deceive a predator or prey animal (the *dupe*) (Pasteur 1982). Most cases of mimicry are static whereby the organism is in a permanent state of mimicry; for example the nonvenomous king snake *Lampropeltis elapsoides* (Holbrook, 1838*)* has similar ringed markings to the venomous coral snake *Micrurus fulvius* (Linnaeus, 1766), and these markings cannot be changed (Pfennig et al. 2001). Some organisms, however, have the ability to choose when to mimic a model. The bluestriped fangblenny *Plagiotremus rhinorhynchos* (Bleeker, 1852) for example, can change its appearance by "turning off" the colours which allow it to mimic the bluestreak cleaner wrasse *Labroides dimidiatus* (Valenciennes, 1839, Côté and Cheney 2005).

Cephalopods have a highly advanced ability to rapidly change their appearance by altering various body pattern components such as colour, texture, posture, and locomotion (Hanlon 2007; Hanlon and Messenger 2018). The ability to swiftly change their body pattern is used by cephalopods for predator defence, feeding, mating, and communication (Hanlon and Messenger 2018). A range of cephalopod species employ changes to body pattern in order to mimic other animals: Various species of octopus such as *Macrotritopus defilippi* (Vérany 1851) and *Thaumoctopus mimicus* (Norman 2005) mimic several animals including flatfish, parrotfish and banded sea-snakes (Norman et al. 2001; Hanlon et al. 2010; Huffard et al. 2010), while juvenile meso-pelagic squid *Chiroteuthis calyx* (Young 1972) have been documented to mimic siphonophores (Burford et al. 2015). Evidence of mimicry in cuttlefish is limited to a few species, including reports of small male giant cuttlefish *Sepia apama* (Gray, 1849) mimicking females to improve their chances of mating (Norman et al. 1999; Hall and Hanlon 2002), and juvenile stumpy-spined cuttlefish *Sepia bandensis* (Adam, 1939) mimicking snails (Warnke et al. 2012).

Okamoto et al. (2017) described for the first time a unique 'arm-flapping' behaviour observed in the pharaoh cuttlefish *Sepia pharaonis* in aquaria during a laboratory study and hypothesised it to be a case of hermit crab mimicry. This "crustacean-like aggressive mimicry" is described in detail by Nakajima and Ikeda (2017). Here, we report the first known observation in the wild of decapod mimicry by a cuttlefish *Sepia* sp. and discuss possible reasons for this behaviour by comparing the situation in which this behaviour was observed with the descriptions of this behaviour by Okamoto et al. (2017) and Nakajima and Ikeda (2017).

4.3 Materials and methods

Stereo baited remote underwater video systems (BRUVS) are a non-destructive sampling method that is well established for documenting diversity, abundance, size structure and biomass of marine communities (Cappo et al. 2006). Stereo-BRUVS have been developed for benthic and mid-water environments (Whitmarsh et al. 2017; Bouchet et al. 2018), and because they are relatively inexpensive, they can be deployed across large spatial scales (Letessier et al. 2013b). BRUVS have been shown to sample wide range of taxa beyond those attracted by the bait, including herbivores and planktivores, and documented taxa ranging from krill to turtles to whales (Watson et al. 2010; Letessier et al. 2013a, 2015; De Vos et al. 2014; Bouchet et al. 2018; Thompson et al. 2019). More recently, stereo-BRUVS have been used to document the behaviour of a variety of marine organisms observed as part of broader studies, including fish, sharks, and lobsters (Weiss et al. 2006; Fox and Bellwood 2008; Barley et al. 2016; Birt et al. 2019).

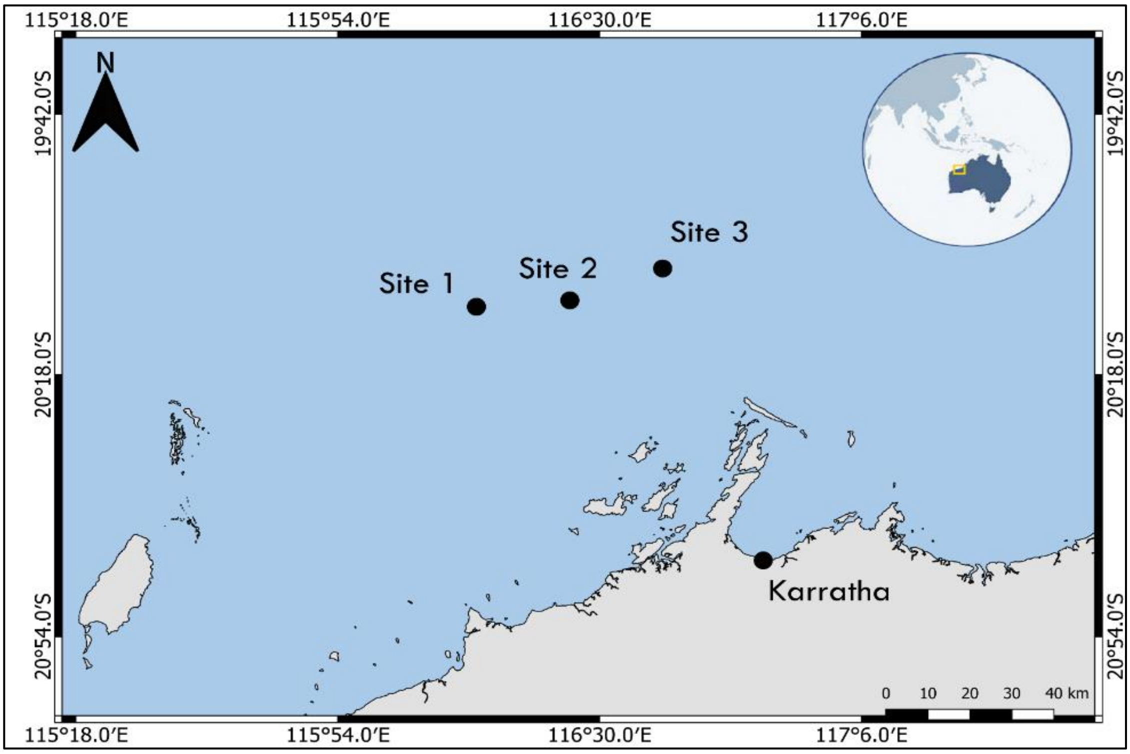


Figure 4.1 Map showing the three sites where stereo-BRUVS were deployed in this study. The mimicry observation occurred at the site 2.

Stereo-BRUVS were deployed at three sites in north-western Australia (Fig. 4.1) as part of a broad ecological study: a reef comprising rocky substrate and various sessile invertebrates such as sponges and gorgonian corals (Site 1); an offshore oil platform situated on flat, sandy habitat with patchy sessile invertebrate coverage, predominantly sponges and gorgonian corals (Site 2); and an open, sandy site with no physical relief (Site 3). Mean sea surface temperature (SST) was similar across all three sites, ranging from 23.55 °C in the Austral spring to 30.20 °C in the Austral autumn. Depth was also similar across all sites, with a depth range of 43 - 57.7 m. Stereo-BRUVS consist of two video cameras mounted on a base bar 80 cm apart, converging at an angle of four degrees to a common focal point. Cameras were set to medium field of view and 1080 p resolution. As per standard BRUVS practices of using oily, soft-bodied fish as bait (Langlois et al. 2018), stereo-BRUVS were baited with 800 g of pilchards (*Sardinops* spp.), in a bait bag made of galvanised steel mesh, suspended in front of the cameras on the end of a 1.5 m long PVC plastic tube. A total of 125 stereo-BRUVS were deployed in this study using standard practices (Langlois et al. 2018), with five stereo-BRUVS being deployed in a set, and each camera recorded for a minimum of 60 min. All footage was analysed using EventMeasure (Seagis 2017), which allows for identification of all taxa, abundance counts using the MaxN abundance metric, as well as length measurements.

4.4 Results

This behaviour was recorded at a depth of 50.8 m, at 20.124° S and 116.440° E at 07:20 am on 20 April 2018, with the location characterised by flat, mainly sandy habitat. It occurred seven minutes into the 60 minute video sample and the duration of the behaviour was 45 seconds. During this recording, two known predators of cuttlefish were also recorded (Froese and Pauly 2019): the milk shark *Rhizoprionodon acutus* (Rüppell 1837), 65.24 cm long and recorded 26.9 minutes into the video and brushtooth lizardfish *Saurida undosquamis* (Richardson 1848), 41.7 cm long and recorded 27.6 minutes into the video. At the time of the cuttlefish observation, there were no other animals in the video. A total of five cuttlefish were observed across 125 BRUVS deployments, however the reported behaviour was only observed once (Table 4.1). A total of 53 decapods were observed on during this study, including 51 hermit crabs, one blue swimmer crab *Portunus armatus* (A. Milne-Edwards, 1861) and one painted rock lobster *Panulirus versicolor* (Latreille, 1804).

Table 4.1 Record of all cuttlefish seen on BRUVS deployments with information on habitat, depth, sea surface temperature (SST), estimated visibility (vis) and activity for each of the three sites: A reef comprising rocky substrate and various sessile invertebrates

Date	Species	Site	Habitat	Depth (m)	SST (°C)	Activity
4/10/2017	*Sepia* sp	1	Sand	53.2	24.65	Swimming; sitting
20/04/2018	*Sepia* sp cf. *smithi*	2	Sand	50.8	30.13	Crustacean-like aggressive mimicry
23/04/2018	*Sepia* sp	3	Sand	54.3	30.20	Swimming; hovering
17/09/2018	*Sepia* sp	3	Sand	56.4	23.93	Swimming; arms fully extended
7/09/2019	*Sepia* sp	2	Sponges; soft corals	52	23.63	Swimming; arms curled

A cuttlefish, *Sepia* sp. with a mantle length of 12.2 cm was recorded at Site 2 approaching the bait bag. The first (dorsal-most) pair of arms were raised vertically and the distal ends darkened, resembling eyestalks, while the second and third pairs were bent, as if to appear jointed, and were used in a sideways 'walking' motion. The fourth (ventral-most) pair of arms was used to raise the head and arms of the cuttlefish off the substrate, so that the head was higher than the mantle and the tentacles were hidden (Fig. 4.2a). This body pattern is similar to the "crustacean-like aggressive mimicry" shown in Figure 5 of Nakajima and Ikeda (2017). There is also a flashing dark bar present at the base of the arms, similar to that displayed in Electronic Supplementary Material S1 and S2 in Okamoto et al. (2017). When close to the bait bag, the cuttlefish initially raised its head by extending the fourth pair of arms before swimming up to the bait bag approximately 30 cm above the seabed, and partially extending the second and third pairs of arms while approaching the bait (Fig. 4.2b). The cuttlefish then descended to the seabed and resumed the body pattern described above (Fig. 4.2c). The cuttlefish then moved away from the camera rig, maintaining decapod-like posture but ceasing the "walking" behaviour and instead swimming just above the substrate until it was no longer visible.

4.5 Discussion

We report here the first apparent observation of putative decapod mimicry by a cuttlefish in the wild. While the motivation for this behaviour is unclear from the video footage, we hypothesise that this mimicry may be a method of predator avoidance. The observation occurred in a habitat with little to no benthic cover to offer the cuttlefish protection from predators. A hard-bodied organism such as a crab would present as a less attractive target for the typical predators of a soft-bodied cuttlefish. This was also consistent with Okamoto et al. (2017), where they hypothesised that *S. pharaonis* was mimicking a hermit crab in their study possibly to avoid predators, but recommended further investigation of this behaviour both experimentally and in the wild to validate their hypothesis.

An alternative hypothesis is that the cuttlefish here was using an aggressive, rather than defensive, form of mimicry. Okamoto et al. (2017) observed this aggressive mimicking behaviour while cuttlefish were hunting prey in an aquarium. In our observation, it is possible that the cuttlefish was also using mimicry while approaching potential prey, in this case the bait. Due to this observation occurring in the wild, without control of all potential behavioural triggers, we can only hypothesise on the motivation behind the observed mimicry.

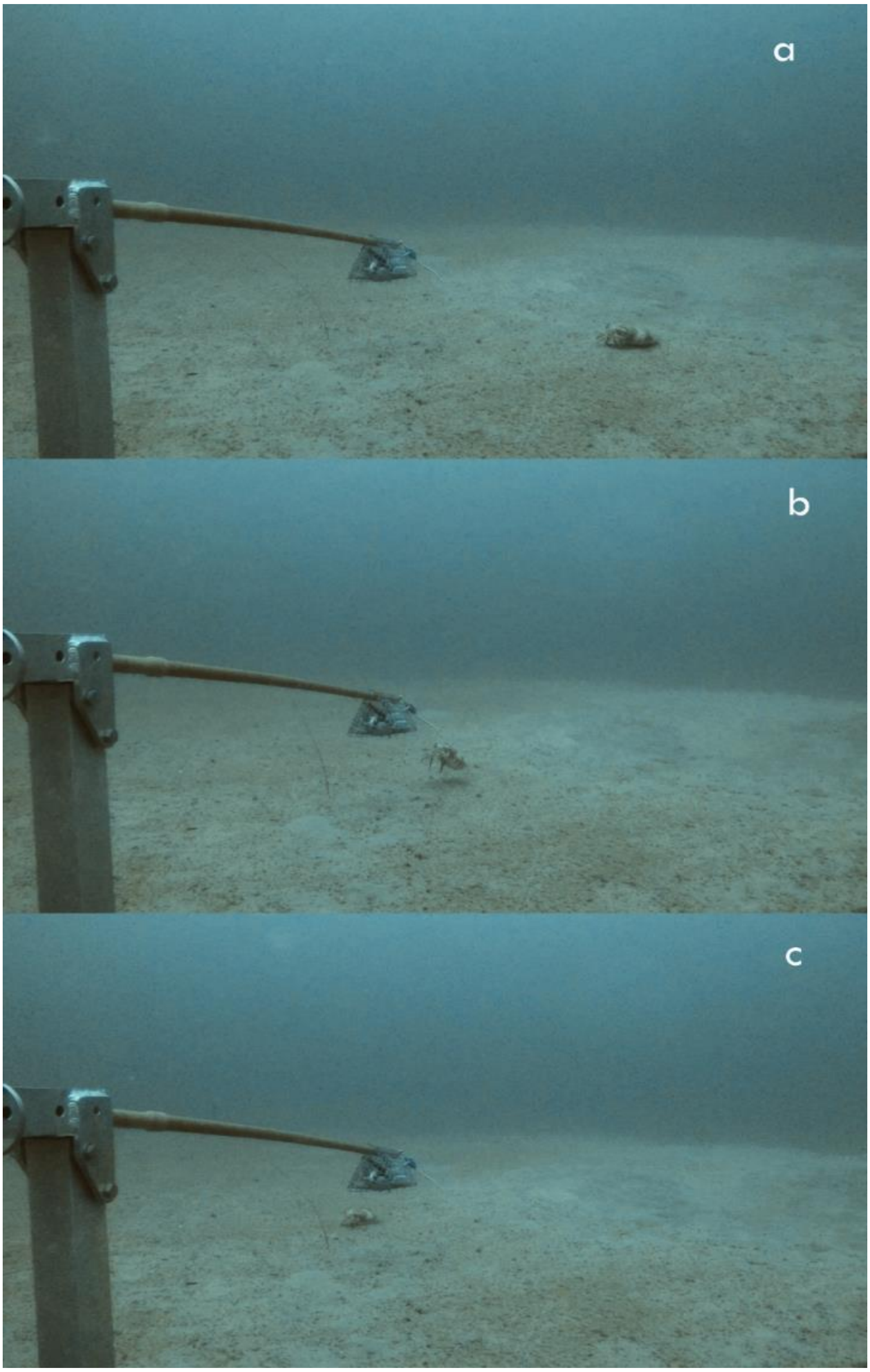


Figure 4.2 Selected frames from the video image (Online Resource 1): *Sepia* sp. approaches the bait bag while mimicking decapod (a), extends its arms while investigating the bait bag (b), before moving away from the camera mimicking decapod locomotion (c).

As non-cuttlefish experts, we elicited expert advice (Dr Mandy Reid, Malacology Collection Manager, Australian Museum) who suggested the individual was most likely *Sepia smithi,* with the caveat that it is difficult to identify cuttlefish based on images alone. This tentative identification was made based on the white band over the mantle and eyes of the cuttlefish, as well as the habitat type and time of day of the activity.

We also reviewed the literature to narrow down the potential species pool based on parameters such as distribution, depth of observation, size, and habitat as derived from SeaLifeBase (Palomares and Pauly 2019), Atlas of Living Australia (www.ala.org.au) and CephBase (www.cephbase.eol.org). Of the approximate 111 species of the genus *Sepia* globally, 35 are found in Australia. There are five species that overlap in distribution, depth and size with our observed individual, allowing some latitude on habitat association and maximum mantle length (Table 4.2), including *S. smithi*.

Sepia smithi is native to the region where our observations were made and this individual's size would make it an adult of this species. While the species of cuttlefish could not be confirmed from the stereo-BRUVS footage, the behaviour observed in the wild and presented in this paper appears to provide an *in situ* example of "crustacean-like aggressive mimicry" previously only described from captive cuttlefish behaviour (Nakajima and Ikeda 2017; Okamoto et al. 2017). It is unclear if there are other cases of congeneric dynamic mimicry in the animal kingdom, but we speculate that the high cognitive ability of coleoid cephalopods would make it possible for multiple species of cuttlefish to display similar forms of crustacean-like aggressive mimicry, where the particular model being mimicked could vary by species and/or environment. This could be a case of convergent evolution as these congeners overlap in habitat and are possibly targeted by similar predators. Crustacean-like aggressive mimicry is still a relatively novel behaviour in the literature, and as such warrants significant further research in order to determine the prevalence of this behaviour within the genus *Sepia*.

Table 4.2 Cuttlefish species found in the study region with information on maximum length (TL; cm), mantle length at maturity (ML; cm), depth range (m), habitat association and diel activity where available. Taxon authorities are provided for species not previously mentioned in this manuscript. Derived from SeaLifeBase (Palomares and Pauly 2019), Atlas of Living Australia (www.ala.org.au) and Cephbase (www.cephbase.eol.org).

Species	Common name	TL	ML	Depth	Habitat	Diel Activity
Sepia latimanus Quoy and Gaimard 1832	broadclub cuttlefish	50	16	8-55	Reef associated	Diurnal
Sepia pharaonis	pharaoh cuttlefish	43	12	0-130	Soft bottom (sand); seagrass	Nocturnal
Sepia elliptica Hoyle 1885	ovalbone cuttlefish	17.5	9.9	10-142	Benthic (unspecified)	n/a
Sepia smithi	Smith's cuttlefish	17	na	7-138	Soft bottom (sand; mud)	Diurnal
Sepia papuensis Hoyle 1885	Papuan cuttlefish	11	na	10-155	Soft bottom (sand; silt; mud)	Nocturnal

The use of stereo BRUVS to document diversity, abundance, size distributions and biomass in a wide range of marine environments is well documented (Whitmarsh et al. 2017). More recently, they have also been the basis of reports on unusual behaviours such as knotting in moray eels (Barley et al. 2016). As BRUVS allow us to increase the time spent observing marine animals *in situ*, without the influence of humans in the vicinity, they are likely to continue to unveil a wide range of rare and elusive behaviours, including mimicry, which will help further our understanding of how marine animals behave and interact in their natural environments. The five cuttlefish observations in 250 hours of video in this study show that stereo-BRUVS may not be the most efficient method for studying cuttlefish behaviour, however stereo-BRUVS have previously been adapted to target particular species. The same stereo-BRUVS used here have been used, unbaited, to successfully observe the behaviour of Antarctic krill *Euphausia superba* (Dana, 1850) in aquaria (Letessier et al. 2013a). A significant amount of the existing knowledge about cephalopod behaviour has come from experiments and laboratory studies (Hanlon and Messenger 2018). Further video observation of cephalopods in the wild increases our understanding of their complex behaviour, interactions with other species, and the environmental factors that drive these behaviours.

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4.7 Statements

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Conflicts of interest/Competing interests

The authors declare that they have no conflict of interest.

Ethical Approval

Experimental protocols were approved by the University of Western Australia's Animal Ethics Committee (RA/3/100/1484), and were carried out in accordance with the approved guidelines.

Sampling and field studies:

All necessary permits for sampling and observational field studies have been obtained by the authors from the competent authorities.

Data Availability

All data generated or analysed during this study are included in this published article and its supplementary information files.

Authors' contributions

SVE and JM conceived the manuscript. SVE wrote the first draft of the manuscript. Both authors contributed to the manuscript revision, read, and approved the submitted version.

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CHAPTER 5 ELEVATED ABUNDANCE OF THREATENED ELASMOBRANCHS AROUND AN OFFSHORE OIL FIELD IN AUSTRALIA

Target Journal: Conservation Biology

KEY WORDS: OFFSHORE PLATFORMS; *DE FACTO* MPA; IUCN RED LIST; PLATFORM ECOLOGY; STEREO-BRUVS

5.1 ABSTRACT

Human activity is degrading ecosystems around the world. Overfishing is ubiquitous and poses a threat to both target and non-target animals. Elasmobranchs are at particularly high conservation risk as a result of exploitation due to their conservative life histories, with most target fisheries for these animals assessed as unsustainable, and high mortality rates for elasmobranch bycatch. Offshore oil and gas platforms are productive marine ecosystems that support a wide range of species. These platforms act as both artificial reefs and fish aggregating devices, and can be classified as novel ecosystems. Offshore platforms may also function as *de facto* marine protected areas (MPAs) by excluding fishing activity which renders them potential refuges for species at risk from fishing. We here contrast the threatened elasmobranch community at the Wandoo oil platform and adjacent natural habitats in Northwest Australia, with those from other comparable regions across tropical Australia. The abundance of threatened elasmobranchs was higher around the offshore platform than most other regions, including locations in the Great Barrier Reef and Ningaloo Reef MPAs. The Wandoo platform is located in an area of commercial fishing activity, and many of the elasmobranchs observed at the Wandoo locations are captured as bycatch in the Pilbara Fish Trawl Interim Managed Fishery. Fishing is excluded around the Wandoo infrastructure, and we suggest that Wandoo acts as an important refuge from fishing pressure for these threatened elasmobranchs. A network of *de facto* MPAs created by offshore platforms in NW Australia may augment populations of threatened elasmobranchs both around the platforms and in adjacent natural habitats.

5.2 INTRODUCTION

Human activity is significantly altering Earth's natural habitats and threatening the survival of the species that depend on them. Globally, species are being lost at an unprecedented rate in the Holocene in what is now referred to as the sixth mass extinction (Ceballos et al., 2017; Dulvy et al., 2009; Turvey and Fritz, 2011). The number of threatened species, those classified as Vulnerable, Endangered or Critically Endangered on the International Union for Conservation of Nature (IUCN) Red List (Mace et al., 2008[a]), is rising every year. There are over 32,000 threatened species in 2020, more than double the 15,465 threatened species in 2010 (IUCN, 2020). Threatened species are those at highest risk of extinction, and these species are consequently an important consideration in conservation research and management (Mace et al., 2008).

Overfishing is arguably the biggest threat to ocean wildlife for both target and non-target species (Jackson et al., 2001). Catching power of industrial fisheries increased rapidly post World War II, with significant advances in technology allowing fleets to travel further and catch more fish (Tickler et al., 2018). Global catch peaked at 130 million tonnes in 1996 and has declined in the years since (Pauly and Zeller, 2016). Signs of overfishing are detectable in large marine ecosystems as far back as the 1800s (Roberts, 2007) and today, even remote coral reefs show signs of overexploitation (Coll et al., 2008; Greer et al., 2014). Approximately 4.5 billion people are dependent on the oceans for at least 15% of their protein consumption, particularly in developing nations, and this number is anticipated to grow as the global population increases and climate change threatens global food security (Béné et al., 2015). The effects of overfishing are exacerbated by various other human impacts, including climate change, coastal development, noise pollution, aquaculture, eutrophication, and ocean plastification (Jackson, 2010; McCauley et al., 2015; Perring and Ellis, 2013). Human activity is causing habitat loss and defaunation, with 29% of seagrasses, 30% of coral reefs, and 35% of mangroves lost or degraded (Jackson, 2010). Populations of large marine animals have on average declined by 89% (Lotze and Worm, 2009), and it is estimated that by 2050, 99% of seabirds will have ingested plastic in their lifetimes (Wilcox et al., 2015).

Elasmobranchs are particularly at risk in degraded oceans. Their K-selected life histories mean that they mature slowly and have low fecundity which translates to

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generally low productivity (Dulvy et al., 2008; Field et al., 2009). Each year, millions of elasmobranchs are either targeted for their meat, fins, livers, and gill plates, or captured as bycatch (Oliver et al., 2015). The global annual elasmobranch catch is approximately 1.7 million tonnes (Sadovy de Mitcheson et al., 2018), while the proportion of this catch estimated as sustainable is 200,000 tonnes, or 12 % (Simpfendorfer and Dulvy, 2017). Targeted shark fisheries have a poor record of sustainability and a significant portion of these shark populations will require decades to recover from overfishing (Stevens et al., 2000). Of the 1,084 elasmobranch taxa assessed by the IUCN, 213 species (20%) are listed as threatened (IUCN, 2020). A further 411 elasmobranch species (38%) are listed as Data Deficient, which should be afforded the same protection as threatened species until more information is available (Mace et al., 2008).

Australia's fisheries management is considered to be at the forefront globally (Ogier et al., 2016). However, many of Australia's shark populations are in decline due to overfishing and other destructive practices such as beach netting (Gibbs et al., 2020; Momigliano et al., 2014; Roff et al., 2018). Furthermore, Australia's MPAs are highly residual to commercial uses (Devillers et al., 2015), established largely offshore in non-representative habitats, and are predominantly comprised of multiple-use zones, with only 9.5% of Australian waters fully protected from extraction (Marine Conservation Institute, 2018). Multiple-use zones allow recreational and/or commercial fishing activity, and therefore only offer partial protection for marine species including elasmobranchs (Lynch et al., 2010; Sciberras et al., 2015). Partial protection has limited conservation outcomes relative to high levels of protection (Lester and Halpern, 2008; Sciberras et al., 2015) and may be even less effective for large predators.

There are 73 elasmobranch species in Australia listed as either Vulnerable, Endangered or Critically Endangered by the IUCN (IUCN, 2020). Of these species, 25 are also protected under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) Appendix II (CITES Secretariat, 2020). However, only six of Australia's internationally listed elasmobranch species are included under the national Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act;

Commonwealth of Australia, 1999). Furthermore, the EPBC Act allows for limited exports of certain species protected under CITES Appendix II, including the Endangered scalloped hammerhead *Sphyrna lewini* (up to 200 tonnes a year), and Critically Endangered great hammerhead *Sphyrna mokarran* (up to 100 tonnes a year). Australia also allows for recreational fishing of Endangered shortfin mako sharks *Isurus oxyrinchus* and longfin mako sharks *Isurus paucus* (Bruce et al., 2014; Commonwealth of Australia, 1999).

Studying threatened elasmobranchs is inherently challenging since they occur in low abundance and are highly mobile (Guttridge et al., 2017; Moore, 2015). Most data on threatened elasmobranchs is collected from dead animals in the form of data collected from commercial fisheries, fins and gill plates found in markets, and jaws and rostra kept as curios (Abercrombie et al., 2005; Moore et al., 2010; Morgan et al., 2011; Pank et al., 2001; Stobutzki et al., 2002). However studying these species while they are still alive, using non-invasive techniques, is possible. Baited remote underwater video systems (BRUVS) have recorded a variety of threatened elasmobranchs from locations all over the world, from scalloped hammerheads in Fiji (Brown, 2014), to green sawfish *Pristis zijsron* in Australia (Bond et al., 2018), to leopard sharks *Stegostoma tigrinum* in the Red Sea (Spaet et al., 2016). BRUVS have also proved useful for recording novel behaviours and identifying hotspots for these species (Birt et al., 2019; Letessier et al., 2019).

Offshore oil and gas platforms (hence offshore platforms) function as artificial reefs (Shinn, 1974) and are increasingly recognised as important marine habitats (Claisse et al., 2014; Sommer et al., 2019). These platforms create novel ecosystems with attributes not present at the site pre-installation, and are characterised by a shift in marine communities and generation of beneficial ecosystem services (Sommer et al., 2019; van Elden et al., 2019). Offshore platforms often exclude various fishing activities, effectively creating *de facto* marine protected areas (MPAs) that provide a refuge for marine wildlife. Various threatened elasmobranchs have been observed around offshore platforms and associated infrastructure, including grey nurse sharks *Carcharias taurus*, scalloped hammerheads, and great hammerheads (Franks, 2000;

McLean et al., 2018, 2019; Robinson et al., 2013). There have also been reports of aggregations of threatened elasmobranchs around offshore platforms, including whale sharks *Rhincodon typus* in Qatar and porbeagle sharks *Lamna nasus* in the North Sea (Haugen and Papastamatiou, 2019; Robinson et al., 2013).

We investigate whether an offshore platform in Northwest (NW) Australia, the Wandoo oil field that lies approximately 75 km off the NW coast, could be acting as a refuge for threatened elasmobranchs across tropical Australia. We harnessed two curated stereo-BRUVS databases, one for demersal and one for pelagic taxa, to examine the abundance of threatened elasmobranchs around the Wandoo oil field infrastructure and two associated natural habitats, relative to abundances across five comparable regions around Australia with a geographical span of approximately 16 degrees of latitude and 50 degrees of longitude. We tested the hypothesis that the novel ecosystem that has emerged at the Wandoo Field, which has functioned as a *de facto* MPA for over 25 years, is acting as a refuge for threatened elasmobranchs.

5.3 Materials and Methods

Video-based sampling of elasmobranchs

Our analysis is based on video imagery derived from stereo-BRUVS. Stereo-BRUVS are an established, non-destructive sampling method for studying the distribution, abundance, biomass and diversity of marine fauna (Barley et al., 2017; Cappo et al., 2006; Watson et al., 2010). More recently, stereo-BRUVS have also documented rare and highly mobile species (Letessier et al., 2013, 2015a; Thompson et al., 2019). Stereo-BRUVS are relatively inexpensive, allowing them to be deployed across large spatial scales (Letessier et al., 2013, 2015b), and they have been developed to sample both benthic and mid-water habitats (Letessier et al., 2013; Whitmarsh et al., 2017). BRUVS-derived data should be interpreted recognising the potential higher representation of piscivores (Lowry et al., 2012), the potential variability of bait plumes (Whitmarsh et al., 2017). Depsite these constraints, BRUVS can be used to document clear signals in marine communities relative to other methods (Cappo et al., 2006; Lowry et al., 2012).

Stereo-BRUVS, whether used on the seabed or in mid-water habitats, share a common design. Seabed stereo-BRUVS consist of a 95 cm long aluminium horizontal base bar that supports two small action (e.g. GoPro) video cameras. The video cameras are mounted 80 cm apart and converge to a common focal point at an angle of four degrees per camera, and each camera records for a minimum of 60 min. The stereo-BRUVS are baited with ~800 g of pilchards *Sardinops* sp. in a bag made of either plastic coated wire or galvanised steel mesh. Bait is suspended on a pole 1.5 m in front of the cameras (Supplementary Fig. 5.1a). Each camera is set to record in medium field of view to maximise the area in the video frame and to increase rates of detection to a distance up to 8 m. Seabed stereo-BRUVS are deployed individually on the seabed with a minimum of 200 m between stations. Mid-water stereo-BRUVS use the same basebar as seabed stereo-BRUVS. The basebar is mounted on a 1.45 m long steel upright to provide stability in the water column (Supplementary Fig. 5.1b). Mid-water stereo-BRUVS are baited with 1 kg of crushed pilchards, contained in a perforated PVC canister. The canister is mounted 1.5 m in front of the cameras on a steel bait arm, which acts as a rudder to minimise rotation and maintain a down-current field of view for the duration of the deployment. Mid-water stereo-BRUVS are suspended 10 m below the surface and each camera records for a minimum of 120 minutes and are generally deployed in long-lines (strings) of five rigs separated by 200 m of line. However, rigs were randomly moored in sets of five within stratified zones in the Wandoo field to avoid entanglement with infrastructure.

Established stereo-BRUVS protocols are followed pre-survey, during deployments and post-survey to ensure consistency in obtaining and recording stereo-BRUVS imagery (Bouchet et al., 2018; Langlois et al., 2018). Prior to fieldwork, stereo-BRUVS are calibrated in an enclosed swimming pool using the CAL software (SeaGIS Pty Ltd, 2020), following standard protocols (Harvey and Shortis, 1998). Collected videos are converted to AVI format using Xilisoft Video Converter Ultimate (Xilisoft Corporation, 2016) before being imported into the EventMeasure software package (SeaGIS Pty Ltd, 2020) for processing. Prior to the deployment of each BRUV in the field, a slow hand clap is recorded in the shared field of view to enable synchronising of the left and right cameras videos in the lab prior to processing. Processing commences either once the seabed stereo-BRUVS have settled on the substrate or once the mid-water stereo-BRUVS have stabilised at the set depth of 10 m. All animals are identified to the lowest possible taxonomic level. Relative abundance is estimated as the maximum number of individuals of a given species in a single frame (MaxN; Cappo et al., 2006).

We accessed the seabed and mid-waters BRUVS databases curated by the Marine Futures Lab (https://meeuwig.org/resources). To reflect known latitudinal gradients in biodiversity (Forster, 1778), our analysis was restricted to regional, largely tropical locations, comparable to the Wandoo locations, with a latitude range of 16 degrees (9.9 to 26.3 °S) and longitude range of 50 degrees (96.8 to 146.8 °E). Demersal elasmobranch data were extracted from the seabed database of 3,920 seabed stereo-BRUVS deployments collected at 26 locations over 33 surveys. The data for pelagic elasmobranchs were extracted from 2,268 two hour video samples from 26 locations over 33 surveys (Supplementary Tables 1 and 2). These locations were assigned to six regions (Fig. 5.1), roughly corresponding to the Western Australian (WA) bioregions as defined by the WA Department of Primary Industries and Regional Development (DPIRD; Gaughan and Santoro, 2019), and the Integrated Marine and Coastal Regionalisation of Australia (IMCRA) provincial bioregions (Commonwealth of Australia, 2006). The regions are: Northeast (NE), which encompasses IMCRA's Northeast IMCRA transition and Cape Province; the remote offshore Australian territory of the Cocos-Keeling Islands (CI); Northwest (NW), corresponding to DPIRD's North Coast with latitudes less than 21.46° S and including the Muiron Islands; Central North (CN), corresponding to IMCRA's Central Western IMCRA Transition (21.46° S to 24° S); and Central South (CS), corresponding to IMCRA's Central Western IMCRA Province (24° S to 27° S). The Wandoo locations were classified separately from the NW region in which they are located to allow them to be contrasted against the other locations. The Wandoo locations include both artificial and natural habitats. The Wandoo Platform location (WP) is located 75 km northwest of Dampier, Western Australia (Fig. 5.2). The infrastructure at the Wandoo Platform includes: a catenary anchored leg mooring (CALM) buoy, secured by six moorings around the pipeline end manifold (PLEM); Wandoo A, an unmanned monopod platform consisting of production infrastructure with a helideck supported by a 2.5 m diameter shaft; and Wandoo B, a concrete gravity structure (CGS) with a caisson measuring 114 m long by 69 m wide, and four shafts 11 m in diameter supporting the superstructure 18 m above the surface (Fig. 5.3). The Wandoo Reef (WR) location is located approximately 15 km west of the Wandoo Platform, and is characterised by a natural rocky reef rising approximately 30 m from the seafloor. It was chosen as a natural comparison of the artificial structure that is the Wandoo field. The Wandoo Sand (WS) location is situated approximately 15 km northeast of WP and is characterised by little to no physical relief and a dense, silty sand habitat. This habitat is likely similar to the Wandoo field prior to the installation of infrastructure.

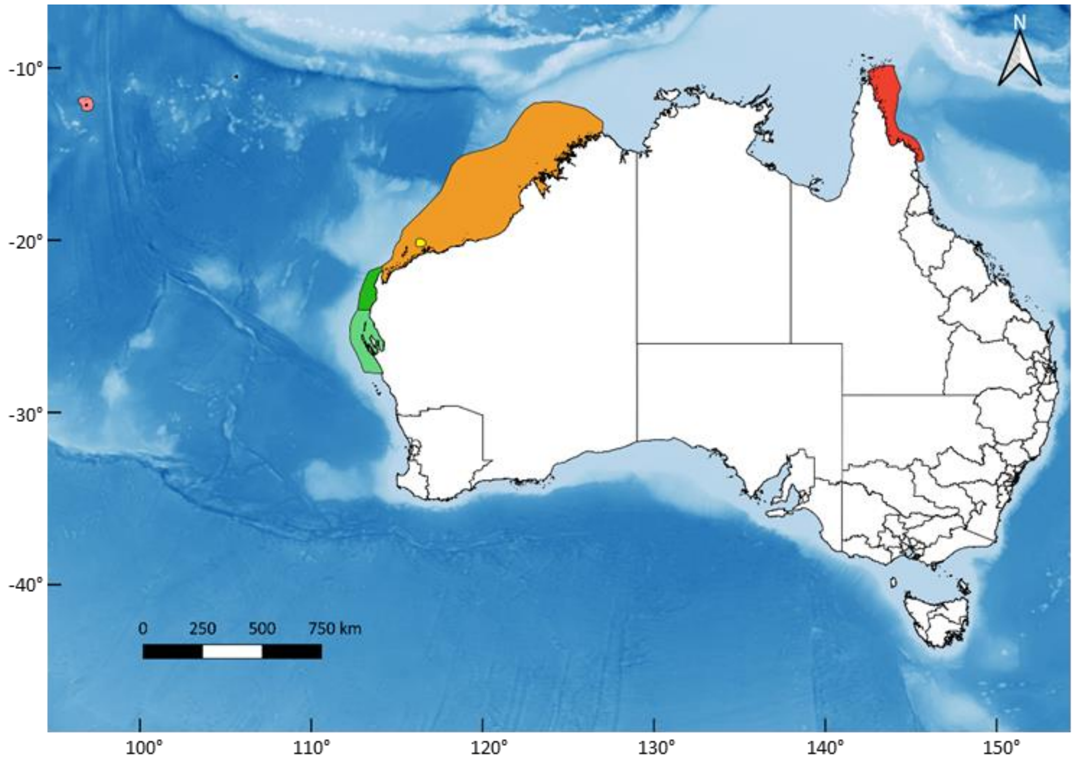

Figure 5.1 Location of the study regions around Australia: Cocos-Keeling Islands (pink; to the northwest of the Australian mainland); Northeast (red); Northwest (orange); Wandoo (yellow) Central North (dark green); and Central South (light green).

All stereo-BRUVS were deployed according to standard practices (Bouchet et al., 2018; Langlois et al., 2018). Sampling occurred during daylight hours to minimise any effects of crepuscular animal behaviour. We used a generalised random tessellation stratified (GRTS) approach (Stevens and Olsen, 2004) or random stratified approach (Kenkel et al., 1989), depending on the purpose of each survey. Surveys were conducted under UWA ethics permit RA/3/100/1484 and, if conducted on private vessels, under exemptions from the Australian Maritime Safety Authority (EX2016/A185; EX2017/A007). All necessary jurisdictional permits were obtained.

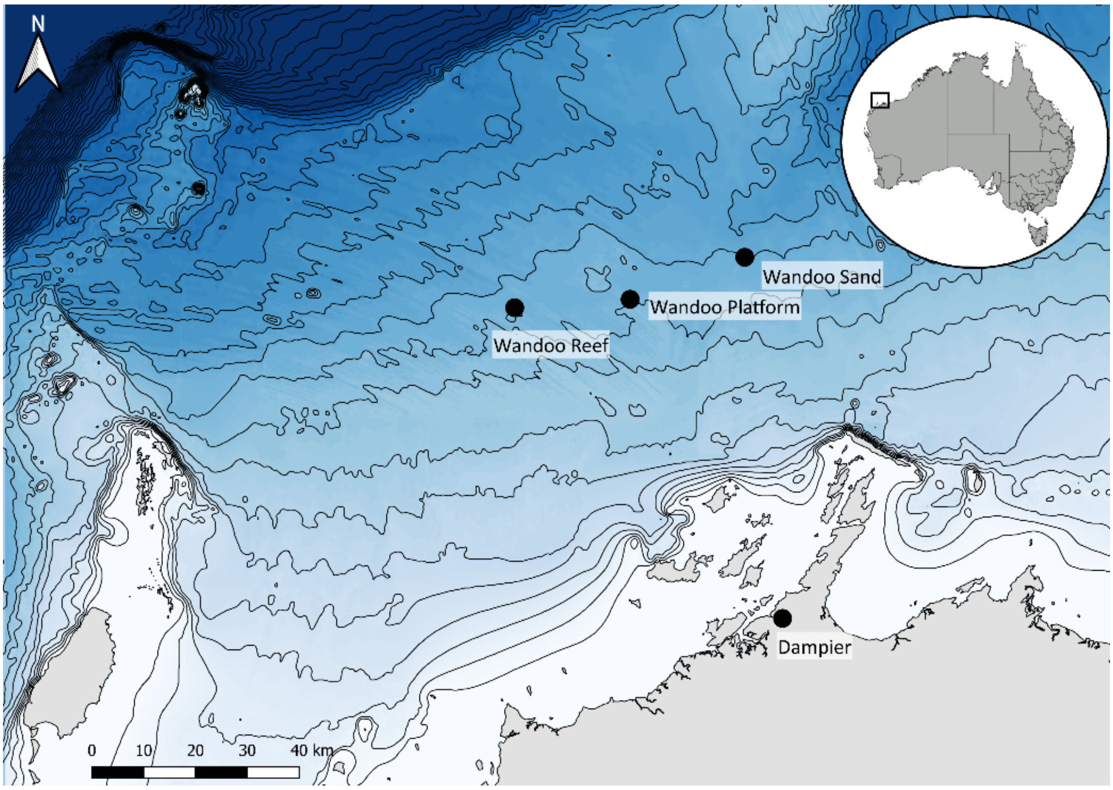

Figure 5.2 Location of the Wandoo Platform and the two nearby natural locations, Wandoo Reef and Wandoo Sand, approximately 75 km north-west of Dampier, Western Australia

Elasmobranch analyses

All elasmobranch records from the tropical regions were extracted from the demersal and pelagic databases. Taxa observed on seabed stereo-BRUVS were classified as 'demersal taxa', even though some taxa recorded on seabed stereo-BRUVS were not necessarily demersal species. The same approach was applied to taxa observed on mid-water stereo-BRUVS which were classified as 'pelagic taxa' regardless of their habitat. The latest IUCN Red List classifications (accessed 27th August 2020) were obtained and used to classify all elasmobranch records according to the seven IUCN classifications: Critically Endangered (CR); Endangered (EN); Vulnerable (VU); Near Threatened (NT); Data Deficient (DD); Least Concern (LC); and Not Evaluated (NE) (IUCN, 2020). Where taxa were only identified to genus (9%) or family (2%), the most common IUCN classification was used for species in that genus or family known to occur at that location based on reported distributions accessed via Aquamaps (Kaschner et al., 2019). Threatened elasmobranchs, comprised of taxa listed as CR, EN and VU, were extracted from the demersal and pelagic databases as the focus of the analysis.

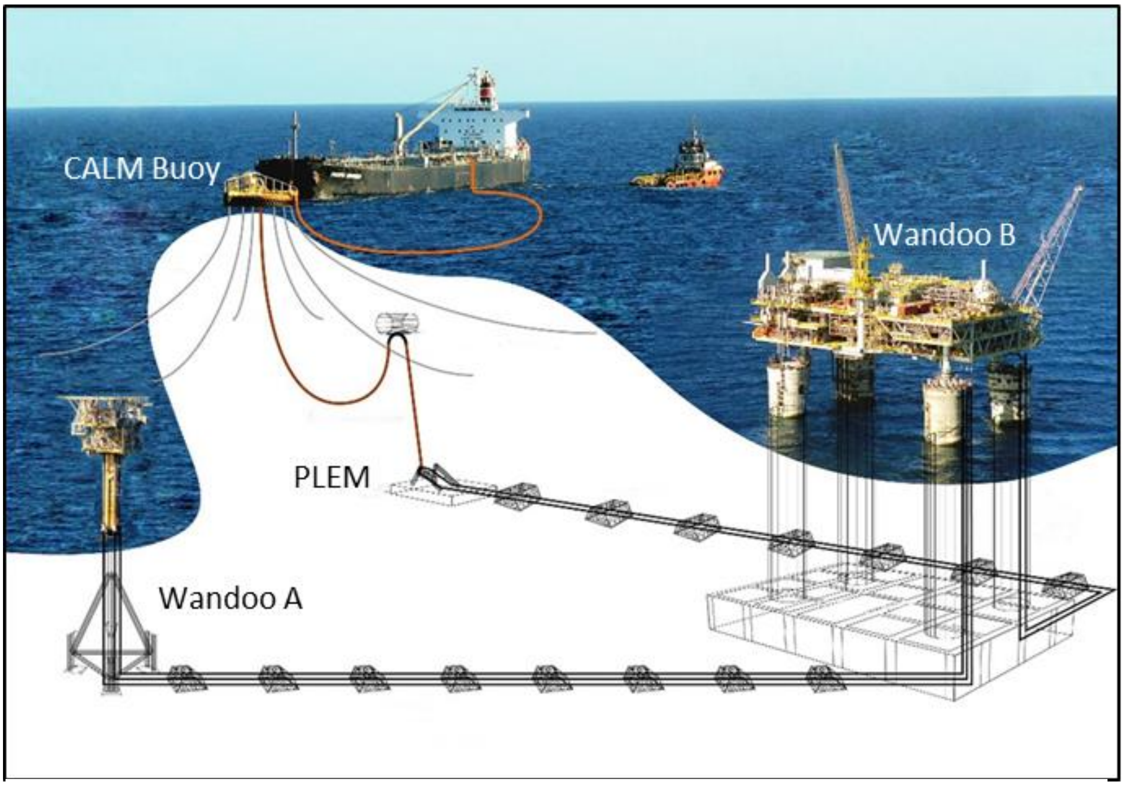

Figure 5.3 Wandoo oil field schematic adapted from Vermilion Oil and Gas Australia (2014). The infrastructure at the Wandoo field includes the unmanned monopod Wandoo A, the concrete gravity structure Wandoo B, the pipeline end manifold (PLEM), and the catenary anchored leg mooring (CALM) Buoy. Not to scale.

Our analysis was based on total abundance (TA) of elasmobranchs as we were interested in numeric abundance rather than size-based metrics such as length or weight. Total abundance of threatened elasmobranchs was calculated for each sample as the sum of abundances for all taxa on that sample. For demersal elasmobranchs, TA_D was calculated for each individual deployment. For pelagic elasmobranchs, TA_P was calculated as the mean abundance of the five deployments within the string or across each moored set in the case of the Wandoo locations. All samples with no threatened elasmobranchs were retained as zeros so that the mean abundances for locations reflected absences as well. For each taxa observed at the Wandoo locations, mean taxa-specific abundances were calculated for all locations. For each location, TA_D and TA_P were calculated as the average of all samples from that location as the basis for assessing differences between regions.

The demersal and pelagic abundance data were compared between regions using both univariate and multivariate analyses. Total elasmobranch abundances at the three Wandoo locations were compared with those of the other regions using Wilcoxon Signed Rank tests with regions as replicates (Zar, 1999). The Wilcoxon Signed Rank test was also used to contrast the abundance of each threatened elasmobranch taxa at the Wandoo locations to their abundances across all other locations. For the multivariate analyses, we tested for differences in the composition of threatened elasmobranchs at the Wandoo locations relative to the other regions for both demersal and pelagic elasmobranchs. The abundance data for each location were log(x+1) transformed to increase the influence of rare taxa and reduce the influence of common taxa, and a Bray-Curtis resemblance matrix was calculated. A one-way permutational analysis of variance (PERMANOVA) was applied based on unrestricted permutations with “region” as the factor, followed by post-hoc paired tests where appropriate (Anderson, 2001). A canonical analysis of principal coordinates (CAP) was used to visualise a constrained ordination of the data for both the demersal and pelagic composition data.

Environmental drivers

We compiled a database of anthropogenic, physical, chemical and biological oceanographic variables for the seabed and mid-water locations (Supplementary Tables 3 and 4 respectively) to determine whether there were underlying environmental or anthropogenic drivers of the elasmobranch abundances. Anthropogenic variables using travel time were based on human accessibility assessments undertaken by Maire et al. (2016). Distance to market and population were computed using the LandScan 2016 database (Dobson et al., 2000), while distances to marine features were computed using bathymetry data (Yesson et al., 2020). Environmental data were derived from the following datasets:

- Geoscience Australia (GA) 250 m bathymetry (Whiteway, 2009);
- GA Australian submarine canyons (Huang et al., 2014);
- CSIRO Atlas of Regional Seas (CARS) (Ridgway et al., 2002); and
- Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS) (IMOS, 2020)

The degree of colinearity amongst these independent variables was calculated using Pearson's correlation coefficient (Kirch, 2008) such that if variables were highly correlated (r>0.6), only one of the pair was retained. We then examined the influence of these variables on the abundance of threatened elasmobranchs at the level of location, using a distance-based linear model (DistLM) (Anderson et al., 2015). All analyses were completed using the Primer 7 software package with the PERMANOVA + add-on (Anderson et al., 2015).

5.4 Results

Elasmobranchs across Australia's tropics were diverse and numerous. Across all regions, we counted 5,360 elasmobranchs from 93 taxa, representing 18 families. Threatened elasmobranchs accounted for 866 individuals (16%) from 35 taxa (38%), representing 15 families (83%). In the demersal dataset, we counted 3,538 individuals from 85 elasmobranch taxa, representing 17 families. Threatened elasmobranchs accounted for 592 individuals (17%) from 27 threatened elasmobranch taxa (31%), representing 12 families (71%) (Supplementary Table 5.5). The remaining taxa were classified as: NT (16%), DD (5%), LC (21%), and NE (27%). Within the threatened taxa, the majority were classified as Vulnerable (55%), followed by Critically Endangered (35%) and Endangered (10%). Wedgefish *Rhychobatus* sp. was the most prevalent threatened taxon, recorded on 127 deployments. At the Wandoo locations, we recorded 313 elasmobranchs (6% of all elasmobranchs recorded) from 27 taxa (29%), representing ten families (6%). Threatened elasmobranchs accounted for 89 individuals (28%) from 13 taxa (48%), representing eight families (80%). The remaining taxa at Wandoo were classified as: NT (26%), DD (7%), LC (4%), and NE (15%). Vulnerable species comprised 48% of all threatened elasmobranch taxa at the Wandoo locations, followed by Critically Endangered (38%) and Endangered (14%) species. The most prevalent taxa at the Wandoo locations were requiem shark *Carcharhinus* sp. and wedgefish, each recorded on 24 deployments.

In the pelagic dataset we counted 1,822 individuals from 28 elasmobranch taxa, representing six families. Threatened elasmobranchs accounted for 1,041 individuals (56%) from 14 taxa (50%), representing five families (83%) (Supplementary Table 5.6). The remaining taxa were classified as: NT (28%), DD 11%), and LC (11%). Within the threatened taxa 55% were Vulnerable, while Endangered accounted for 34% and Critically Endangered accounted for 11%. The most prevalent threatened taxon was the requiem shark, recorded on 128 strings. At the Wandoo locations, we recorded 417 elasmobranchs (22% of all elasmobranchs recorded) from 17 taxa (61%), representing three families (50%). Threatened elasmobranchs accounted for 302 individuals (72%) from seven taxa (41%), representing three families (100%). The remaining taxa at Wandoo were classified as: NT (35%), DD (12%), and LC (12%). The threatened elasmobranchs at the Wandoo locations were mostly Vulnerable (73%), followed by Endangered (22%) and Critically Endangered (5%). The most prevalent threatened taxon at Wandoo was also the requiem shark, recorded on 60 strings.

Comparing threatened elasmobranch abundance between regions

Abundance of threatened elasmobranchs varied significantly between regions in both the demersal and pelagic datasets. Mean abundance of demersal threatened elasmobranchs was generally higher at the individual Wandoo locations than at most other regions, across which abundance ranged from 0.003 in the Cocos-Keeling Islands region to the highest value of 0.08 at the Wandoo Sand location (Fig. 5.4a). The remaining Wandoo locations, Wandoo Reef (TA_D=0.06) and Wandoo Platform (TA_D=0.05), had the second and fourth highest abundances respectively. Abundance of both Vulnerable and Endangered taxa were highest at Wandoo Sand (0.05 and 0.01 respectively), while abundance of Critically Endangered taxa was highest at Wandoo Reef (0.04). In terms of the individual Wandoo locations, Wandoo Platform was significantly higher than the other regions in the abundance of Endangered taxa and did not differ in the abundance of Vulnerable, Critically Endangered, and combined threatened taxa (Table 5.1). Wandoo Sand had significantly higher abundance of

Vulnerable, Endangered, and combined threatened taxa than the other regions, and did not differ in the abundance of Critically Endangered taxa. Wandoo Reef had significantly higher abundance of Critically Endangered taxa, and did not differ from the other regions in the abundance of Vulnerable, Endangered or combined threatened taxa.

Mean abundance of pelagic threatened taxa was also generally higher at the individual Wandoo locations, with abundance across all regions ranging from 0.07 in the Cocos-Keeling Islands region to 0.22 at Wandoo Sand (Fig. 5.4b). Pelagic abundance was highest at Wandoo Sand (0.22), followed by Central South (0.16), Wandoo Platform (0.15), and Wandoo Reef (0.15). Abundance of Vulnerable taxa was highest at Wandoo Sand (0.18), while abundance of Endangered taxa was highest at Central South (0.10). Abundance of Critically Endangered taxa was highest at Northeast (0.01). In terms of the individual Wandoo locations, Wandoo Platform did not differ from the other regions in the abundance of Vulnerable, Endangered, or combined threatened taxa, and was significantly lower in the abundance of Critically Endangered taxa (Table 5.1). Abundance at Wandoo Sand was significantly higher than the other regions in terms of both Vulnerable and combined threatened taxa, and did not differ in the abundance of either Endangered or Critically Endangered taxa. Abundance at Wandoo Reef was higher than the other regions in terms of Critically Endangered taxa, and did not differ in the abundance of Vulnerable, Endangered or combined threatened taxa.

The taxa-specific comparisons of the abundances of the 13 demersal and six pelagic taxa recorded at the Wandoo locations indicated higher numbers for only a limited number of species. In terms of demersal taxa, Wandoo Platform had higher abundance of leopard sharks (EN) than the other locations ($Z = -2.98$; $p = 0.003$). Wandoo Sand had higher abundance of silky sharks *Carcharhinus falciformis* (VU) ($Z = -2$; $p = 0.046$) and requiem sharks ($Z = -3.26$; $p = 0.001$), and Wandoo Reef had higher abundance of wedgefish (CR) ($Z = -3.23$; $p = 0.001$). In terms of pelagic taxa, only requiem sharks (VU) were higher at the Wandoo locations than at other locations and this held for each location: Wandoo Platform ($Z = -2.08$; $p = 0.038$), Wandoo Sand ($Z = -3.09$; $p = 0.002$), and Wandoo Reef ($Z = -2.47$; $p = 0.013$).

Table 5.1 Wilcoxon Signed Rank tests comparing mean abundance of demersal and pelagic threatened elasmobranchs at the three Wandoo locations with the means of the other tropical regions. Tests were conducted for Vulnerable (VU), Endangered (EN) and Critically Endangered (CR) taxa, as well as all of these taxa combined (Total). P-values in bold and marked with an asterisk are < 0.05.

		VU	EN	CR	Total
Demersal					
Wandoo Platform	Z	-0.423	-2.113	-0.085	-0.592
	P-value	0.673	***0.035**	0.933	0.554
Wandoo Sand	Z	-2.282	-2.282	-0.592	-2.282
	P-value	***0.022**	***0.022**	0.554	***0.022**
Wandoo Reef	Z	-0.761	-0.930	-2.282	-1.606
	P-value	0.447	0.353	***0.022**	0.108
Pelagic					
Wandoo Platform	Z	-1.099	-0.423	-2.113	-1.099
	P-value	0.272	0.673	***0.034**	0.272
Wandoo Sand	Z	-2.282	-0.085	-1.268	-2.282
	P-value	***0.022**	0.933	0.205	***0.022**
Wandoo Reef	Z	-1.268	-0.592	-2.113	-0.930
	P-value	0.205	0.554	***0.035**	0.353

Differences in threatened elasmobranch community assemblages

Our analysis showed weak differences in the composition of threatened elasmobranchs between the locations in the Wandoo field and the regional locations. Following exploratory analyses, we excluded the offshore Cocos-Keeling Islands region as it was strongly separated from the other tropical regions, and this separation overwhelmed the differences in taxonomic assemblages among the other regions. Demersal assemblages did not show strong separation among the regions, with no significant differences between the locations in the PERMANOVA ($p = 0.242$). Some regions still showed spatial separation and were characterised by different threatened taxa (Fig. 5.5a). The demersal assemblage at Wandoo Platform was characterised by leopard sharks, and was more similar to the Northeast and Northwest regions than to Wandoo Sand and Wandoo Reef. Wandoo Sand was characterised by requiem sharks while Wandoo Reef was characterised by wedgefish. Locations in the Northeast, Northwest, and Central North regions did not show any strong similarity within their regions, and the assemblages in these regions were similar, characterised by leopard sharks and tawny nurse sharks *Nebrius ferrugineus* (VU). Locations in the Central South region were more distinct from other regions, characterised by bentfin devilrays *Mobula thurstoni* (EN).

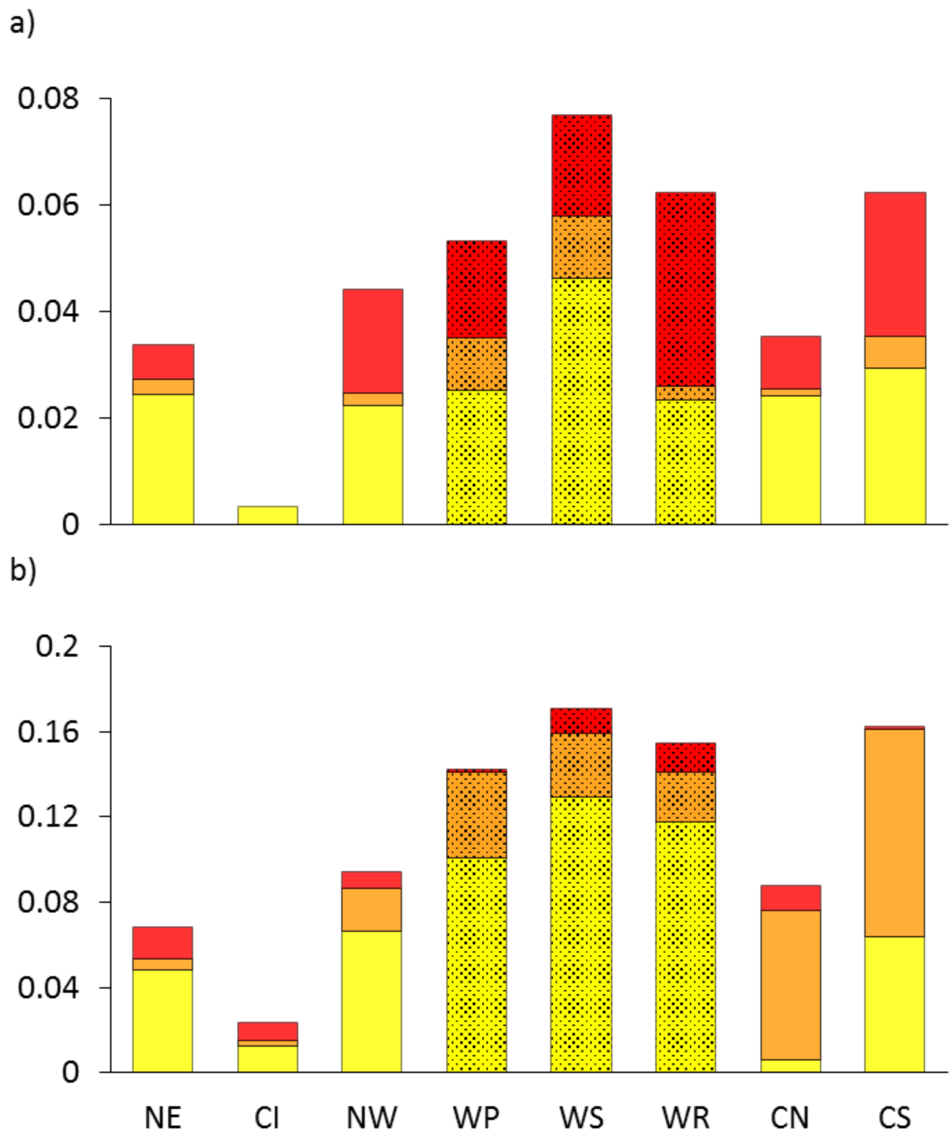

Figure 5.4 Demersal (a) and pelagic (b) abundance of threatened elasmobranchs by region: Northeast (NE); Cocos-Keeling Islands (CI); Northwest (NW); Wandoo Platform (WP); Wandoo Sand (WS); Wandoo Reef (WR); Central North (CN); and Central South (CS). Classifications depicted are Vulnerable (yellow); Endangered (Orange) and Critically Endangered (Red). Patterned bars indicate the Wandoo locations.

Pelagic taxonomic assemblages did not show strong separation In terms of abundance, with no significant differences between the assemblages in the PERMANOVA ($p = 0.263$), but spatial separation was observed between some regions (Fig. 5.5b). The pelagic assemblage at Wandoo Platform was characterised by mobula rays (VU) and sandbar sharks (VU), and was similar to the nearby natural locations Wandoo Sand and Wandoo Reef, Northeast, and one Northwest location (Long Reef West). These locations were characterised by mobula rays and great hammerheads (CR). Northwest showed weak grouping, with locations in this region characterised by great hammerheads and silvertips *Carcharhinus albimarginatus* (VU). Central North and Central South were each more separated from other regions, with Central North characterised by oceanic whitetips *Carcharhinus longimanus* (CR) and shortfin makos (EN), and Central South characterised by dusky sharks (EN).

Environmental and anthropogenic drivers

The environmental and anthropogenic variables explained a significant proportion of the variation in both the demersal and pelagic taxonomic assemblages in terms of abundance. The most parsimonious DistLM model explained 26% of the variation in the demersal taxonomic assemblages, with the most influential variables being depth, dissolved oxygen (O_2), and sea surface temperature (SST) (Table 5.2). Demersal assemblages were predominantly influenced by environmental variables (O_2 and SST, 15%), followed by physical variables (depth, 11%). In terms of the pelagic taxonomic assemblages, the most parsimonious DistLM model explained 41% of the variation, driven by travel time to market (TT_Market), distance to port (DistPort), and chlorophyll concentration (Chl-a) (Table 5.2). Pelagic assemblages were predominantly influenced by anthropogenic variables (TT_Market and DistPort, 29%), followed by environmental variables (Chl-a, 12%).

Table 5.2 Distance-based linear model (DistLM) results based on the most parsimonious model predicting abundance of demersal and pelagic threatened elasmobranchs. Variables included are: depth (m); dissolved oxygen (O_2; mmol/L); sea surface temperature (SST; °C); travel time to market (TT_Market; mins) distance to port (DistPort; km); Chlorophyll concentration (Chl-a; mg/m3). The degrees of freedom (d.f.) are reported in parentheses after the Pseudo-F value. proportion of variation in abundance explained by each variable (Prop.) and cumulative proportion of variation explained by the variables (Cumul. Prop.) are also included.

Variable	SS(trace)	Pseudo-F (d.f.)	P	Prop.	Cumul. Prop.
Demersal					
Depth	6,435	2.9 (21)	0.005	0.11	0.11
O_2	5,465	2.2 (21)	0.031	0.09	0.20
SST	3,785	1.5 (21)	0.141	0.06	0.26
Pelagic					
TT_Market	8,461	3.0 (15)	0.007	0.17	0.17
DistPort	6,206	2.4 (15)	0.025	0.12	0.29
Chl-a	5,856	2.5 (15)	0.039	0.12	0.41

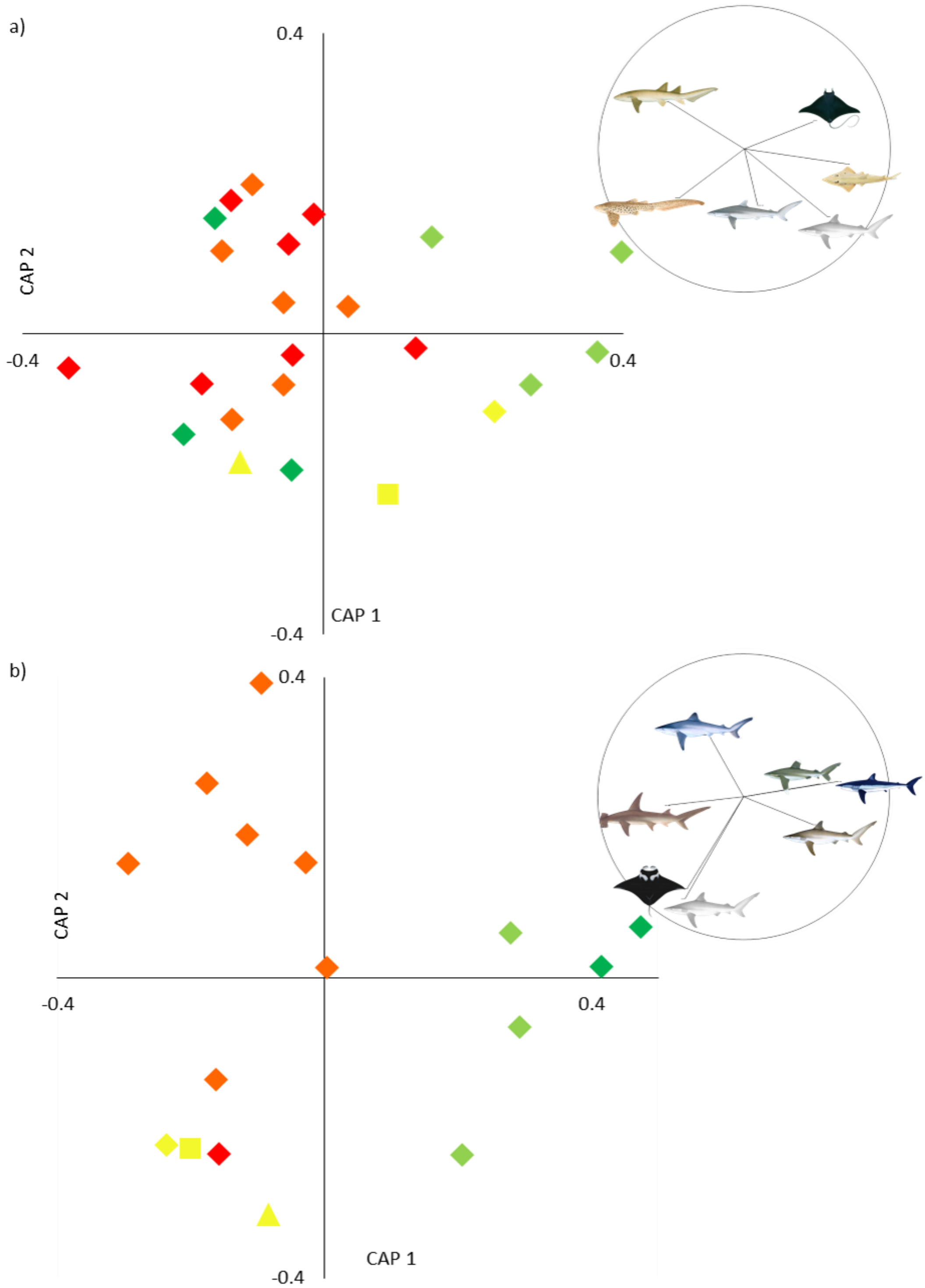

Figure 5.5 Canonical analysis of principal coordinates (CAP) for abundance of (a) demersal and (b) pelagic taxonomic assemblages. Locations shown are: Northeast (Red); Northwest (Orange); Wandoo Platform (yellow triangle); Wandoo Reef (yellow diamond); Wandoo Sand (yellow square); Central North (dark green); and Central South (light green). Taxa clockwise from top in (a) are: bentfin devilray *Mobula thurstoni*; wedgefish *Rhychobatus* sp.; requiem shark *Carcharhinus* sp.; silky shark *Carcharhinus falciformis*; leopard shark *Stegostoma tigrinum*; and tawny nurse shark *Nebrius ferrugineus*. Taxa clockwise from top in (b) are: oceanic whitetip *Carcharhinus longimanus*; dusky shark *Carcharhinus obscurus*; sandbar shark *Carcharhinus plumbeus*; requiem shark; mobula ray *Mobula* sp.; great hammerhead *Sphyrna mokarran*; and silvertip shark *Carcharhinus albimarginatus*. Images © R. Swainston/anima.fish

5.5 Discussion

The Wandoo oil field and adjacent natural habitats had elevated abundance of threatened elasmobranchs compared with other tropical regions around Australia. Wandoo is located within the Pilbara Offshore meso-scale region, which was subjected to decades of destructive seabed trawling activity (Sainsbury et al., 1993), and is still targeted by three commercial fisheries as well as recreational fishing activity (WAFIC, 2020). Despite this fishing pressure, there were more threatened elasmobranchs at the Wandoo locations than at locations on the Ningaloo Reef and the Great Barrier Reef, both of which are managed as multiple-use MPAs. Several threatened taxa were found in higher abundance at the Wandoo locations than other regions, suggesting that this area could be of particular importance for elasmobranchs such as silky sharks (VU), leopard sharks (EN), and wedgefishes (CR).

The demersal taxonomic assemblages varied between the Wandoo locations, despite environmental similarities between these sites. It is likely that the distinction in the Wandoo Platform demersal assemblage is driven by the presence of the artificial infrastructure. Offshore platforms have a significant impact on demersal communities by creating greater habitat complexity that results in higher fish diversity and production (Claisse et al., 2014; Love et al., 2019). In the case of the Wandoo Platform location, the installation of the infrastructure resulted in a change in in both habitat composition and community assemblages from what would have existed pre-installation (van Elden et al. *in prep*) and consequently may provide unique habitat and ecosystem services for demersal threatened elasmobranchs. The novel nature of this location was further emphasised by the presence of two threatened species at Wandoo Platform that were not recorded at either of the nearby natural locations: round ribbontail ray *Taeniurops meyeni* (VU) and giant shovelnose ray *Glaucostegus typus* (CR). The habitat at these two natural locations was generally sandy with little to no macrobenthos cover (van Elden et al. *in prep*). This is due to habitat modification from trawling activity, which would have historically removed macrobenthos (Sainsbury et al., 1993), particularly at Wandoo Sand, but also in the flat areas surrounding the Wandoo Reef. However, this sandy habitat remains important for various threatened demersal elasmobranchs including leopard sharks, wedgefishes, and bowmouth guitarfish *Rhina ancylostoma* (CR) (Compagno, 1984). It is likely that the presence of various undisturbed habitats over a relatively small spatial scale, including sandy and macrobenthos habitats, natural reefs, and artificial reefs, create a complex network of valuable habitats for demersal threatened elasmobranchs.

The pelagic assemblages were similar across the Wandoo locations. This similarity is to be expected given the relatively close proximity of these sites, and the highly mobile nature of pelagic elasmobranchs (Andrzejaczek et al., 2020; Bonfil, 2008). Offshore platforms provide hard substrate vertically through the water column to the surface, which provides a unique physical environment not present in most natural habitats (Todd et al., 2018). Offshore platforms also function as fish aggregating devices (FADs; Franks 2000), attracting pelagic fishes which are a food source for threatened elasmobranchs such as dusky sharks, great hammerheads and sandbar sharks (Compagno, 1984), all of which were observed at the Wandoo locations. All three of these species are also known to feed on demersal species (Compagno, 1984), which are also found throughout the water column on the shafts of the Wandoo platforms (Tothill, 2019). It is likely that pelagic elasmobranchs, which are highly mobile, could be utilising all of the habitats in and around the Wandoo field for feeding and refuge from predators.

Fishing activity has been identified as one of the key human-driven pressures on marine environments in NW Australia (Anon., 2018). We found that anthropogenic variables related to fishing activity, namely travel time to market and distance to port, were the most influential factors impacting pelagic assemblages. There were 12 threatened elasmobranch taxa recorded at the Wandoo locations (excluding unidentified requiem sharks) and of these, nine were reported as bycatch in the Pilbara Fish Trawl Interim Managed Fishery: sandbar sharks, round ribbontail rays, mobula rays, and tawny nurse sharks (all Vulnerable); leopard sharks (Endangered); and wedgefish, great hammerheads, bowmouth guitarfish and giant shovelnose rays (all Critically Endangered) (Jaiteh et al., 2014; Western Australia Department of Fisheries, 2010). Fishing mortality, both immediate and post-release, varies greatly in elasmobranchs (Ellis et al., 2017). Dapp et al. (2016) predicted that average mortality in trawl fisheries was 41.9% in stationary-respiring elasmobranchs (e.g. leopard and tawny nurse sharks) and 84.2% for obligate ram-ventilating species (e.g. sandbar and hammerhead sharks). However, in the case of the Pilbara Fish Trawl Interim Managed Fishery, independent observations of 44 trawls found that 91% of sharks and 66% of batoids captured in trawls were dead and consequently discarded (Jaiteh et al., 2014). As such, this fishery represents an ongoing and significant risk to threatened elasmobranchs.

The Wandoo field and adjacent natural habitats had higher abundance of threatened elasmobranchs than locations at the Ningaloo Reef and Great Barrier Reef MPAs. Both of these MPAs are predominantly multiple-use, with only 34% of the Ningaloo Reef MPA and 33% of the Great Barrier Reef MPA protected in no-take zones (CALM and MPRA, 2005; Fernandes et al., 2005). MPAs that are multiple-use offer only partial protection from extractive activities, and are significantly less effective than no-take MPAs (Edgar et al., 2014; Lester and Halpern, 2008; Sciberras et al., 2015). Fish populations in partially protected zones of the Ningaloo and Great Barrier Reef MPAs show fishing-related impacts not observed in no-take zones (Fraser et al., 2019; McCook et al., 2010; Westera et al., 2003). Elasmobranchs are also at risk in multiple-use marine parks. In the Great Barrier Reef MPA, various elasmobranch species continue to be caught in the Queensland East Coast Inshore Finfish Fishery (Harry et al., 2011) as well as by recreational fishers (Lynch et al., 2010). There are no commercial shark fisheries operating in the Ningaloo area of Western Australia, but elasmobranchs are still caught as bycatch in other commercial fisheries, and are targeted by recreational fishers (CALM and MPRA, 2005). It is thus not surprising that the *de facto* Wandoo MPA is supporting higher numbers of threatened elasmobranchs than areas that remain open to fishing despite their MPA status.

Active oil fields in NW Australia, like Wandoo, may be important refuges from fishing pressure for threatened elasmobranchs. Apart from limited recreational fishing from some platforms, all other fishing activity is excluded within the 500 m petroleum safety zone around platforms, and vessels are advised to avoid the larger 2.5 nautical mile to 5 nautical mile cautionary zones (Commonwealth of Australia, 2010). There are around

60 platforms in this region, effectively creating a network of highly protected *de facto* MPAs. These *de facto* MPAs meet four of the five criteria for effective MPAs (Edgar et al., 2014): they are no-take, or only permit a small amount of recreational fishing; they are well enforced due to the prescribed petroleum safety zones (Commonwealth of Australia, 2010); they are old, as almost half of the offshore platforms in NW Australia have been in place for more than 20 years (Geoscience Australia, 2009); and they are isolated, with most platforms being at least 50 km from nearest port (Geoscience Australia, 2009). Pipelines are not protected by petroleum safety zones, but represent physical obstacles to seabed trawling (de Groot, 1982). Threatened elasmobranchs have been found associated with both platforms and pipelines in NW Australia, including silvertip sharks, whale sharks, grey nurse sharks, and oceanic manta rays *Mobula birostris* (Bond et al., 2018; McLean et al., 2018, 2019). Spillover from these *de facto* MPAs may also drive increased abundance of threatened elasmobranchs in surrounding natural habitats.

Australian legislation currently stipulates that when offshore platforms reach the end of their productive lives, they must be completely removed from the marine environment. However, this legislation is currently under review to potentially allow for *in situ* decommissioning options (Offshore Resources Branch, 2018; Taylor, 2020). Many of the offshore platforms in the NW region have been in place for decades, and removing them would result in the loss of complex novel ecosystems (van Elden et al. *in prep*.; Pradella et al. 2014). Furthermore, the *de facto* MPAs that exclude fishing around platforms would be lost. The importance of these platforms to threatened elasmobranchs should be a consideration in future decommissioning decision-making, as well as decisions around allowing fishing at decommissioned platforms. Continuing to exclude fishing at offshore platforms post-decommissioning would maintain a network of highly protected *de facto* MPAs across NW Australia that would also contribute to Australia's international commitments (Convention on Biological Diversity, 2020).

The Wandoo field is a refuge for elasmobranchs vulnerable to seabed trawling activity and creates spillover of these species into natural habitats. Offshore platforms function as artificial reefs as well as FADs (Franks, 2000; Shinn, 1974), and Wandoo could therefore provide enhanced foraging opportunity for threatened elasmobranchs in the region. Parts of NW Australia are considered hotspots for endangered elasmobranchs, and the region has globally significant populations of threatened species (Anon., 2018; Morgan et al., 2011). However, protection measures for elasmobranchs in the region tend to focus only on those very few species protected under the EPBC Act, specifically grey nurse sharks, whale sharks, green sawfish, and white sharks *Carcharodon carcharias* (Commonwealth of Australia, 2012). Offshore platforms may collectively augment populations of threatened elasmobranchs by creating a network of *de facto* MPAs, providing food, habitat, and refuge for these species.

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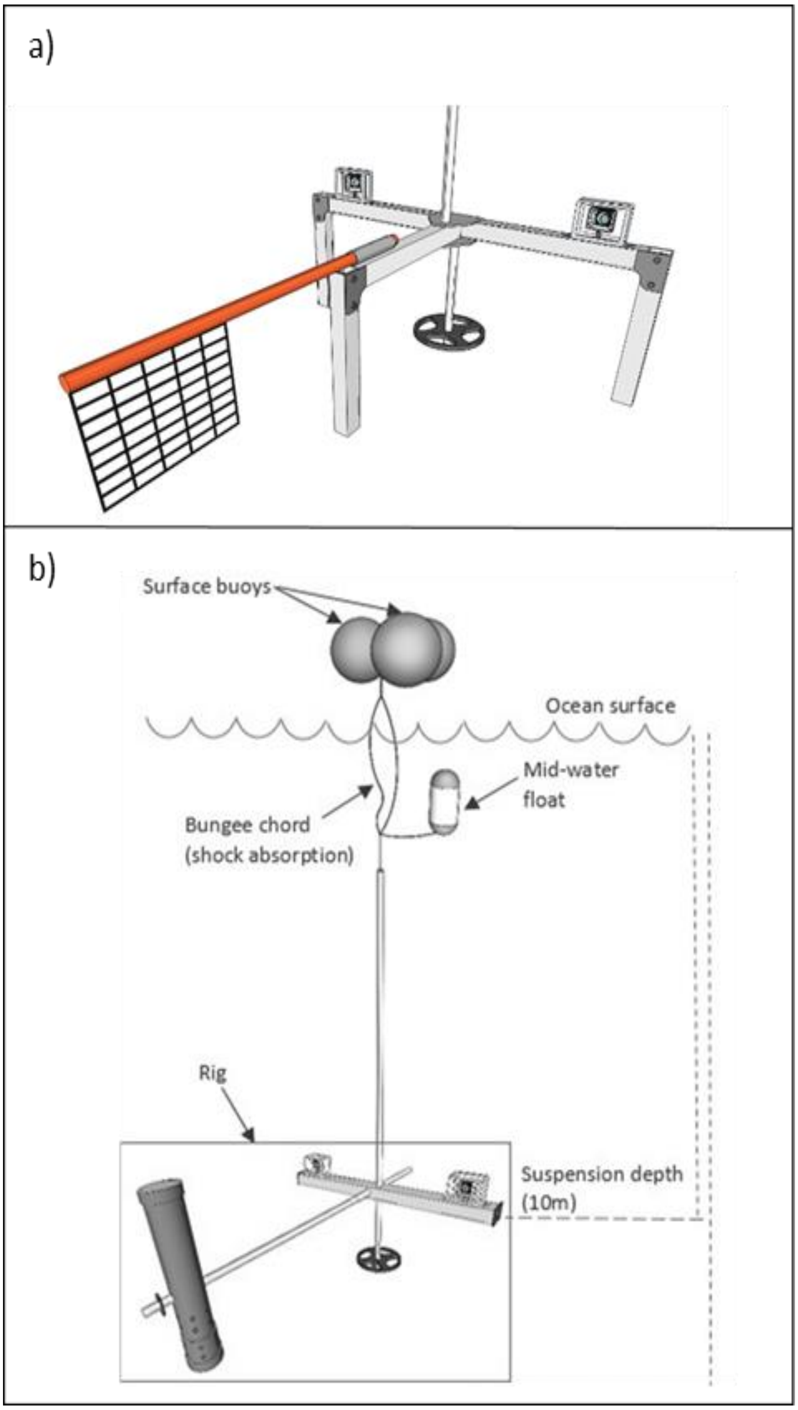
5.7 SUPPLEMENTARY INFORMATION



Supplementary Figure 5.1 Schematics of (a) seabed and (b) mid-water stereo-

Supplementary Table 5.1 Demersal regions and locations, including average coordinates for each location (decimal degrees), years in which the locations were surveyed, and number of surveys per location. Bolded text indicates the regions, with the locations listed below each region.

Location	Latitude (°S)	Longitude (°E)	Survey Years	No. of Surveys
Northeast				
East Cape York - North	11.5	143	2017, 2018	3
East Cape York - Middle	12.2	143.2	2017, 2018	3
East Cape York - South	14.1	144.2	2017	1
Ribbons - North	11.3	143.7	2017,2018	2
Ribbons - Central	13.7	143.8	2017,2018	3
Ribbons - South	14.4	144.8	2017,2018	2
Torres Strait - East	10.1	143.6	2017	2
Torres Strait - West	9.9	143.3	2017	1
Cocos-Keeling Islands				
Cocos Island	12.1	96.9	2016	1
Northwest				
Adele Island	15.6	123.2	2017	1
Ashmore Reef	12.2	123	2017	1
Barrow Island	20.8	115.5	2008-2010	3
Dampier Archipelago	20.5	116.7	2008	1
Holothuria Reef	13.6	126	2017	1
Long Reef	13.9	125.7	2017	1
Rowley Shoals	17.2	119.5	2017	1
Wandoo				
Wandoo Platform	20.1	116.4	2017-2019	6
Wandoo Sand	20.1	116.6	2018, 2019	3
Wandoo Reef	20.1	116.2	2017-2019	4
Central North				
Ningaloo Reef - North	22.1	113.8	2006, 2007, 2009	3
Ningaloo Reef - Middle	22.7	113.6	2006, 2009	2
Ningaloo Reef - South	23.8	113.3	2009	1
Central South				
Shark Bay - Dirk Hartog Island	26	113.1	2017, 2018	2
Shark Bay - Gulf	25.5	113.2	2009	1
Shark Bay - South Passage	26.1	113.2	2018	1
Shark Bay - Steep Point	26.3	113.3	2017, 2018	2

Supplementary Table 5.2 Pelagic regions and locations, including average coordinates for each location, years in which the locations were surveyed, and number of surveys per location. Bolded text indicates the regions, with the locations listed below each region.

Location	Latitude (°S)	Longitude (°E)	Survey Years	No. of Surveys
Northeast				
Great Barrier Reef	-11.2	143.2	2017	2
Cocos-Keeling Islands				
Cocos Island	-12.1	96.8	2016	1
Northwest				
Ashmore Reef - North	-12.2	123.1	2017	1
Ashmore Reef - South	-12.2	123.1	2018	1
Long Reef - East	-13.9	125.9	2017; 2018	2
Long Reef - West	-13.8	125.6	2017; 2018	2
Montebello Islands	-20.3	115.4	2018	1
Montebellos Islands - Offshore	-19.9	115.4	2018	1
Muiron Islands	-21.6	114.2	2018; 2019	2
Rowley Shoals	-17.1	119.4	2017	1
Rowleys Shoals - Offshore	-15.4	118.5	2017; 2018	2
Wandoo				
Wandoo Platform	-20.1	116.4	2017; 2018; 2019	6
Wandoo Sand	-20.1	116.6	2018; 2019	3
Wandoo Reef	-20.1	116.2	2017; 2018 2019	4
Central North				
Ningaloo Reef - Offshore	-21.8	113.5	2016	1
Ningaloo Reef	-21.9	113.8	2016; 2018; 2019	3
Central South				
Shark Bay - Dirk Hartog Island	-26.0	113	2017; 2018; 2019	3
Shark Bay - Gulf	-26.1	113.2	2012	1
Shark Bay - Steep Point	-26.3	113.1	2012; 2017; 2018; 2019	4

Supplementary Table 5.3 Environmental and anthropogenic variables for locations sampled with seabed stereo-BRUVS. Variables include: distance to port (DistPort); distance to coast (DistCoast); dissolved oxygen (O_2); salinity (Sal); sea surface temperature (SST); and depth. Bolded text indicates the regions, with the locations listed below each region. Distance to market and population were computed using the LandScan 2016 database (Dobson et al. 2000), while distances to marine features were computed using bathymetry data (Yesson et al. 2020). Environmental data were derived from: Geoscience Australia (GA) 250 m bathymetry (Whiteway 2009); GA Australian submarine canyons (Huang et al. 2014); CSIRO Atlas of Regional Seas (CARS) (Ridgway et al. 2002); and Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS) (IMOS 2020)

Location	distPort (km)	distCoast (km)	O_2 (mmol/L)	Sal (psu)	SST (°C)	Depth (m)
Northeast						
East Cape York - North	35	1.6	4.2	35	27.3	7.8
East Cape York - Middle	72.5	5.6	4.2	35.7	28	8.7
East Cape York - South	93.9	1.9	4.1	35.1	27.7	9.1
Ribbons - North	61.9	7.5	4.2	35	27.6	8
Ribbons - Central	88.5	53.7	4.2	35.1	27.1	9.7
Ribbons - South	71.3	5.2	4.3	35.1	27.5	9.4
Torres Strait - East	89.3	26.2	4.2	35.1	27.5	10.3
Torres Strait - West	77	1.1	4.2	35.1	26.4	11.5
Cocos-Keeling Islands						
Cocos Island	11.1	1.9	4.4	34.4	27.7	5.3
Northwest						
Adele Island	45.3	6	4.3	34.8	26.8	6.4
Ashmore Reef	242	321.4	4.4	34.5	27.1	10
Barrow Island	52.6	6.5	4.5	35.2	27.7	9.4
Dampier Archipelago	12.3	3.4	4.4	35.4	21.9	15.7
Holothuria Reef	242.7	25.3	4.3	34.8	26.2	3.1
Long Reef	220.3	13.2	4.3	34.8	25.9	9.4
Rowley Shoals	169.3	264	4.5	34.7	28.5	9
Wandoo						
Wandoo Platform	36	40.2	4.5	35.2	26.3	52
Wandoo Sand	36.2	38.4	4.5	35.2	27.6	54.8

Wandoo Reef	42.4	50.8	4.5	35.2	26.5	51.8
Central North						
Ningaloo Reef - North	60.1	9.9	4.7	35.1	26.9	64.8
Ningaloo Reef - Middle	28.1	2.9	4.6	35	25.1	20.5
Ningaloo Reef - South	27.6	22.4	4.6	35.3	26.4	67.6
Central South						
Shark Bay - Dirk Hartog Island	34.4	0.4	4.8	35.3	21	18.6
Shark Bay - Gulf	5.7	15.3	4.8	35.3	19.7	4.9
Shark Bay - South Passage	12.6	0.8	4.8	35.3	20.6	4.9
Shark Bay - Steep Point	18.2	0.6	4.7	35.5	21	26.3

Supplementary Table 5.4 Environmental and anthropogenic variables for locations sampled with midwater stereo-BRUVS. Variables include: linear distance to cities (LinDistCities); travel time to market (TravelTime_market); time to nearest population (TravelTime_pop); linear distance to nearest population (LinDistPop); distance to port (DistPort); distance to seamounts (DistSeamounts); distance to coral reef (DistCoralReef); depth; slope; distance to coast (DistCoast); chlorophyll concentration (Chl); and sea surface temperature (SST). Bolded text indicates the regions, with the locations listed below each region. Distance to market and population were computed using the LandScan 2016 database (Dobson et al. 2000), while distances to marine features were computed using bathymetry data (Yesson et al. 2020). Environmental data were derived from: Geoscience Australia (GA) 250 m bathymetry (Whiteway 2009); GA Australian submarine canyons (Huang et al. 2014); CSIRO Atlas of Regional Seas (CARS) (Ridgway et al. 2002); and Australia's Integrated Marine Observing System (IMOS) Moderate Resolution Imaging Spectroradiometer (MODIS) (IMOS 2020)

Location	**LinDistcities (km)**	**TravelTime_market (mins)**	**TravelTime_pop (mins)**	**LinDistpop (km)**	**distPort (km)**
Northeast					
Great Barrier Reef	267	812	72	22.9	144
Cocos-Keeling Islands					
Cocos Island	10	51	17	5.2	11.1
Northwest					
Ashmore Reef - North	260	821	31	9.8	185
Ashmore Reef - South	261	826	33	10.3	184
Long Reef - East	509	1,526	33	10.4	280
Long Reef - West	498	817	57	18.2	249
Montebello Islands	1,339	4,017	101	32.3	30.8

Montebellos Islands - Offshore	1,369	4,107	220	69.3	29.9
Muiron Islands	1,190	3,569	36	11.7	15.3
Rowley Shoals	948	2,845	83	26.3	296
Rowleys Shoals - Offshore	812	2,436	778	241	470
Wandoo					
Wandoo Platform	1,358	4,073	134.4	41.5	36
Wandoo Sand	1,366	4,099	125.6	39	36.2
Wandoo Reef	1,355	4,064	154	49.4	42.4
Central North					
Ningaloo Reef - Offshore	1,181	3,543	165.6	52.2	70.9
Ningaloo Reef	1,163	3,490	47.4	15.2	36.1
Central South					
Shark Bay - Dirk Hartog Island	727	2,181	31.4	10	34.4
Shark Bay - Gulf	701.3	2,104	1.9	0.8	5.7
Shark Bay - Steep Point	686.7	2,060	30.9	9.9	18.2

Supplementary Table 5.4 (Cont.)

Location	distSeamounts (km)	distCoralReef (km)	Depth (m)	Slope	distCoast (km)	Chl (mg/m3)	SST (°C)
Northeast							
Great Barrier Reef	442.5	2.8	37	89.7	21	0.4	27.5
Cocos-Keeling Islands							
Cocos Island	37.5	2.9	1,149	90	3.8	0.2	27.8
Northwest							
Ashmore Reef - North	99.6	6.2	208.2	90	10.4	0.2	29.4
Ashmore Reef - South	99.8	6.6	222.2	90	10.9	0.2	29.4
Long Reef - East	408.7	7.3	42	89.7	11	0.6	29.2
Long Reef - West	379	9.4	48.6	89.4	18.7	0.4	29.4
Montebello Islands	445	31.7	68.6	89.6	32.9	0.3	26.9
Montebellos Islands - Offshore	405.2	69.2	380.7	89.6	69.8	0.2	27.3
Muiron Islands	373.2	9.8	86.3	89.9	12	0.4	26.1

Rowley Shoals	269.5	11.9	434.1	89.9	259.6	0.1	28.6
Rowleys Shoals - Offshore	87	221.5	4,840	90	339.8	0.1	29
Wandoo							
Wandoo Platform	468.6	41.4	50	89.5	41.5	0.3	27.2
Wandoo Sand	466.5	45.7	53.3	83.6	39.2	0.3	27.2
Wandoo Reef	466.1	50.8	51.3	88.5	49.8	0.3	27.1
Central North							
Ningaloo Reef - Offshore	304	50.6	1,464	90	52.9	0.2	26.1
Ningaloo Reef	340.3	13.2	499.1	90	15.8	0.3	26
Central South							
Shark Bay - Dirk Hartog Island	470.7	8.1	91.9	89.7	10.6	0.3	23.4
Shark Bay - Gulf	496.5	9.3	11.5	89.8	1	1	23.1
Shark Bay - Steep Point	479.8	25.3	99.4	89.7	10.5	0.4	23.3

Supplementary Table 5.5 List of threatened elasmobranchs recorded on seabed stereo-BRUVS by location, including their IUCN Red List classifications (IUCN): Vulnerable (VU); Endangered (EN) and Critically Endangered (CR). The regions are Northeast (NE), Cocos-Keeling Islands (CI), Northwest (NW), Wandoo (WN), Central North (CN) and Central South (CS). Bolded text indicates the regions, with the locations listed below each region.

			NE							**CI**
Family	Binomial	IUCN	East Cape York - North	East Cape York - Middle	Ribbons - North	Ribbons - Central	Ribbons - South	Torres Strait - East	Torres Strait - West	Cocos Island
Aetobatidae	*Aetobatus ocellatus*	VU			**X**					
Carcharhinidae	Carcharhinidae sp.	VU	**X**	**X**		**X**		**X**		
Carcharhinidae	*Carcharhinus albimarginatus*	VU								
Carcharhinidae	*Carcharhinus falciformis*	VU								
Carcharhinidae	*Carcharhinus obscurus*	EN								
Carcharhinidae	*Carcharhinus plumbeus*	VU				**X**		**X**		
Carcharhinidae	*Carcharhinus* sp.	VU	**X**	**X**		**X**		**X**		
Carcharhinidae	*Negaprion acutidens*	VU		**X**	**X**	**X**		**X**	**X**	
Dasyatidae	*Himantura* sp.	VU	**X**							
Dasyatidae	*Himantura uarnak*	VU								

Dasyatidae	*Himantura undulata*	VU									
Dasyatidae	*Pateobatis fai*	VU									
Dasyatidae	*Pateobatis jenkinsii*	VU									
Dasyatidae	*Taeniurops meyeni*	VU			**X**		**X**			**X**	
Dasyatidae	*Urogymnus granulatus*	VU									
Ginglymostomatidae	*Nebrius ferrugineus*	VU		**X**	**X**	**X**	**X**	**X**	**X**		
Glaucostegidae	*Glaucostegus typus*	CR		**X**	**X**		**X**				
Hemigaleidae	*Hemipristis elongata*	VU		**X**	**X**		**X**				
Mobulidae	*Mobula alfredi*	VU									**X**
Mobulidae	*Mobula thurstoni*	EN									
Myliobatidae	*Mobula birostris*	VU									
Odontaspididae	*Carcharias taurus*	VU									
Pristidae	*Pristis clavata*	EN									
Rhinidae	*Rhina ancylostoma*	CR									
Rhinidae	*Rhynchobatus* sp.	CR		**X**	**X**		**X**		**X**		
Sphyrnidae	*Sphyrna lewini*	CR			**X**						
Sphyrnidae	*Sphyrna mokarran*	CR	**X**				**X**	**X**			
Stegostomatidae	*Stegostoma tigrinum*	EN				**X**	**X**	**X**		**X**	

Supplementary Table 5.5 (Cont.)

Family	Binomial	IUCN	**NW** Ashmore Reef	Barrow Island	Dampier Archipelago	Holothuria Reef	Long Reef	Rowley Shoals	**WN** Wandoo Platform	Wandoo Sand	Wandoo Reef
Aetobatidae	*Aetobatus ocellatus*	VU					**X**				**X**
Carcharhinidae	Carcharhinidae sp.	VU							**X**		
Carcharhinidae	*Carcharhinus albimarginatus*	VU									
Carcharhinidae	*Carcharhinus falciformis*	VU		**X**						**X**	
Carcharhinidae	*Carcharhinus obscurus*	EN								**X**	
Carcharhinidae	*Carcharhinus plumbeus*	VU		**X**	**X**				**X**	**X**	**X**
Carcharhinidae	*Carcharhinus* sp.	VU	**X**	**X**			**X**		**X**	**X**	**X**
Carcharhinidae	*Negaprion acutidens*	VU	**X**	**X**	**X**		**X**				

Dasyatidae	*Himantura* sp.	VU									
Dasyatidae	*Himantura uarnak*	VU		X	X						
Dasyatidae	*Himantura undulata*	VU		X							
Dasyatidae	*Pateobatis fai*	VU		X							
Dasyatidae	*Pateobatis jenkinsii*	VU		X	X						
Dasyatidae	*Taeniurops meyeni*	VU			X				X		
Dasyatidae	*Urogymnus granulatus*	VU		X							
Ginglymostomatidae	*Nebrius ferrugineus*	VU	X	X	X	X	X	X	X		
Glaucostegidae	*Glaucostegus typus*	CR		X					X		
Hemigaleidae	*Hemipristis elongata*	VU	X	X							
Mobulidae	*Mobula alfredi*	VU	X					X			
Mobulidae	*Mobula thurstoni*	EN									
Myliobatidae	*Mobula birostris*	VU		X							
Odontaspididae	*Carcharias taurus*	VU									
Pristidae	*Pristis clavata*	EN		X							
Rhinidae	*Rhina ancylostoma*	CR		X	X					X	
Rhinidae	*Rhynchobatus* sp.	CR	X	X	X	X	X		X	X	X
Sphyrnidae	*Sphyrna lewini*	CR		X	X						
Sphyrnidae	*Sphyrna mokarran*	CR		X	X	X	X		X	X	X
Stegostomatidae	*Stegostoma tigrinum*	EN	X	X	X		X		X	X	X

Supplementary Table 5.5 (Cont.)

			CN			**CS**			
Family	Binomial	IUCN	Ningaloo - North	Ningaloo - Middle	Ningaloo - South	Shark Bay - Dirk Hartog Island	Shark Bay - Gulf	Shark Bay - South Passage	Shark Bay - Steep Point
Aetobatidae	*Aetobatus ocellatus*	VU							
Carcharhinidae	Carcharhinidae sp.	VU							X

Carcharhinidae	*Carcharhinus albimarginatus*	VU			X					
Carcharhinidae	*Carcharhinus falciformis*	VU								
Carcharhinidae	*Carcharhinus obscurus*	EN						X		
Carcharhinidae	*Carcharhinus plumbeus*	VU				X	X			
Carcharhinidae	*Carcharhinus* sp.	VU			X	X	X		X	X
Carcharhinidae	*Negaprion acutidens*	VU	X					X		
Dasyatidae	*Himantura* sp.	VU								
Dasyatidae	*Himantura uarnak*	VU								
Dasyatidae	*Himantura undulata*	VU	X							
Dasyatidae	*Pateobatis fai*	VU								
Dasyatidae	*Pateobatis jenkinsii*	VU								
Dasyatidae	*Taeniurops meyeni*	VU		X	X					
Dasyatidae	*Urogymnus granulatus*	VU								
Ginglymostomatidae	*Nebrius ferrugineus*	VU		X	X	X				
Glaucostegidae	*Glaucostegus typus*	CR	X					X		
Hemigaleidae	*Hemipristis elongata*	VU	X							
Mobulidae	*Mobula alfredi*	VU					X			
Mobulidae	*Mobula thurstoni*	EN								X
Myliobatidae	*Mobula birostris*	VU								
Odontaspididae	*Carcharias taurus*	VU	X							
Pristidae	*Pristis clavata*	EN								
Rhinidae	*Rhina ancylostoma*	CR								
Rhinidae	*Rhynchobatus* sp.	CR	X			X		X	X	X
Sphyrnidae	*Sphyrna lewini*	CR						X		
Sphyrnidae	*Sphyrna mokarran*	CR		X	X	X				
Stegostomatidae	*Stegostoma tigrinum*	EN				X				

Supplementary Table 5.6 List of threatened elasmobranchs recorded on midwater stereo-BRUVS by location, including their IUCN Red List classification (IUCN)s: Vulnerable (VU); Endangered (EN) and Critically Endangered (CR). The regions are Northeast (NE), Cocos (Keeling) Islands (CI), Northwest (NW), Wandoo (WN), Central North (CN) and Central South (CS). Bolded text indicates the regions, with the locations listed below each region.

			NE	**CI**	**NW**				
Family	Binomial	IUCN	Great Barrier Reef	Cocos Island	Ashmore Reef - North	Ashmore Reef -South	Long Reef - East	Long Reef - West	Montebello Islands
Carcharhinidae	Carcharhinidae sp.	VU	**X**		**X**			**X**	
Carcharhinidae	*Carcharhinus albimarginatus*	VU			**X**	**X**			**X**
Carcharhinidae	*Carcharhinus falciformis*	VU		**X**					**X**
Carcharhinidae	*Carcharhinus longimanus*	CR							
Carcharhinidae	*Carcharhinus obscurus*	EN	**X**	**X**					**X**
Carcharhinidae	*Carcharhinus plumbeus*	VU	**X**			**X**			**X**
Carcharhinidae	*Carcharhinus* sp.	VU	**X**		**X**		**X**	**X**	**X**
Lamnidae	*Isurus oxyrinchus*	EN							
Myliobatidae	*Mobula birostris*	VU		**X**					
Myliobatidae	*Mobula* sp.	VU	**X**	**X**					
Carcharhinidae	*Negaprion acutidens*	VU							**X**
Rhincodontidae	*Rhincodon typus*	EN				**X**			
Sphyrnidae	*Sphyrna lewini*	CR		**X**		**X**			
Sphyrnidae	*Sphyrna mokarran*	CR	**X**			**X**	**X**	**X**	**X**

Supplementary Table 5.6 (Cont.)

			NW			WN			CN	
Family	Binomial	IUCN	Montebello Islands - Offshore	Muiron Islands	Rowley Shoals	Wandoo Platform	Wandoo Sand	Wandoo Reef	Ningaloo Reef - Offshore	Ningaloo Reef
Carcharhinidae	Carcharhinidae sp.	VU				**X**				
Carcharhinidae	*Carcharhinus albimarginatus*	VU	**X**	**X**						
Carcharhinidae	*Carcharhinus falciformis*	VU	**X**	**X**	**X**	**X**	**X**	**X**		**X**
Carcharhinidae	*Carcharhinus longimanus*	CR	**X**						**X**	
Carcharhinidae	*Carcharhinus obscurus*	EN	**X**	**X**		**X**	**X**	**X**		**X**
Carcharhinidae	*Carcharhinus plumbeus*	VU	**X**	**X**		**X**	**X**	**X**		
Carcharhinidae	*Carcharhinus* sp.	VU	**X**	**X**		**X**	**X**	**X**		**X**
Lamnidae	*Isurus oxyrinchus*	EN	**X**						**X**	**X**
Myliobatidae	*Mobula birostris*	VU								
Myliobatidae	*Mobula* sp.	VU				**X**	**X**	**X**		
Carcharhinidae	*Negaprion acutidens*	VU								
Rhincodontidae	*Rhincodon typus*	EN								
Sphyrnidae	*Sphyrna lewini*	CR								
Sphyrnidae	*Sphyrna mokarran*	CR	**X**			**X**	**X**	**X**		**X**

Supplementary Table 5.6 (Cont.)

Family	Binomial	IUCN	**CS** Shark Bay - Dirk Hartog Island	Shark Bay - Gulf	Shark Bay - Steep Point
Carcharhinidae	Carcharhinidae sp.	VU		**X**	**X**
Carcharhinidae	*Carcharhinus albimarginatus*	VU			
Carcharhinidae	*Carcharhinus falciformis*	VU			**X**
Carcharhinidae	*Carcharhinus longimanus*	CR			
Carcharhinidae	*Carcharhinus obscurus*	EN	**X**		**X**
Carcharhinidae	*Carcharhinus plumbeus*	VU	**X**		**X**
Carcharhinidae	*Carcharhinus* sp.	VU	**X**		**X**
Lamnidae	*Isurus oxyrinchus*	EN			**X**
Myliobatidae	*Mobula birostris*	VU			
Myliobatidae	*Mobula* sp.	VU			
Carcharhinidae	*Negaprion acutidens*	VU			
Rhincodontidae	*Rhincodon typus*	EN			
Sphyrnidae	*Sphyrna lewini*	CR			**X**
Sphyrnidae	*Sphyrna mokarran*	CR			**X**

CHAPTER 6 GENERAL DISCUSSION

The world's oceans are being industrialised at an unprecedented rate in what is being called a marine industrial revolution (Salcido, 2008; Wright, 2015). The major sectors of ocean industrialisation are mineral and energy resources, transport and communication, leisure, and coastal engineering (Smith, 2000). Increases in global energy consumption, along with advances in technology, have driven offshore energy exploration and production on a global scale. The first offshore oil and gas platform (offshore platform) was installed in 1947, and the industry has subsequently grown rapidly to an estimated 12,000 offshore platforms in 2017 (Aagard and Besse, 1973; Ars and Rios, 2017).

Offshore platforms around the world create diverse and productive marine ecosystems. These platforms may play important ecological roles, including increasing hard substrate on regional scales, providing habitat for juvenile fish species, and functioning as *de facto* marine protected areas (MPAs) (Friedlander et al., 2014; Love et al., 2006; Schroeder and Love, 2004). The novel ecosystem concept (Hobbs et al., 2013b) has only recently been used to describe the ecosystems that emerge around offshore platforms, and contributes to decision-making around the decommissioning process.

Australia's tropical marine regions are vast and diverse (Lough, 2008). Many of these regions are impacted by increasing ocean industrialisation despite the implementation of networks of multiple-use MPAs (Parks Australia 2020). Australia's Northwest Shelf (NWS) is a marine biodiversity hotspot that is also rich oil and gas reserves. The offshore infrastructure in this region includes over 60 platforms and thousands of kilometres of pipeline, which are inhabited by endangered megafauna and commercially important fish species (Bond et al., 2018; McLean et al., 2019; Pradella et al., 2014). The offshore platforms on the NWS are potentially regionally important ecosystems given the area is generally characterised by sandy habitats with little hard substrate and low habitat complexity. Industry-funded independent research provides insight not only into the marine communities associated with these platforms, but also into the ecology of this largely understudied biodiversity hotspot.

This dissertation has two emergent themes:

Offshore platforms as novel ecosystems. The novel ecosystem concept can be applied to offshore platforms on a case-by-case basis, and more generally, can be integrated into existing decommissioning analysis frameworks. The Wandoo platform was treated as a first case study for the novel ecosystem concept and the associated ecosystem was found to be significantly altered from what prevailed historically. The marine communities at Wandoo are distinct from those found at natural habitats, with diverse, reef-associated demersal fishes characterising the platform-associated community. The Wandoo field also functions as a *de facto* MPA allowing macrobenthos communities to recover from historical trawling, and provides a refuge for threatened elasmobranchs.

The use of stereo-BRUVS to study offshore platform-associated communities. Stereo-BRUVS have only been used in a handful of studies on the ecology of offshore platform to date. Rare and highly mobile species, as well as novel behaviours, are reported from the stereo-BRUVS deployed around the Wandoo field. Stereo-BRUVS can effectively sample the ecological halo created by offshore platforms, and allow platform-associated communities to be compared with other habitats sampled with BRUVS on regional, national and international scales. The combined use of both seabed and mid-water stereo-BRUVS allows for effective sampling of offshore platform habitats that extend from the seafloor to the surface. The ability to cost-effectively and safely obtain large quantities of video data, and without the influence of human presence, means that stereo-BRUVS are an effective tool for recording ecological data around offshore platforms.

6.1 Synergy amongst chapters

This dissertation provides insight into how the installation of offshore platforms can result in the emergence of novel ecosystems. There is synergy among these chapters on several themes, with all four data chapters providing insight into the novel marine communities found around offshore platforms. In chapters 2 and 3, I explicitly evaluate offshore platforms as novel ecosystems. Chapters 3 to 5 describe the ecology of the Wandoo oil field in both a local and regional context, and provide evidence for the advantages of using stereo-BRUVS to study offshore platform-associated communities.

Offshore platforms as novel ecosystems

In chapters 2 and 3 of this dissertation, I tested the application of the novel ecosystem concept to offshore platforms using a combination of a literature review (Chapter 2) and a field-based case study (Chapter 3). At face value, offshore platforms appear to be ideal candidates for classification as novel ecosystems, however prudence is necessary when combining these two contentious subjects. Offshore platforms and Rigs-to-Reefs (RTR) have faced significant public criticism, particularly in the cases of the Brent Spar (Löfstedt and Renn, 1997) and public opposition in California (Schroeder and Love, 2004). In contrast, criticisms of the novel ecosystem concept have predominantly come from the scientific community (Murcia et al., 2014; Simberloff et al., 2015). Classifying offshore platforms as novel ecosystems could potentially be viewed as simply an excuse for energy companies to dump unwanted platforms at sea, minimising costs associated with their end-of-life decommissioning. It is therefore crucial that the application of the novel ecosystem concept to offshore platforms is backed by solid scientific evidence.

In Chapter 2 (van Elden et al., 2019), I demonstrate that the novel ecosystem concept can be applied to offshore platforms and can be incorporated into existing decommissioning frameworks. I developed three criteria for applying the novel ecosystem concept to offshore platforms, based on its most recent definition (Hobbs et al., 2013a). These criteria cover various aspects of offshore platform ecology and decommissioning, including ecosystem alteration, the lack of human management of the ecosystems, and considerations preventing the ecosystem from being restored with respect to ecological, environmental and social factors. The criteria are often context-specific, and should therefore be applied to platforms on a case-by-case basis. However, existing decommissioning decision analysis frameworks can be adapted to incorporate the novel ecosystem criteria, alongside typical decommissioning considerations such as water quality, social opposition to the platform, marine communities, and financial cost (Fowler et al., 2014; Henrion et al., 2015) for a more generalised approach.

In Chapter 3, I applied the criteria developed in Chapter 2 to the Wandoo oil field on Australia’s NWS. I found that the Wandoo field has been ecologically altered by the presence of infrastructure, and that the self-organising ecosystem at Wandoo has novel qualities that would not have been present historically. The study of the marine communities in the Wandoo field assessed demersal and pelagic communities as well as habitat composition, all of which are impacted by the presence of offshore infrastructure. A critical component of this case study was the identification of a site which resembled the reported historical state of the Wandoo field. Studying this site allowed me to infer what the marine communities would have looked like at Wandoo prior to the installation of infrastructure, and determine how these communities have changed over time. In the absence of comprehensive ecological studies pre-installation, historical proxies represent a method for assessing offshore platforms against Criterion 1: *The abiotic, biotic and social components of the system "differ from those that prevailed historically"* (Hobbs et al., 2013a). The assessment presented in Chapter 3 found that a novel ecosystem has emerged in the Wandoo field. Classifying Wandoo as a novel ecosystem provides a mechanism for recognising the various ecological roles played by the infrastructure in the Wandoo field, all of which should be considered in the decommissioning assessment process. This case study can be used as a template for applying the novel ecosystem criteria to other offshore platforms.

Ecology of the Wandoo field

The three year ecological study into the marine communities in the Wandoo field forms the basis of this dissertation. The outcomes of this study are presented in chapters 3 to 5, and provide insight into a diverse and important novel ecosystem which influences surrounding natural habitats. Chapter 2, in describing the ecological traits of offshore platforms around the world, provides context for the assessment of the Wandoo field as a novel ecosystem.

In Chapter 3, I found that the marine community at Wandoo differs from those found at adjacent natural habitats. The seabed habitat at Wandoo was dominated by macrobenthos communities, whereas the two natural sites were dominated by bare sand habitats. The exclusion of seabed trawling around Wandoo has protected the macrobenthos communities, and allowed them to recover after decades of destructive seabed trawling activity in this region (Sainsbury et al., 1993). Both demersal and pelagic communities at Wandoo had shifted from their likely historical state: the demersal community was more diverse than the natural habitats and was characterised by reef-associated species not seen at the sandy site. Whilst the pelagic communities were more similar across the three sites than the demersal communities, the Wandoo pelagic community was characterised by species that are strongly 'platform-associated', namely rainbow runner *Elagatis bipinnulata* and great barracuda *Sphyraena barracuda* (Friedlander et al., 2014; McLean et al., 2019).

In Chapter 4 (van Elden and Meeuwig, 2020), I report the first wild record of dynamic decapod mimicry by a cuttlefish. The cuttlefish, tentatively identified as Smith's cuttlefish *Sepia smithi*, was observed approaching the bait bag while employing crustacean-like aggressive mimicry. This is the first wild observation of crustacean-like aggressive mimicry by a cuttlefish, and provides further evidence of the usefulness of stereo-BRUVS for studying animal behaviour. Stereo-BRUVS allow for remote sampling without human influence and have recorded a range of novel animal behaviours (Barley et al., 2016; Birt et al., 2019).

In Chapter 5, I found that the abundance of threatened elasmobranchs in the Wandoo field and adjacent natural habitats was higher than that in most of Australia's tropical regions, including locations in the Ningaloo Reef and Great Barrier Reef multiple-use MPAs. Several taxa were also found in higher abundance around the Wandoo field than in other regions, including silky sharks *Carcharhinus falciformis*, wedgefishes *Rhynchobatus* sp., and leopard sharks *Stegostoma tigrinum*. The demersal threatened elasmobranch assemblages differed between Wandoo and the adjacent natural sites, likely due to the presence of the Wandoo infrastructure and associated abiotic and biotic changes to the environment. The Wandoo field is a *de facto* MPA, excluding the seabed trawl fishery operating in the region. This *de facto* MPA not only provides refuge for threatened elasmobranchs, but is also likely to increase their abundance in adjacent natural habitats through spillover as is generally the case for MPAs (Halpern et al., 2009; Roberts et al., 2001).

Wandoo has several ecological traits that are characteristic of offshore platforms around the world. Wandoo functions as an artificial reef dominated by reef-associated species, and is also an important habitat for commercially important fish species and threatened elasmobranchs, as has been reported from other infrastructure on the NWS and elsewhere (Bond et al., 2018; Love et al., 2006; McLean et al., 2019; Pradella et al., 2014; Robinson et al., 2013). The cuttlefish mimicry recorded at Wandoo

(Chapter 4) adds to the literature on novel behavioural records near offshore infrastructure (Bond et al., 2020a; Haugen and Papastamatiou, 2019; Robinson et al., 2013). Many offshore platforms are located in remote, understudied regions such as Australia's NWS. The collection of novel behavioural records at offshore platforms likely reflects the lack of research into the remote regions where these platforms are located. Increasing ecological research around offshore platforms may reveal more novel records and behaviours, and increase our knowledge of remote offshore ecosystems.

The use of stereo-BRUVS to study offshore platform-associated communities

Chapters 3 to 5 of this dissertation demonstrate the usefulness of stereo-BRUVS for studying offshore platform-associated communities. Stereo-BRUVS have only been used in a handful of ecological studies on offshore infrastructure to date (Bond et al., 2018; Reynolds et al., 2018). Stereo-BRUVS are relatively inexpensive, particularly in comparison with other commonly used sampling methods such as remotely operated vehicles (ROVs; Letessier et al. 2015b). They can also be deployed over large spatial scales, which allows for sampling of the ecological halo created by offshore platforms as well as surrounding natural habitats, as demonstrated in Chapter 3. Stereo-BRUVS can be used to obtain a significant amount of data over a short period of time, which is advantageous for sampling offshore platforms located far from shore or in areas prone to severe weather conditions. The expeditions to the Wandoo field were restricted to about six to ten days, due to extreme tide ranges and unpredictable weather conditions. Despite this restriction on sampling time, an average of 250 hours of video data were collected on each of the six expeditions, sufficient to detect spatial and temporal differences between sites.

Chapter 5 demonstrates how the use of stereo-BRUVS around Wandoo allows for comparisons with existing stereo-BRUVS data on a large scale. Comparing platform-associated communities with those found at natural habitats is an effective method for assessing the way these platforms alter regional ecology, and potentially create novel ecosystems. The relative ecological value of offshore platforms is also an important consideration in the decommissioning decision-making process (Fowler et al., 2014). There are several global stereo-BRUVS databases containing video-derived data from various habitats, and using stereo-BRUVS to assess platform-associated communities allows for like-for-like comparisons with these databases. Stereo-BRUVS studies on offshore platforms allow for comparisons with data from nearby habitats, or from similar regions around the world. The increased use of stereo-BRUVS to study offshore platform communities would also allow for comparisons between platforms, which are lacking in regions such as Australia's Northwest Shelf.

Several studies have reported elusive animals or novel behaviours observed on stereo-BRUVS imagery (Barley et al., 2016; Birt et al., 2019; Bond et al., 2018; Letessier et al., 2015a; Thompson et al., 2019). Stereo-BRUVS allow us to spend significantly more time observing marine habitats and the wildlife therein, without the influence of human presence, and increase the likelihood of observing rare animals and behaviours. Chapter 4 reports a stereo-BRUVS observation of a behaviour not previously reported outside of a laboratory setting. Stereo-BRUVS enabled me to measure the mantle length of the cuttlefish, which significantly helped in obtaining a tentative species identification. Novel behaviours have been reported from offshore infrastructure in the past, including megafauna aggregations and pufferfish nests (Bond et al., 2020a; Haugen and Papastamatiou, 2019; Robinson et al., 2013). Stereo-BRUVS deployed around offshore infrastructure are likely to observe more of these rare species and novel behaviours in future. These novel records and behaviours provide insight into understudied ecosystems and increase our understanding of complex animal behaviours and interactions.

6.2 Caveats and future directions

This dissertation demonstrates the effectiveness of stereo-BRUVS in obtaining large quantities of data over a large spatial scale, which is useful when documenting the status of communities associated with offshore platforms. The six expeditions to Wandoo and adjacent natural habitats yielded over 1,600 hours of video footage from 595 seabed and 530 mid-water stereo-BRUVS deployments. In analysing the video imagery, I counted 35,070 individual animals from 358 taxa, representing 85 families. One constraint on the data collection for this dissertation was the health and safety restrictions on sampling around infrastructure. Stereo-BRUVS had to be deployed at least 50 m away from all infrastructure to avoid possible entanglement or damage. This sampling constraint was mitigated by sampling 50 m away from the reef at the Control Reef site, which allowed for like-for-like comparisons between the sites. An unexpected positive outcome of this sampling constraint was the discovery of a large ecological halo around the Wandoo infrastructure, with elevated fish diversity and denser macrobenthos habitat. The ecological halo around Wandoo also appears to be larger than previous reports of ecological halos around offshore platforms. It would nevertheless be beneficial to obtain comparable data on the communities residing directly on the infrastructure. ROVs have previously been used to assess these communities (Tothill, 2019) and in Appendix 1 (van Elden at al. 2020) I used these ROV data, along with the BRUVS data obtained from Wandoo, to demonstrate that a combination of these two methods allows for complete sampling of an offshore platform and the surrounding ecological halo. However, with no ROV data from the two control sites in this study, I could not compare the communities at Wandoo with those found on the natural reef site. A combination of sampling methods, such as SCUBA diver surveys and ROVs, has been used to survey platform-associated communities in other regions (Ajemian et al., 2015; Bond et al., 2020b; Love et al., 1994) and future studies could involve a combination of ROVs and BRUVS surveys at both offshore platforms and natural habitats.

The novel ecosystem that has emerged in the Wandoo field has likely impacted a range of marine taxa beyond those assessed in this dissertation. The roles Wandoo plays for these taxa need to be assessed before Wandoo is decommissioned. Benthic communities attached to the infrastructure were observed in abundance on archival ROV footage from Wandoo, and should be assessed. The presence of hard substrate extending to the surface is not found in natural habitats, where the hard substrate is more than 30 m deep and there is less available light. The benthic species that have colonised the Wandoo infrastructure are therefore likely to be different from those found in natural habitats.

During field work, I observed marine megafauna in close proximity to the platform, including reef mantas *Mobula alfredi*, humpback whales *Megaptera novaeangliae* and flatback turtles *Natator depressus*. These species are frequently observed both from the platforms and from vessels operating in the Wandoo field. Wandoo may serve an important ecological function for these animals, however their presence around the infrastructure needs to be quantified. Platform-based observations have been successfully used to record megafauna around offshore platforms in the North Sea, and could be implemented at Wandoo (Todd et al., 2016). A variety of seabirds have also been observed on the Wandoo infrastructure. Offshore platforms attract seabirds through the provision of roosting sites and shelter from severe weather, as well as enhanced feeding opportunity (Tasker et al., 1986). The decommissioning of Wandoo could have significant impacts for these birds, as the offshore platforms in the area represent the only roosting sites for a considerable distance. The seabird populations could also be assessed through platform-based observations, and the existing stereo-BRUVS database can be used to determine whether the prey species of these birds are found in abundance at Wandoo.

6.3 Implications for Decommissioning

The novel ecosystem criteria developed in Chapter 2 provide a mechanism for recognising the ecological roles played by offshore platforms, and can complement current decommissioning decision analysis tools. I applied these criteria to the Wandoo field in Chapter 3, and concluded that a novel ecosystem has emerged due to the presence of the Wandoo infrastructure. I found that many of the positive novel qualities present at Wandoo would be lost under either ‘topping’ or ‘complete removal’ decommissioning scenarios. The mid-water portions are important for juvenile fishes and may act as FADs for pelagic fauna (Franks, 2000; Tothill, 2019), while the lower portions of the structures exclude seabed trawling and protect important macrobenthos habitat (Culwell, 1997).

The exclusion of fishing in the Wandoo field has created a *de facto* MPA. This *de facto* MPA not only provides refuge for threatened elasmobranchs, but is also likely to increase their abundance in adjacent natural habitats through spillover. It is likely that the Wandoo field, along with the other offshore infrastructure on the NWS, is providing important habitat and refuge for threatened elasmobranchs in this marine biodiversity hotspot. The *de facto* MPA at Wandoo has several features of highly effective MPAs, and may be more effective than many multiple-use MPAs in Australia’s tropical regions (Edgar et al., 2014). It is likely that the petroleum safety zone around the Wandoo infrastructure would cease to exist post-decommissioning, which would expose much of the Wandoo field to commercial and recreational fishing. Maintaining an exclusion zone around the infrastructure would allow Wandoo to continue functioning as a *de facto* MPA, which should be an important consideration under any decommissioning scenario.

The best ecological outcome for the decommissioning of Wandoo would involve the two platforms, Wandoo A and Wandoo B, being left standing in place. This scenario would maintain the roles Wandoo plays as an artificial reef and a FAD. The exclusion zone around the Wandoo infrastructure should be maintained in order to exclude both recreational and commercial fishing activity around the decommissioned infrastructure. This exclusion zone would ensure the protection of the ecological halo around the infrastructure, and maintain the *de facto* MPA that has been in place for decades. It is likely that this outcome would not only maintain the novel ecosystem that has emerged at Wandoo, but also enhance regional productivity through spillover from the MPA.

6.4 Conclusion

Ocean industrialisation, driven by the insatiable demands of an increasing human population, is altering marine habitats and degrading the oceans (Salcido, 2008; Smith, 2000; Wright, 2015). Offshore energy production contributes a large percentage of global energy consumption, and has involved installing offshore platforms weighing thousands of tonnes, and thousands of kilometres of pipelines in the world's oceans (OGP Decommissioning Committee, 2012; Planete Energies, 2015). Offshore platforms create ecosystems that support a wide range of marine species from corals and sponges to fishes and marine megafauna (Gass and Roberts, 2006; Love et al., 2006; McLean et al., 2017; Todd et al., 2016).

In this dissertation I have found that the installation of offshore platforms significantly alters the environment and ecology of the installation site, and creates an ecosystem with novel qualities not present pre-installation. In many cases, the ecosystem changes caused by the installation of offshore platforms result in the emergence of beneficial outcomes for marine communities. The novel ecosystem concept is a mechanism for recognising and managing these important habitats, but must be used prudently. I argue that a novel ecosystem has emerged in the Wandoo field, located in Australia's NWS marine biodiversity hotspot. The presence of the Wandoo infrastructure has significantly altered the marine communities from those which would have existed previously, and these communities are distinct from those found in comparable natural habitats. The exclusion of fishing activity around Wandoo has resulted in a *de facto* MPA, allowing for the recovery of macrobenthos communities from historical trawling impacts, increased diversity of reef-associated fishes, and acting as a refuge for threatened elasmobranchs. The use of stereo-BRUVS has provided insight into various ecological aspects of the Wandoo platform, including rare and critically endangered fauna, novel animal behaviour, and diverse demersal and pelagic communities. I demonstrate that the combination of seabed and midwater stereo-BRUVS is an effective method for sampling the demersal and pelagic communities associated with offshore platforms, which are both impacted by the presence of infrastructure extending through the water column. This dissertation characterises the ecology of the novel ecosystem that has emerged in the Wandoo field, and presents strong ecological evidence for the Wandoo infrastructure to be maintained as an artificial reef, and protected as a MPA, post-decommissioning.

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APPENDIX 1 STRATEGIES FOR OBTAINING ECOLOGICAL DATA TO ENHANCE DECOMMISSIONING ASSESSMENTS

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A1.1 ABSTRACT

Many offshore oil and gas platforms around the globe are reaching their end-of-life and will require decommissioning in the next few decades. Knowledge on the ecology of offshore platforms and their ecological role within a regional context in Australia is limited and the subsequent consequences of decommissioning remain poorly understood. Remotely operated vehicle (ROV) video is often collected during standard industry operations and may provide insight into the marine life associating with offshore platforms, however the utility of this video for scientific purposes remains unclear. We propose a standardised method of analysing this large database of archival ROV footage with specific interest in analysing the vertical distribution of fish species. Baited remote underwater video systems (BRUVS) are a widely used tool for studying marine faunal communities, and we demonstrate the value of BRUVS for understanding the regional ecology around offshore platforms. A combination of BRUVS and ROV data can be used to determine the relative ecological value of offshore platforms within a regional context. The Wandoo oil platform on Australia's North West Shelf was used as a case study to test these proposed methods by assessing demersal and pelagic fish populations both on and around the Wandoo platform and various natural habitats in the region.

A1.2 INTRODUCTION

There are over 12,000 oil and gas platforms around the world, many of which have been in place for decades (Ars & Rios 2017). Over this time, the sub-surface infrastructure of these platforms is colonised by sessile marine organisms such as algae, corals and sponges, which provide habitat and/or food for a variety of marine fauna (Forteath et al. 1982). Within about 5-6 years, offshore platforms can develop reef-type communities and by the end of their lifespans, they have effectively become complex artificial reefs (Driessen 1986; Shinn 1974). Some offshore platforms are among the most ecologically productive ecosystems globally (Claisse et al. 2014) and can become novel ecosystems, with unique species assemblages that were not present prior to the installation of the platform (van Elden et al. 2019).

Legislation in most countries states that at the end of their lifespans, offshore platforms must be completely removed from the marine environment for onshore disposal. In many cases, this means the loss of a diverse and productive marine community. Understanding the ecological role played by offshore platforms should be a key part of the decommissioning process.

Part of the challenge in understanding the potential ecological benefits of offshore platforms is the lack of data. Targeted ecological research is an expensive enterprise, however a wealth of ecological information is collected indirectly during standard industry operations, such as maintenance inspections on infrastructure and environmental surveys using remotely operated vehicles (ROV). The video footage collected during ROV surveys provide a previously un-utilised resource to 'look back in time' and assess ecosystem dynamics through a temporal lens (Macreadie et al. 2018), with archives often dating back to the original installation period. However, the ecological value of ROV video, which is often collected haphazardly, remains unclear.

Industry ROV videos collected for inspection or other purposes need to be standardised prior to scientific evaluation. Several studies have utilised such ROV video for scientific purposes such as assessing marine algal and invertebrate growth (Gass & Roberts 2006; van der Stap et al. 2016; Thomson et al. 2018) and the ecology of fish populations on and around offshore platforms (Pradella et al. 2014; McLean et al. 2018a) and pipelines (Bond et al. 2018; McLean et al. 2017). All studies have implemented some form of standardisation of the video archives with varying degrees of success.

Stereo baited remote underwater video systems (BRUVS) are a well-established method for studying the abundance, biomass and diversity of marine communities (Cappo et al. 2006). Stereo-BRUVS are a relatively inexpensive and non-destructive sampling method that can be deployed across large spatial scales (Letessier et al.

2015). While usually used to study demersal communities, stereo-BRUVS have more recently been adapted to sample mid-water environments (Bouchet et al. 2018). A combination of benthic and mid-water stereo-BRUVS allows for the study of both demersal and pelagic marine faunal communities. Stereo-BRUVS are deployed in various marine environments around the world, according to standard operating procedures (see Bouchet et al*.* 2018; Langlois et al*.* 2018).

While ROV's and BRUVS have been used individually to study the ecology of offshore platforms, we propose using both sampling methods in tandem in order to gain a more complete understanding of the associated faunal communities. ROVs allow for targeted sampling of the infrastructure from the surface to the seafloor (McLean *et al.*, 2018b) whilst BRUVS are useful for larger-scale sampling. In Australia, the 500 m exclusion zone around offshore infrastructure effectively constitutes a *de facto* Marine Protected Area (MPA) (Friedlander et al. 2014). BRUVS allow for sampling of the pelagic and demersal species in this extended area – often called the ecological halo (Reeds et al. 2018) – which is influenced by the presence of the offshore platforms.

As a case study, we opportunistically utilised industry-collected ROV footage and conducted BRUVS surveys in the Wandoo oil field in north-west Australia, which is owned and operated by Vermillion Oil and Gas Australia. Wandoo is located on the north-west shelf of Australia, approximately 70 km offshore of Dampier, and consists of an unmanned monopod, Wandoo A, a four-shaft concrete gravity structure, Wandoo B, a Catenary Anchor Leg Mooring (CALM) Buoy, and associated subsea pipelines. Wandoo A and B were installed in 1993 and 1997 respectively and both sit in 54 m water depth.

ROV videos of the Wandoo platforms were available for 2007, 2008, 2011 and 2015. The videos varied within and between each year depending on the task, ranging from broad environmental surveys to targeted inspections and cleaning protocols, resulting in highly variable and non-standardised video. Based on the analysis of the Wandoo ROV videos and previous studies utilising industry ROV, we propose a new method of selecting videos for ecological studies that involves a stringent scoring system adapted from Pradella et al. (2014), with specific interest given to assessing vertical distributions of fish species. Using this scoring system, videos deemed useful for analysis must (1) follow the shaft or structure of interest in a distinct vertical transect, either descending or ascending, (2) have ≥ 5 m visibility, (3) be slow moving (<0.5 m/s, McLean et al. (2019)) to allow identification of fish species with no speed blur and (4) have the shaft/structure take up between 60-80% of the field of view (FOV).

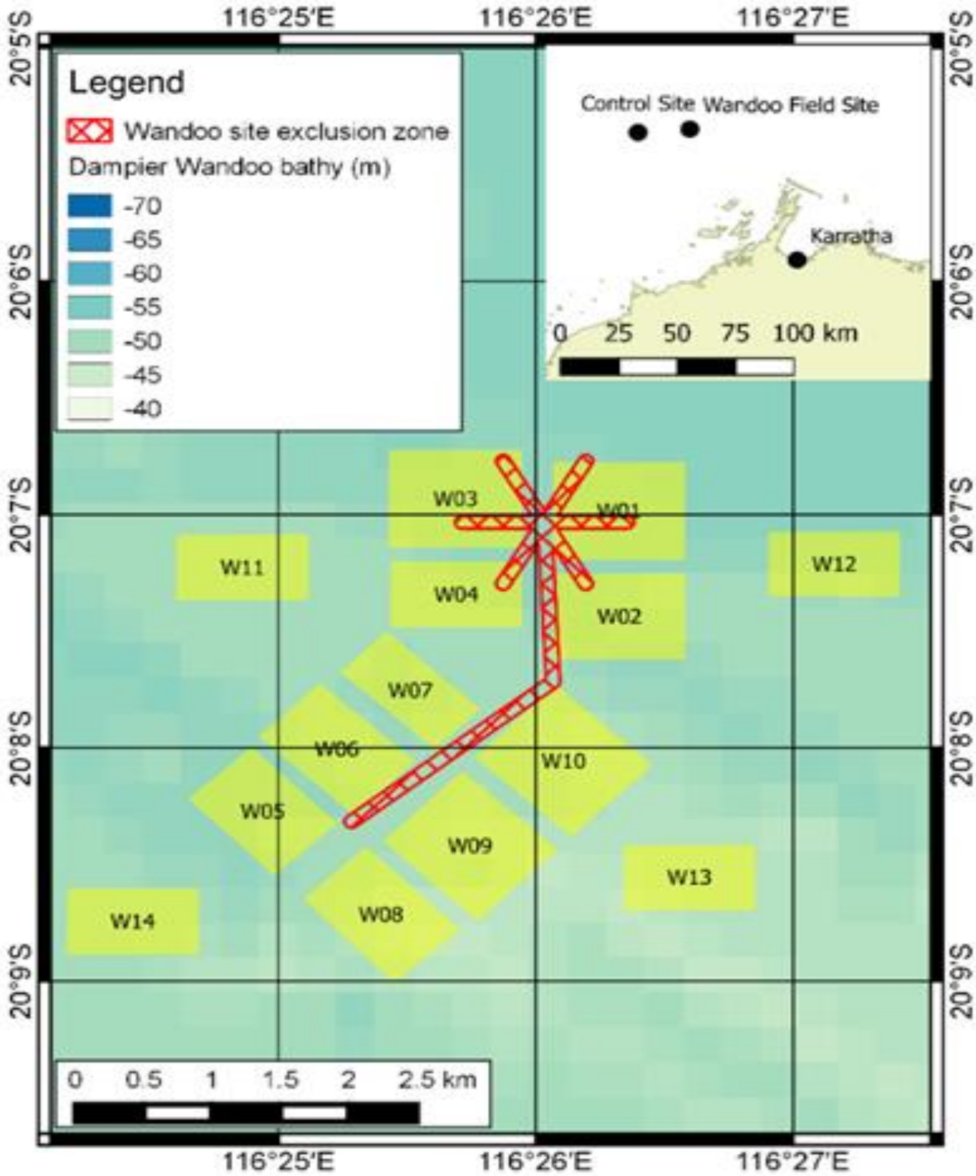

Figure A1.1 Stereo-BRUVS sampling sectors in the Wandoo Field.

Due to the varying speed of transects, analysis of the subset of usable ROV video should be conducted using a frame-by-frame method. This involves stratifying each video transect into standardised depth categories (e.g. 0-10 m, 10-20 m etc.) and analysing a set of individual video frames (i.e. paused video at a selected depth) within each depth category for fish identification. Subsampling by frame reduces the risk of speed bias, whereby transects conducted at slower speeds may have a greater number of fish visible. Following this method of selecting and analysing ROV footage may result in fewer videos being useful for analysis but will provide a more accurate representation of the fish communities that directly inhabit and associate with offshore infrastructure.

To understand the extent of the ecological halo of the Wandoo platforms, seabed BRUVS were deployed with a stratified random distribution throughout the Wandoo Field, with particular focus on Wandoo A, Wandoo B and the CALM Buoy. Specifically, multiple sectors were established throughout the Wandoo Field with five deployments of seabed BRUVS within each sector (Figure 1). This allows full coverage of the area of interest and addresses safety concerns associated with sampling near the infrastructure. Mid-water BRUVS were deployed in a subset of the sectors as well as at four “remote” sectors at least 2 km from the outer boundary of the near-site sectors. This design helps determine how abundance declines with distance from a central feature. Expeditions were conducted twice per year over a period of three years, allowing for seasonal and inter-annual comparisons of fish assemblages.

BRUVS were also deployed at two control sites: one being an area of natural “structure” of rocky substrate and similar spatial extent to the Wandoo infrastructure, and the other a flat, sandy area which is similar to what the Wandoo infrastructure was like prior to the installation of any subsea infrastructure. Both control sites are exposed to recreational and commercial fishing pressure, adding insight to the effect of the *de facto* MPA around Wandoo.

Ecological studies on offshore platforms have previously focused on the infrastructure and immediate surrounds through use of industry ROV video. However, the ecological influence of these structures can extend far beyond the platform itself, particularly due to the *de facto* MPA created by the 500 m exclusion zone. BRUVS represent an efficient, inexpensive, and well-established method for sampling these larger areas which could be just as ecologically important as the platforms themselves. Using a combination of ROV and BRUVS surveys as outlined here allows for a more complete method of documenting the ecology of offshore oil and gas fields, which can better inform decommissioning decisions.

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A1.4 Conflict of interest

This project is supported by Vermilion Oil and Gas Australia (Pty) Ltd., through the provision of ROV video archives and the VOGA Ph.D. Scholarship in Rigs-to-Reefs Ecology.

A1.5 References

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APPENDIX 2: SUMMARY OF EXPEDITIONS

Table A2.1 Summary of all expeditions in which seabed stereo-BRUVS were deployed. The table includes number of days over which the expedition occurred (Days), latitude (LAT) and longitude (LONG) of the locations in decimal degrees, and the number of seabed stereo-BRUVS deployed (n).

Location	Year	Start Date	End Date	Days	LAT	LONG	n
Northeast							
Torres Strait - East	2017	19/06/2017	30/11/2017	5	-10.08	143.63	93
Torres Strait - West	2017	13/06/2017	15/06/2017	3	-9.93	143.33	60
Ribbons - Central	2017	4/04/2017	6/12/2017	16	-13.76	143.89	364
	2018	15/04/2018	3/05/2018	20	-13.69	143.81	225
Ribbons - North	2017	16/06/2017	18/06/2017	3	-10.74	143.97	60
	2018	26/04/2018	28/04/2018	3	-12.24	143.27	40
Ribbons - South	2017	29/04/2017	29/04/2017	1	-14.28	144.77	20
	2018	14/04/2018	14/04/2018	1	-14.40	144.91	20
East Cape York - Middle	2017	7/06/2017	5/12/2017	4	-12.18	143.24	80
	2018	21/04/2018	27/04/2018	7	-12.25	143.22	40
East Cape York - North	2017	9/06/2017	3/12/2017	6	-11.52	142.99	140
	2018	23/04/2018	25/04/2018	3	-11.54	142.99	60
East Cape York - South	2017	6/04/2017	15/04/2017	10	-14.11	144.24	64
Cocos (Keeling Islands)							
Cocos	2016	10/11/2016	20/11/2016	11	-12.12	96.86	203
Northwest							
Adele Island	2017	23/07/2017	23/07/2017	1	-15.55	123.16	20
Ashmore Reef	2017	14/07/2017	21/07/2017	8	-12.24	123.03	160
	2018	2/10/2018	7/10/2018	8	-11.28	114.42	120
Barrow Island	2008	21/10/2008	27/10/2008	7	-20.83	115.51	159
	2009	17/03/2009	24/03/2009	8	-20.82	115.50	218
	2010	23/02/2010	1/03/2010	7	-20.79	115.48	180
Dampier Archipelago	2008	1/08/2008	18/08/2008	18	-20.47	116.72	419
Holothuria Reef	2017	12/07/2017	12/07/2017	1	-13.57	125.98	20
Long Reef	2017	30/06/2017	13/07/2017	14	-13.90	125.75	140
	2018	18/09/2018	23/09/2018	6	-13.97	125.75	120
Rowley Shoals	2017	19/11/2017	22/11/2017	4	-17.19	119.52	85
Wandoo							
Wandoo Platform	2017	4/05/2017	2/10/2017	8	-20.13	116.43	100
	2018	19/04/2018	7/09/2018	9	-20.13	116.43	100
	2019	25/04/2019	11/09/2019	10	-20.13	116.43	95
Wandoo Reef	2017	6/05/2017	4/10/2017	9	-20.15	116.22	100
	2018	25/04/2018	26/04/2018	2	-20.15	116.22	25
	2019	9/09/2019	11/09/2019	3	-20.15	116.22	50
Wandoo Sand	2018	22/04/2018	19/09/2018	6	-20.07	116.64	100
	2019	30/04/2019	30/04/2019	1	-20.07	116.64	25
Central North							
Ningaloo Reef - North	2006	22/04/2006	22/11/2006	30	-22.14	113.86	410
	2007	8/02/2007	11/12/2007	25	-22.10	113.89	350
	2009	26/03/2009	1/05/2009	10	-22.19	113.79	238
Ningaloo Reef - Middle	2006	7/05/2006	16/05/2006	10	-22.62	113.63	108
	2009	28/03/2009	2/05/2009	16	-22.73	113.59	274
Ningaloo Reef - South	2009	1/04/2009	10/04/2009	10	-23.76	113.32	183

Central South							
Shark Bay - Dirk Hartog Island	2017	16/09/2017	20/09/2017	5	-26.00	113.11	20
	2018	12/08/2018	12/08/2018	1	-26.01	113.12	20
Shark Bay - Gulf	2009	16/09/2009	20/09/2009	5	-25.95	113.22	324
Shark Bay - South Passage	2018	4/08/2018	4/08/2018	1	-26.15	113.20	10
Shark Bay - Steep Point	2017	15/09/2017	16/09/2017	2	-26.22	113.22	20
	2018	8/08/2018	11/08/2018	3	-26.32	113.26	45

Table A2.2 Summary of all expeditions in which mid-water stereo-BRUVS were deployed. The table includes number of days over which the expedition occurred (Days), latitude (LAT) and longitude (LONG) of the locations in decimal degrees, and the number of mid-water stereo-BRUVS deployed (n).

Location	Year	Start Date	End Date	Days	LAT	LONG	n
Northeast							
Great Barrier Reef - North	2017	7/06/2017	6/12/2017	22	11.17	143.44	72
Cocos (Keeling) Islands							
Cocos Island	2016	11/10/2016	21/11/2016	11	12.13	96.829	94
Northwest							
Ashmore Reef - North	2017	14/07/2017	21/07/2017	8	12.20	123.05	75
Ashmore Reef - South	2018	2/10/2018	7/10/2018	6	12.21	123.05	75
Long Reef East	2017	6/07/2017	13/07/2017	8	13.85	125.89	14
	2018	18/09/2018	23/09/2018	6	13.90	125.92	55
Long Reef West	2017	30/06/2017	12/07/2017	13	13.82	125.68	48
	2018	18/09/2018	23/09/2018	6	13.85	125.55	59
Montebello Islands	2018	17/08/2018	23/08/2018	7	-20.28	115.36	98
Montebello Islands - Offshore	2018	15/08/2018	22/08/2018	8	19.88	115.35	98
Muiron Islands	2018	25/07/2018	25/07/2018	1	21.61	114.20	19
Rowley Shoals	2017	19/11/2017	22/11/2017	4	17.09	119.42	38
Rowley Shoals - Offshore	2017	16/11/2017	18/11/2017	3	15.14	118.49	59
	2018	4/08/2018	10/08/2018	7	15.45	118.52	179
Wandoo							
Wandoo Platform	2017	1/05/2017	2/10/2017	11	20.13	116.42	42
	2018	19/04/2018	7/09/2018	12	20.13	116.42	43
	2019	26/04/2019	11/09/2019	9	20.12	116.43	37
Wandoo Reef	2017	7/05/2017	4/10/2017	8	20.14	116.21	41
	2018	25/04/2018	26/04/2018	5	20.15	116.2	24
	2019	9/09/2019	11/09/2019	3	20.14	116.21	35
Wandoo Sand	2018	22/04/2018	19/09/2018	6	20.06	116.63	44
	2019	30/04/2019	30/04/2019	1	20.06	116.63	19
Central North							
Ningaloo Reef - Offshore	2016	17/09/2016	22/09/2016	6	21.79	113.45	43
Ningaloo Reef	2016	15/09/2016	22/09/2016	8	21.93	113.77	25
	2018	24/07/2018	30/07/2018	7	21.89	113.80	79

Central South							
Shark Bay -Dirk Hartog Island	2017	16/09/2017	20/09/2017	5	26.04	112.96	30
	2018	6/08/2018	11/08/2018	6	25.94	112.94	32
Shark Bay - Gulf	2012	19/04/2012	25/04/2012	7	26.12	113.17	56
Shark Bay - Steep Point	2012	18/04/2012	19/04/2012	3	26.15	113.13	10
	2017	15/09/2017	21/09/2017	7	26.28	113.12	45
	2018	6/08/2018	11/08/2018	6	26.30	113.13	61

APPENDIX 3: IDENTIFICATION OF POTENTIAL SPECIES POOL

Table A3.1 Potential species pool of tropical Australian threatened elasmobranchs, derived from Fishbase and Atlas of Living Australia (Froese & Pauly 2019; www.ala.org.au 2020). The IUCN Red List classification (IUCN) of each species is included (IUCN 2020). Taxa identifications are in bold. For each family, identifications to genus are listed with all possible species in that genus. Identifications to family are listed thereafter, followed by any possible species in that family not already listed.

Taxa	Common name	IUCN
Carcharhinidae		
***Carcharhinus* sp.**		
Carcharhinus albimarginatus	silvertip shark	Vulnerable
Carcharhinus altimus	bignose shark	Data Deficient
Carcharhinus amblyrhynchoides	graceful shark	Near Threatened
Carcharhinus amblyrhynchos	blacktail reef shark	Near Threatened
Carcharhinus amboinensis	pigeye shark	Data Deficient
Carcharhinus brevipinna	spinner shark	Near Threatened
Carcharhinus cautus	nervous shark	Data Deficient
Carcharhinus falciformis	silky shark	Vulnerable
Carcharhinus fitzroyensis	creek whaler	Least Concern
Carcharhinus leucas	bull shark	Near Threatened
Carcharhinus limbatus	blacktip shark	Near Threatened
Carcharhinus longimanus	oceanic whitetip shark	Critically Endangered
Carcharhinus macloti	hardnose shark	Near Threatened
Carcharhinus melanopterus	blacktip reef shark	Near Threatened
Carcharhinus obscurus	dusky shark	Endangered
Carcharhinus plumbeus	sandbar shark	Vulnerable
Carcharhinus sorrah	spot-tail shark	Near Threatened
Carcharhinus tilstoni	Australian blacktip shark	Least Concern
Carcharhinidae sp.		
Galeocerdo cuvier	tiger shark	Near Threatened
Glyphis garricki	northern river shark	Critically Endangered
Loxodon macrorhinus	sliteye shark	Least Concern
Negaprion acutidens	lemon shark	Vulnerable
Prionace glauca	blue shark	Near Threatened
Rhizoprionodon acutus	milk shark	Least Concern
Rhizoprionodon taylori	Australian sharpnose shark	Least Concern
Triaenodon obesus	white tip reef shark	Near Threatened
Myliobatidae		
***Mobula* sp.**		
Mobula alfredi	reef manta	Vulnerable
Mobula birostris	giant manta	Vulnerable
Mobula eregoodootenkee	longhorned mobula	Endangered
Mobula thurstoni	bentfin devilray	Endangered
Rhinidae		
***Rhynchobatus* sp.**		
Rhynchobatus palpebratus	eyebrow wedgefish	Near Threatened
Rhynchobatus australiae	bottlenose wedgefish	Critically Endangered
Rhynhcobatus laevis	smoothnose wedgefish	Critically Endangered

Table A3.2 Potential species identifications for those taxa identified to family or genus, separated by family. Species records for the area surrounding Wandoo and the two control sites are derived from Fishbase, Sealifebase and Atlas of Living Australia (Froese & Pauly 2019; Palomares & Pauly 2019; www.ala.org.au 2020). Taxa identifications are in bold. For each family, identifications to genus are listed with all possible species in that genus. Identifications to family are listed thereafter, followed by any possible species in that family not already listed.

Binomial	Common Name
Acanthuridae	
***Acanthurus* sp.**	
Acanthurus auranticavus	ringtail surgeonfish
Acanthurus blochii	dark surgeonfish
Acanthurus dussumieri	pencil surgeonfish
Acanthurus grammoptilus	inshore surgeonfish
Acanthurus leucocheilus	pale-lipped surgeonfish
Acanthurus lineatus	bluelined surgeonfish
Acanthurus mata	pale surgeonfish
Acanthurus nigricans	velvet surgeonfish
Acanthurus nigricauda	eyeline surgeonfish
Acanthurus nigrofuscus	dusky surgeonfish
Acanthurus olivaceus	orangeblotch surgeonfish
Acanthurus pyroferus	mimic surgeonfish
Acanthurus triostegus	convict surgeonfish
Acanthurus xanthopterus	yellowmask surgeonfish
***Naso* sp.**	
Naso annulatus	ringtail unicornfish
Naso brevirostris	spotted unicornfish
Naso caesius	silverblotched unicornfish
Naso fageni	horseface unicornfish
Naso hexacanthus	sleek unicornfish
Naso lituratus	clown unicornfish
Naso lopezi	slender unicornfish
Naso mcdadei	squarenose unicornfish
Naso reticulatus	reticulate unicornfish
Naso unicornis	bluespine unicornfish
Naso vlamingii	bignose unicornfish
Apogonidae	
Apogonidae sp.	
Apogon crassiceps	ruby cardinalfish
Apogon semiornatus	halfband cardinalfish
Apogon unicolor	big red cardinalfish

Binomial	Common Name
Apogonichthyoides atripes	bullseye cardinalfish
Apogonichthyoides brevicaudatus	manyband cardinalfish
Apogonichthyoides timorensis	Timor cardinalfish
Apogonichthyoides umbratilis	cryptic cardinalfish
Cheilodipterus macrodon	tiger cardinalfish
Cheilodipterus quinquelineatus	fiveline cardinalfish
Foa fo	samoan cardinalfish
Fowleria aurita	crosseye cardinalfish
Fowleria variegata	variegated cardinalfish
Jaydia argyrogaster	silvermouth siphonfish
Jaydia carinata	keeled cardinalfish
Jaydia melanopus	monster cardinalfish
Jaydia truncata	flagfin cardinalfish
Neamia articycla	circular cardinalfish
Nectamia fusca	ghost cardinalfish
Ostorhinchus angustatus	broadstripe cardinalfish
Ostorhinchus atrogaster	blackbelly cardinalfish
Ostorhinchus aureus	ringtail cardinalfish
Ostorhinchus cavitensis	whiteline cardinalfish
Ostorhinchus cookii	Cook's cardinalfish
Ostorhinchus cyanosoma	orangelined cardinalfish
Ostorhinchus doederleini	fourline cardinalfish
Ostorhinchus fasciatus	striped cardinalfish
Ostorhinchus monospilus	moluccan cardinalfish
Ostorhinchus novemfasciatus	nineline cardinalfish
Ostorhinchus pallidofasciatus	palestriped cardinalfish
Ostorhinchus properuptus	coral cardinalfish
Ostorhinchus rueppellii	western gobbleguts
Ostorhinchus semilineatus	blacktip cardinalfish
Ostorhinchus septemstriatus	sevenband cardinalfish
Ostorhinchus taeniophorus	pearly-line cardinalfish
Ostorhinchus wassinki	Kupang cardinalfish
Ozichthys albimaculosus	creamspotted cardinalfish
Paxton concilians	Paxton's cardinalfish
Pristiapogon exostigma	oneline cardinalfish
Pristiapogon fraenatus	spinyeye cardinalfish
Pristiapogon unitaeniatus	singlestripe cardinalfish
Pristicon rhodopterus	twobar cardinalfish
Pristicon trimaculata	threespot cardinalfish
Pseudamia gelatinosa	gelatinous cardinalfish
Quinca mirifica	sailfin cardinalfish
Rhabdamia gracilis	slender cardinalfish

Binomial	Common Name
Siphamia majimai	striped siphonfish
Siphamia roseigaster	pinkbreast siphonfish
Siphamia tubifer	urchin cardinalfish
Taeniamia fucata	painted cardinalfish
Taeniamia melasma	blackspot cardinalfish
Asteroidea	
Asteroidea sp.	
Anthenea aspera	cake star
Anthenea pentagonula	-
Anthenea sibogae	-
Anthenea viguieri	-
Anthenoides dubius	-
Archaster angulatus	sand sea star
Asterodiscides soelae	-
Astropecten granulatus	-
Astropecten polyacanthus	comb sea star
Astropecten zebra	-
Coronaster halicepus	-
Ctenodiscus orientalis	starfish
Culcita novaeguineae	cushion seastar
Culcita schmideliana	pincushion starfish
Echinaster luzonicus	Luzon seastar
Echinaster varicolor	-
Fromia indica	red starfish
Gomophia sphenisci	-
Goniodiscaster acanthodes	-
Goniodiscaster forficulatus	-
Goniodiscaster rugosus	-

Binomial	Common Name
Gymnanthenea globigera	-
Hacelia helicosticha	-
Halityle regularis	mosaic cushion star
Heteronardoa carinata	-
Indianastra sarasini	-
Linckia guildingi	common comet star
Linckia laevigata	blue seastar
Linckia multifora	spotted linckia
Luidia hardwicki	luidia sand star
Luidia maculata	-
Metrodira subulata	-
Nardoa galatheae	galathea sea star
Ogmaster capella	-
Ophidiaster granifer	grained seastar
Pentaceraster gracilis	gracilis seastar

Binomial	Common Name
Pentaceraster regulus	-
Protoreaster nodulosus	knobbly seastar
Pseudoreaster obtusangulus	-
Rosaster symbolicus	-
Stellaster childreni	-
Stellaster equestris	-
Stellaster inspinosus	-
Stellaster squamulosus	-
Tamaria tumescens	-
Thromidia brycei	-
Balistidae	
***Pseudobalistes* sp.**	
Pseudobalistes flavimarginatus	yellowmargin triggerfish
Pseudobalistes fuscus	yellowspotted triggerfish
Balistidae sp.	
Abalistes filamentosus	hairfin triggerfish
Abalistes stellatus	starry triggerfish
Balistapus undulatus	orangestripe triggerfish
Odonus niger	redtooth triggerfish
Rhinecanthus aculeatus	hawaiian triggerfish
Sufflamen chrysopterus	eye-stripe triggerfish
Sufflamen fraenatum	bridled triggerfish
Xanthichthys lineopunctatus	lined triggerfish
Blenniidae	
***Meiacanthus* sp.**	
Meiacanthus grammistes	linespot fangblenny
Meiacanthus luteus	yellow fangblenny
***Plagiotremus* sp.**	
Plagiotremus rhinorhynchos	bluestriped fangblenny
Plagiotremus tapeinosoma	piano fangblenny
Blenniidae sp.	
Aspidontus dussumieri	lance blenny
Aspidontus taeniatus	false cleanerfish
Atrosalarias fuscus	dusky blenny
Blenniella chrysospilos	redspotted rockskipper
Blenniella periophthalmus	bluestreaked rockskipper
Cirripectes alleni	kimberley blenny
Cirripectes castaneus	chestnut blenny
Cirripectes filamentosus	filamentous blenny
Crossosalarias macrospilus	triplespot blenny
Ecsenius alleni	Allen's combtooth blenny
Ecsenius bicolor	bicolor combtooth blenny
Ecsenius lineatus	lined combtooth blenny

Binomial	Common Name
Ecsenius oculatus	ocular combtooth blenny
Ecsenius yaeyamaensis	palespotted combtooth blenny
Entomacrodus decussatus	wavyline rockskipper
Entomacrodus striatus	blackspotted rockskipper
Entomacrodus thalassinus	twinspot rockskipper
Glyptoparus delicatulus	delicate blenny
Istiblennius edentulus	rippled rockskipper
Istiblennius lineatus	lined rockskipper
Istiblennius meleagris	peacock rockskipper
Laiphognathus multimaculatus	manyspot blenny
Mimoblennius atrocinctus	mimic blenny
Omobranchus germaini	Germain's blenny
Omobranchus punctatus	muzzled blenny
Omobranchus rotundiceps	rotund blenny
Omobranchus verticalis	vertical blenny
Petroscirtes breviceps	shorthead sabretooth blenny
Petroscirtes mitratus	crested sabretooth blenny
Salarias fasciatus	banded blenny
Salarias sexfilum	Spalding's blenny
Stanulus talboti	Talbot's blenny
Xiphasia setifer	hairtail blenny
Bothidae	
***Bothus* sp.**	
Bothus myriaster	oval flounder
Bothus pantherinus	leopard flounder
Brachyura	
Brachyura sp.	
Atergatopsis alcocki	-

Binomial	Common Name
Atergatopsis tweediei	-
Banareia armata	-
Bathypilumnus nigrispinifer	-
Bathypilumnus pugilator	-
Calappa capellonis	-
Calappa clypeata	-
Calappa philargius	red-spotted box crab
Calappa woodmasoni	little crested crab
Charybdis (Charybdis) granulata	-
Charybdis (Charybdis) jaubertensis	-
Cryptodromiopsis unidentata	-
Cycloachelous orbitosinus	-
Demania splendida	-
Dorippe quadridens	-

Binomial	Common Name
Dromidiopsis edwardsi	sponge crab
Eumedonus niger	-
Gaillardiellus rueppelli	-
Glabropilumnus seminudus	-
Hepatoporus guinotae	-
Hyastenus sebae	-
Hyastenus spinosus	-
Izanami curtispina	-
Izanami inermis	-
Laleonectes nipponensis	-
Lissocarcinus laevis	-
Lissoporcellana pectinata	-
Lissoporcellana quadrilobata	-
Lupocyclus rotundatus	-
Lupocyclus tugelae	-
Menaethius monoceros	-
Myra eudactylus	-
Myrine kessleri	-
Naxioides taurus	-
Neopalicus jukesii	-
Neoxanthops lineatus	-
Oncinopus aranea	thin-shelled spider crab
Pachycheles sculptus	sculptured porcelain crab
Palapedia quadriceps	-
Palapedia roycei	-
Paramaya spinigera	-
Paranaxia serpulifera	-
Petrolisthes militaris	-

Binomial	Common Name
Petrolisthes scabriculus	-
Pilumnus minutus	-
Pilumnus scabriusculus	-
Pilumnus semilanatus	ragged crab
Platypodia semigranosa	-
Polyonyx biunguiculatus	-
Portunus armatus	blue swimmer crab
Portunus gladiator	-
Portunus gracilimanus	-
Portunus longispinosus	-
Portunus rugosus	-
Portunus tuberculosus	-
Prismatopus longispinus	-
Schizophrys dama	pronghorn decorator crab
Thalamita quadrilobata	-

Binomial	Common Name
Thalamita sexlobata	-
Thalamita spinifera	-
Tokoyo eburnea	-
Trigonoplax spathulifera	-
Urnalana pulchella	-
Zebrida adamsi	-
Caesionidae	
***Pterocaesio* sp.**	
Pterocaesio chrysozona	yellowband fusilier
Pterocaesio digramma	doubleline fusilier
Pterocaesio tile	neon fusilier
Caesionidae sp.	
Caesio caerulaurea	goldband fusilier
Caesio cuning	yellowtail fusilier
Caesio teres	blue fusilier
Dipterygonotus balteatus	mottled fusilier
Carangidae	
***Alepes* sp.**	
Alepes apercna	smallmouth scad
Alepes kleinii	razorbelly trevally
Alepes vari	herring scad
***Carangoides* sp.**	
Carangoides armatus	longfin trevally
Carangoides chrysophrys	longnose trevally
Carangoides coeruleopinnatus	onion trevally
Carangoides dinema	shadow trevally
Carangoides equula	whitefin trevally
Carangoides ferdau	blue trevally

Binomial	Common Name
Carangoides fulvoguttatus	turrum
Carangoides gymnostethus	bludger trevally
Carangoides hedlandensis	bumpnose trevally
Carangoides humerosus	epaulette trevally
Carangoides malabaricus	Malabar trevally
Carangoides oblongus	coachwhip trevally
Carangoides orthogrammus	thicklip trevally
***Caranx* sp.**	
Caranx bucculentus	bluespotted trevally
Caranx ignobilis	giant trevally
Caranx melampygus	bluefin trevally
Caranx papuensis	brassy trevally
Caranx sexfasciatus	bigeye trevally
Caranx tille	tille trevally
***Decapterus* sp.**	

Binomial	Common Name
Decapterus macarellus	mackerel scad
Decapterus macrosoma	slender scad
Decapterus russelli	Indian scad
Decapterus tabl	rough-ear scad
***Scomberoides* sp.**	
Scomberoides commersonnianus	giant queenfish
Scomberoides lysan	lesser queenfish
Scomberoides tol	needleskin queenfish
***Selar* sp.**	
Selar boops	oxeye scad
Selar crumenophthalmus	bigeye scad
***Seriola* sp.**	
Seriola dumerili	amberjack
Seriola hippos	samsonfish
Seriola rivoliana	highfin amberjack
Carangidae sp.	
Atule mate	barred yellowtail scad
Gnathanodon speciosus	golden trevally
Megalaspis cordyla	finny scad
Naucrates ductor	pilotfish
Parastromateus niger	black pomfret
Pseudocaranx georgianus	silver trevally
Selaroides leptolepis	yellowstripe scad
Trachinotus baillonii	smallspotted dart
Trachinotus blochii	snubnose dart
Trachurus declivis	common jack mackerel
Trachurus novaezelandiae	yellowtail scad
Ulua aurochs	silvermouth trevally

Binomial	Common Name
Ulua mentalis	longraker trevally
Uraspis uraspis	whitemouth trevally
Carcharhinidae	
***Carcharhinus* sp.**	
Carcharhinus altimus	bignose shark
Carcharhinus amblyrhynchos	grey reef shark
Carcharhinus amboinensis	pigeye shark
Carcharhinus brevipinna	spinner shark
Carcharhinus coatesi	whitecheek shark
Carcharhinus falciformis	silky shark
Carcharhinus galapagensis	Galapagos shark
Carcharhinus leucas	bull shark
Carcharhinus limbatus	common blacktip shark
Carcharhinus longimanus	oceanic whitetip shark
Carcharhinus melanopterus	blacktip reef shark

Binomial	Common Name
Carcharhinus obscurus	dusky sharl
Carcharhinus plumbeus	sandbar shark
Carcharhinus sorrah	spot-tail shark
Carcharhinus tilstoni	Australian blacktip shark
Carcharhinidae sp.	
Galeocerdo cuvier	tiger shark
Glyphis garricki	northern river shark
Loxodon macrorhinus	sliteye shark
Negaprion acutidens	lemon shark
Prionace glauca	blue shark
Rhizoprionodon acutus	milk shark
Rhizoprionodon taylori	Australian sharpnose shark
Triaenodon obesus	white tip reef shark
Chaetodontidae	
***Coradion* sp.**	
Coradion altivelis	highfin coralfish
Coradion chrysozonus	orangebanded coralfish
***Heniochus* sp.**	
Heniochus acuminatus	longfin bannerfish
Heniochus chrysostomus	pennant bannerfish
Heniochus diphreutes	schooling bannerfish
Heniochus monoceros	masked bannerfish
Heniochus singularius	singular bannerfish
Chaetodontidae sp.	
Chaetodon adiergastos	Philippine butterflyfish
Chaetodon assarius	western butterflyfish
Chaetodon aureofasciatus	goldstripe butterflyfish
Chaetodon auriga	threadfin butterflyfish

Binomial	Common Name
Chaetodon bennetti	eclipse butterflyfish
Chaetodon citrinellus	citron butterflyfish
Chaetodon ephippium	saddle butterflyfish
Chaetodon kleinii	Klein’s butterflyfish
Chaetodon lineolatus	lined butterflyfish
Chaetodon lunula	racoon butterflyfish
Chaetodon lunulatus	pinstripe butterflyfish
Chaetodon melannotus	blackback butterflyfish
Chaetodon ornatissimus	ornate butterflyfish
Chaetodon plebeius	bluespot butterflyfish
Chaetodon speculum	ovalspot butterflyfish
Chaetodon trifascialis	chevron butterflyfish
Chaetodon ulietensis	doublesaddle butterflyfish
Chaetodon unimaculatus	teardrop butterflyfish
Chaetodon vagabundus	vagabond butterflyfish

Binomial	Common Name
Chelmon marginalis	margined coralfish
Chelmon muelleri	Muller’s coralfish
Chelmon rostratus	beaked coralfish
Forcipiger flavissimus	forceps fish
Parachaetodon ocellatus	ocellate butterflyfish
Roa australis	tripleband butterflyfish
Cheloniidae	
Cheloniidae sp.	
Caretta caretta	loggerhead turtle
Chelonia mydas	green turtle
Eretmochelys imbricata	hawksbill turtle
Natator depressus	flatback turtle
Cirrhitidae	
***Paracirrhites* sp.**	
Paracirrhites forsteri	freckled hawkfish
Paracirrhites arcatus	arc-eye hawkfish
Paracirrhites hemistictus	whitespot hawkfish
***Cirrhitidae* sp.**	
Cirrhitichthys aprinus	blotched hawkfish
Cirrhitichthys falco	dwarf hawkfish
Cyprinocirrhites polyactis	lyretail hawkfish
Clupeidae	
Sardinella sp.	
Sardinella albella	white sardinella
Sardinella gibbosa	goldstripe sardinella
Sardinella lemuru	bali sardinella
Sardinella melanura	blacktip sardinella
Clupeidae sp.	

Binomial	Common Name
Amblygaster sirm	spotted sardine
Dussumieria elopsoides	slender sardine
Herklotsichthys koningsbergeri	largespotted herring
Herklotsichthys lippa	smallspotted herring
Spratelloides delicatulus	blueback sprat
Spratelloides gracilis	slender sprat
Spratelloides robustus	blue sprat
Congridae	
***Gorgasia* sp.**	
Gorgasia maculata	whitespotted garden eel
Gorgasia preclara	splendid garden eel
Crinoidea	
Crinoidea sp.	
Amphimetra tessellata	-
Cenometra cornuta	-

Binomial	Common Name
Colobometra perspinosa	-
Comatella nigra	-
Comatula rotalaria	-
Dorometra parvicirra	-
Heterometra crenulata	-
Petasometra clarae	-
Phanogenia distinctus	-
Pterometra pulcherrima	-
Dasyatidae	
***Neotrygon* sp.**	
Neotrygon annotata	plain maskray
Neotrygon australiae	bluespotted maskray
Neotrygon leylandi	painted maskray
Dasyatidae sp.	
Bathytoshia brevicaudata	smooth stingray
Hemitrygon parvonigra	dwarf black stingray
Himantura australis	reticulate whipray
Himantura leoparda	leopard whipray
Maculabatis astra	blackspotted whipray
Maculabatis toshi	brown whipray
Pastinachus ater	cowtail stingray
Pateobatis fai	pink whipray
Pateobatis jenkinsii	Jenkins' whipray
Taeniura lymma	bluespotted fantail ray
Taeniurops meyeni	blotched fantail ray
Urogymnus asperrimus	porcupine ray
Urogymnus granulatus	mangrove whipray
Delphinidae	
Delphinidae sp.	
Sousa sahulensis	Australian humpbacked dolphin
Stenella longirostris	spinner dolphin
Tursiops truncatus	bottlenose dolphin
Echeneidae	
***Remora* sp.**	
Remora remora	remora
Remora albescens	white suckerfish
Remora australis	whalesucker
Remora osteochir	marlin sucker
Echeneidae sp.	
Echeneis naucrates	sharksucker
Echinoidea	
Echinoidea sp.	
Astropyga radiata	-

Binomial	Common Name
Breynia australasiae	-
Breynia desorii	-
Brissus latecarinatus	heart urchin
Chaetodiadema granulatum	sea urchin
Clypeaster telurus	-
Clypeaster virescens	-
Diadema savignyi	-
Diadema setosum	needlespined sea urchin
Echinocyamus crispus	-
Echinocyamus planissimus	-
Echinodiscus auritus	-
Echinolampas ovata	-
Echinometra mathaei	burrowing sea urchin
Echinostrephus molaris	-
Lovenia elongata	-
Metabonellia haswelli	-
Metalia angustus	heart urchin
Metalia sternalis	heart urchin
Nudechinus darnleyensis	-
Nudechinus scotiopremnus	-
Peronella lesueuri	-
Peronella orbicularis	-
Peronella tuberculata	-
Phyllacanthus imperialis	-
Phyllacanthus longispinus	-
Prionocidaris baculosa	-
Prionocidaris bispinosa	-
Rhynobrissus hemiasteroides	-
Salmaciella dussumieri	-
Salmacis belli	-
Schizaster (Schizaster) compactus	-
Stylocidaris bracteata	sea urchin
Temnopleurus alexandri	-
Temnopleurus michaelseni	-
Toxopneustes pileolus	-
Tripneustes gratilla	collector sea urchin
Elapidae	
***Aipysurus* sp.**	
Aipysurus apraefrontalis	short-nosed seasnake
Aipysurus duboisii	reef shallows seasnake
Aipysurus laevis	golden seasnake
Aipysurus tenuis	brown-lined seasnake
***Hydrophis* sp.**	

Binomial	Common Name
Hydrophis czeblukovi	fine-spined seasnake
Hydrophis elegans	elegant seasnake
Hydrophis major	olive-headed seasnake
Hydrophis ocellatus	spotted seasnake
Hydrophis ornatus	ornate reef sea snake
Hydrophis stokesii	Stokes's seasnake
Elapidae sp.	
Brachyurophis approximans	north-western shovel-nosed snake
Demansia rufescens	rufous whipsnake
Emydocephalus annulatus	turtle-headed seasnake
Ephalophis greyi	north-western mangrove seasnake
Furina ornata	orange-naped snake
Parasuta monachus	monk snake
Pseudechis australis	king brown snake
Pseudonaja mengdeni	western brown snake
Suta punctata	little spotted snake
Ephippidae	
***Platax* sp.**	
Platax batavianus	humphead batfish
Platax orbicularis	round batfish
Platax pinnatus	longfin batfish
Ephippidae sp.	
Zabidius novemaculeatus	shortfin batfish
Fistulariidae	
***Fistularia* sp.**	
Fistularia commersonii	smooth flutemouth
Fistularia petimba	rough flutemouth
Gastropoda	

Binomial	Common Name
Gastropoda sp.	
Adamnestia arachis	-
Akera soluta	-
Allochroa layardi	-
Amalda lineata	-
Amoria dampieria	-
Amoria grayi	Gray's volute
Amoria macandrewi	Macandrew's volute
Amoria praetexta	juvenile volute
Ancillista muscae	elongate ancilla
Angaria delphinus	imperial delphinula
Aplysia dactylomela	-
Aplysia parvula	-
Archimediella dirkhartogensis	-
Archimediella fastigiata	-

Binomial	Common Name
Aspella platylaevis	-
Astralium calcar	spurred turban shell
Astralium pileolum	frilled star
Astralium squamiferum	scaly star shell
Astralium stellare	blue mouthed turban
Atys cylindricus	-
Atys naucum	-
Atys semistriatus	-
Austrocochlea zeus	dory austrocochlea
Berthella martensi	-
Berthellina citrina	-
Bistolida hirundo	swallow cowry
Blasicrura pallidula	-
Bostrycapulus pritzkeri	-
Bufonaria rana	frog shell
Bulla ampulla	-
Bulla vernicosa	-
Bullina lineata	-
Bursa granularis	granulated bursa
Cabestana tabulata	Waterhouse's triton
Calthalotia mundula	-
Canarium mutabile	flower stromb
Cantharidus crenelliferus	-
Cantharidus gilberti	-
Cantharidus polychroma	-
Cantharus erythrostomus	-
Cassidula (Cassidula) aurisfelis	-
Cavolinia uncinata	-

Binomial	Common Name
Cellana radiata	radiate patellid limpet
Cellana turbator	-
Cerithium atromarginatum	-
Cerithium balteatum	-
Cerithium novaehollandiae	creeper
Cerithium torresi	-
Cerithium traillii	-
Cerithium zonatum	-
Cheilea equestris	cup & saucer limpet
Chelidonura amoena	-
Chelidonura hirundinina	-
Chicoreus (Chicoreus) cornucervi	single tooth murex
Chicoreus (Triplex) cervicornis	murex shell
Chicoreus (Triplex) microphyllus	short-fronded murex
Chicoreus (Triplex) strigatus	Penchinatt's murex

Binomial	Common Name
Chicoreus (Triplex) torrefactus	the scorched murex
Cinguloterebra marrowae	-
Cirsotrema varicosa	varicose ladder shell
Clanculus atropurpureus	-
Clanculus comarilis	-
Clanculus margaritarius	-
Clivipollia incarnata	fleshy peristernia
Clypeomorus batillariaeformis	creeper
Clypeomorus bifasciata	double-banded creeper
Colina macrostoma	-
Colsyrnola sericea	-
Cominella (Cominella) acutinodosa	nodoluse cominella
Conasprella (Fusiconus) orbignyi	d'Orbigny's cone
Conuber conicus	conical sand snail
Conus (Cylinder) textile	the cloth-of-gold cone
Conus (Cylinder) victoriae	Queen Victoria'sp. cone
Conus (Gastridium) geographus	geographer cone
Conus (Leporiconus) glans	acorn cone
Conus (Lividoconus) eximius	choice cone
Conus (Lividoconus) lischkeanus	-
Conus (Phasmoconus) dampierensis	-
Conus (Plicaustraconus) trigonus	triangular cone
Conus (Rhizoconus) pertusus	pricked cone
Conus (Rhizoconus) vexillum	flag cone
Conus (Tesselliconus) suturatus	-
Conus monachus	-

Binomial	Common Name
Coralliophila confusa	-
Coralliophila costularis	small-ribbed purpura
Crepidula aculeata	slipper limpet
Cribrarula cribraria	-
Cronia (Cronia) avellana	filbert-nut buccinum
Cupidoliva nympha	nymph rice shell
Cyerce nigricans	-
Cyllene sulcata	-
Cymbiola nivosa	blotched snowflake volute
Dermomurex (Viator) antonius	-
Diacavolinia longirostris	-
Diala albugo	-
Diala lirulata	-
Diodora jukesii	keyhole limpet
Diodora singaporensis	-

Binomial	Common Name
Distorsio reticularis	reticulate triton
Dolomena plicata	-
Doxander campbelli	Campbell's stromb
Doxander vittatus	riband marked stromb
Drupella margariticola	oyster drill
Drupella rugosa	hime-shiro-reishi-damashi
Duplicaria duplicata	duplicate auger
Echinolittorina (Granulittorina) vidua	-
Eclogavena quadrimaculata	-
Elysia ornata	-
Elysiella pusilla	-
Emarginula (Emarginula) incisura	-
Eoacmaea calamus	-
Eratoena corrugata	-
Eratoena gemma	-
Ergalatax contracta	contracted buccinum
Erronea caurica	-
Erronea cylindrica	cylindrical cowry
Erronea errones	erroneus cowry
Ethminolia vitiliginea	depressed top shell
Euchelus atratus	the black beaded top shell
Euchelus dampierensis	-
Euchelus rubrus	red bead shell
Euplica bidentata	-
Euplica varians	-
Euselenops luniceps	-
Ficus eospila	-
Fusiaphera macrospira	-
Fusinus (Fusinus) colus	distaff spindle
Fusinus (Fusinus) undatus	-
Fusolatirus paetelianus	-
Gastrocopta hedleyi	brigalow pupasnail
Gastrocopta mussoni	Musson's pupasnail
Gemmula (Gemmula) dampierana	-
Gemmula (Gemmula) diomedea	-
Gibberula striata	-
Granata maculata	-
Gyrineum lacunatum	-
Haliotis clathrata	-
Haliotis diversicolor	-
Haliotis varia	variable abalone

Binomial	Common Name
Haloa cymbalum	-
Harpa articularis	articulate harp shell
Haustator (Kurosioia) cingulifera	-
Heliacus (Heliacus) variegatus	variegated sundial
Herpetopoma instrictum	-
Herpetopoma scabriuscula	scurfy bead shell
Hiatavolva depressa	depressed little egg cowry
Homalocantha secunda	next-allied murex
Hybochelus cancellatus	-
Hydatina amplustre	-
Hydatina physis	-
Indomodulus tectum	-
Inquisitor dampierius	-
Inquisitor intertincta	-
Inquisitor odhneri	-
iravadia (pseudonoba) densilabrum	
Iravadia pilbara	-
Ittibittium parcum	-
Labiostrombus epidromis	sail stromb
Laetifautor monilis	-
Latirus walkeri	-
Liotina crassibassis	-
Liotina peronii	large liotia
Littoraria cingulata	periwinkle
Littoraria scabra	scabra periwinkle
Lophiotoma acuta	-
Lunella (Lunella) cinereus	polished turban
Luria isabella	fawn-coloured cowry

Binomial	Common Name
Lyncina carneola	purple mouthed cowry
Macroschisma madreporaria	-
Macroschisma munita	ridge-backed keyhole limpet
Macroschisma producta	elongated keyhole limpet
Maculotriton serriale	granulated castor bean
Malea pomum	apple tun
Mammilla simiae	monkey sand shell
Mancinella alouina	pimpled purpura
Mancinella echinata	whelk
Marmorofusus nicobaricus	-
Melampus (Melampus) flexuosus	-
Melanella montagueana	-
Melo amphora	melon shell
Melo umbilicatus	bailer shell
Merica melanostoma	-

Binomial	Common Name
Mesoginella brachia	-
Micromelo undata	-
Monetaria caputserpentis	-
Monetaria moneta	money cowrie
Monilea callifera	shrewd trochid
Monodonta labio	lipped periwinkle
Monoplex exaratus	ploughed triton
Monoplex pilearis	northern hairy triton
Monoplex thersites	-
Montfortista excentrica	-
Montfortula pulchra	-
Montfortula rugosa	rough notch limpet
Morula (Habromorula) spinosa	-
Murex (Murex) acanthostephes	murex shell
Murex (Murex) pecten	-
Naria erosa	-
Naria helvola	honey cowry
Nassarius (Alectrion) glans	acorn dog whelk
Nassarius (Niotha) albescens	whitish dog whelk
Nassarius (Niotha) albinus	-
Nassarius (Zeuxis) clarus	-
Nassarius horridus	-
Natica schepmani	-
Natica vitellus	egg yolk sand snail
Nerita (Argonerita) chamaeleon	chamaeleon nerite
Nerita (Cymostyla) undata	wavy nerite
Nerita (Ritena) plicata	plicate nerite
Nerita (Theliostyla) albicilla	tubercular nerite

Binomial	Common Name
Neverita powisianus	chestnut-banded sand snail
Nevia spirata	spirate cross-barred shell
Notarchus indicus	-
Notocochlis gualtieriana	-
Oliva brettinghami	-
Onustus indicus	-
Palmadusta clandestina	-
Patelloida mimula	-
Patelloida saccharina	northern star limpet
Peristernia reincarnata	-
Phasianella solida	-
Phasianella variegata	variegated pheasant
Philine cf. aperta	-
Phos (Phos) senticosus	Pacific phos
Pinaxia versicolor	varicoloured thaid

Binomial	Common Name
Pirenella austrocingulata	-
Pirenella rugosa	-
Pisania (Pisania) ignea	flame pisania
Planaxis sulcatus	ribbed clusterwink
Pleurobranchaea maculata	-
Pleurobranchus grandis	-
Pleurobranchus peronii	-
Pollia undosa	waved buccinum
Profundiconus teramachii	-
Prothalotia baudini	Baudin's top shell
Prothalotia strigata	-
Pseudostomatella papyracea	-
Pseudovertagus (Pseudovertagus) aluco	Cuming's creeper
Pterochelus acanthopterus	murex shell
Pterochelus akation	-
Ptychobela nodulosa	-
Pupa solidula	-
Pupoides contrarius	Abrolhos sinistral pupasnail
Purpuradusta fimbriata	-
Purpuradusta gracilis	-
Purpuradusta hammondae	-
Pyramidella acus	-
Pyramidella dolabrata	-
Pyramidella sulcatus	-
Pyrene flava	yellow dove
Pyrene punctata	-
Quistrachia legendrei	-
Quistrachia montebelloensis	-
Ranularia cynocephalum	dog's-head triton
Rapa rapa	soft coral shell
Reticunassa paupera	poor dog whelk
Rhagada angulata	-
Rhagada convicta	-
Rhagada elachystoma	-
Rhinoclavis (Proclava) kochi	-
Rhinoclavis (Rhinoclavis) articulata	creeper
Rhinoclavis (Rhinoclavis) brettinghami	beautiful creeper
Rhinoclavis (Rhinoclavis) fasciata	banded creeper
Rhinoclavis (Rhinoclavis) vertagus	ribbed cerith
Rissoina (Phosinella) media	-

Binomial	Common Name
Rissoina (Rissoina) ambigua	-
Rissoina (Rissoina) crassa	-
Rissosyrnola aclis	-
Sagaminopteron ornatum	-
Sagaminopteron psychedelicum	-
Scabricola (Scabricola) barrywilsoni	-
Scalptia textilis	-
Scutellastra flexuosa	-
Scutus (Scutus) unguis	northern duck's bill
Sericominolia vernicosa	-
Siphonaria kurracheensis	-
Siphonaria zelandica	air-breathing limpet
Smaragdia (Smaragdella) souverbiana	beautiful neritina
Smaragdinella calyculata	-
Staphylaea limacina	-
Stomatella impertusa	false ear shell
Stomatia phymotis	keeled wide-mouthed shell
Stomatia rubra	-
Strigatella scutulata	banded black mitre
Surrepifungium costulata	-
Talopena vernicosa	-
Tanea euzona	painted sand snail
Tectonatica robillardi	-
Tectus (Tectus) fenestratus	latticed top shell
Tectus (Tectus) pyramis	pyramid trochus
Tenagodus ponderosus	ponderous worm shell

Binomial	Common Name
Tenguella granulata	granulated drupe
Terebellum terebellum	bullet stromb
Terebra amanda	-
Terebralia semistriata	striate mud creeper
Thalessa virgata	prickly thaid
Thuridilla indopacifica	-
Tonna canaliculata	-
Tonna perdix	partridge tun
Tonna variegata	variegated tun
Tricolia variabilis	minute pheasant
Tripterotyphis lowei	-
Trivirostra edgari	-
Trochus hanleyanus	Hanley's trochus
Trochus histrio	-
tubulophilinopsis gardineri	

Binomial	Common Name
Tudivasum inerme	unarmed whelk
Turbo (Carswellena) haynesi	Hayne's turban
Turbo (Marmarostoma) argyrostomus	scaly turban
Turbo (Marmarostoma) bruneus	little burnt turbo
Turbo (Marmarostoma) squamosus	squamose turban
Turbo (Turbo) petholatus	cat's eye turban
Turcica maculata	-
Turricula nelliae	-
Turris crispa	-
Vanitrochus tragema	-
Variegemarginula variegata	-
Vokesimurex multiplicatus	-
Volutoconus hargreavesi	-
Xenophora (Xenophora) cerea	-
Xenophora (Xenophora) solarioides	-
Xenuroturris millepunctata	-
Gobiidae	
***Amblyeleotris* sp.**	
Amblyeleotris diagonalis	diagonal shrimpgoby
Amblyeleotris gymnocephalus	mask shrimpgoby
Amblyeleotris periophthalmus	broadbanded shrimpgoby
Amblyeleotris wheeleri	burgundy shrimpgoby
***Eviota* sp.**	
Eviota bimaculata	twospot eviota
Eviota distigma	distigma eviota
Eviota guttata	whitelined eviota
Eviota inutilis	chestspot eviota
Eviota melasma	headspot eviota
Eviota nebulosa	palespot eviota
Eviota prasina	rubble eviota
Eviota prasites	hairfin eviota
Eviota queenslandica	Queensland eviota
Eviota sebreei	striped eviota
Eviota sigillata	sign eviota
Eviota storthynx	rosy eviota
Eviota zebrina	zebra eviota
***Valenciennea* sp.**	
Valenciennea alleni	Allen's glidergoby
Valenciennea helsdingenii	blacklined glidergoby
Valenciennea longipinnis	ocellate glidergoby

Binomial	Common Name
Valenciennea muralis	mural glidergoby
Valenciennea puellaris	orangespotted glidergoby
Valenciennea wardii	broadbarred glidergoby
Gobiidae sp.	
Amblygobius bynoensis	bynoe goby
Amblygobius decussatus	crosshatch goby
Amblygobius nocturnus	pyjama goby
Amblygobius phalaena	whitebarred goby
Amoya gracilis	bluespotted mangrovegoby
Asterropteryx semipunctata	starry goby
Barbuligobius boehlkei	cryptic bearded goby
Bathygobius cocosensis	cocos frillgoby
Bathygobius fuscus	dusky frillgoby
Bathygobius laddi	Ladd's frillgoby
Bryaninops amplus	large whipgoby
Bryaninops loki	loki whipgoby
Bryaninops yongei	seawhip goby
Callogobius maculipinnis	ostrich goby
Callogobius sclateri	tripleband goby
Cryptocentrus caeruleomaculatus	bluespotted shrimpgoby
Cryptocentrus cinctus	yellow shrimpgoby
Cryptocentrus fasciatus	y-bar shrimpgoby
Ctenogobiops pomastictus	goldspeckled shrimpgoby
Favonigobius melanobranchus	blackthroat goby
Fusigobius duospilus	twospot sandgoby
Fusigobius neophytus	neophyte sandgoby
Fusigobius signipinnis	flasher sandgoby
Gnatholepis argus	peacock sandgoby

Binomial	Common Name
Gnatholepis cauerensis	eye-bar sand-goby
Gobiodon axillaris	red-striped coralgoby
Gobiodon citrinus	lemon coralgoby
Gobiodon erythrospilus	blue-spotted coral-goby
Gobiodon histrio	Māori coralgoby
Gobiodon quinquestrigatus	fiveline coralgoby
Gobiodon rivulatus	rippled coralgoby
Gobiopsis angustifrons	narrow barbelgoby
Hazeus diacanthus	twospine sandgoby
Hazeus elati	eilat sandgoby
Istigobius decoratus	decorated sandgoby
Istigobius goldmanni	Goldmann's sandgoby
Istigobius nigroocellatus	blackspotted sandgoby
Istigobius ornatus	ornate sandgoby
Istigobius rigilius	orangespotted sandgoby

Binomial	Common Name
Larsonella pumilus	dwarf slippery goby
Lobulogobius omanensis	giant lobegoby
Lubricogobius ornatus	ornate slippery goby
Macrodontogobius wilburi	Wilbur's goby
Pandaka lidwilli	Lidwill's dwarfgoby
Paragobiodon echinocephalus	redhead stylophora goby
Paragobiodon lacunicola	blackfin coralgoby
Paragobiodon melanosoma	black coralgoby
Paragobiodon xanthosoma	emerald coralgoby
Periophthalmus argentilineatus	silverlined mudskipper
Pleurosicya annandalei	solenocaulon ghostgoby
Pleurosicya boldinghi	softcoral ghostgoby
Pleurosicya elongata	slender spongegoby
Pleurosicya mossambica	many-host ghostgoby
Pleurosicya plicata	lobed ghostgoby
Priolepis cincta	girdled reefgoby
Priolepis nuchifasciata	threadfin reefgoby
Priolepis profunda	orange convict reefgoby
Priolepis semidoliata	halfbarred reefgoby
Sueviota atrinasa	blacknose sueviota
Sueviota larsonae	Larson's sueviota
Tasmanogobius gloveri	Glover's tasmangoby
Trimma nomurai	Nomura's dwarfgoby
Trimma okinawae	orange-red pygmygoby
Haemulidae	
Haemulidae sp.	
Diagramma labiosum	painted sweetlips
Plectorhinchus caeruleonothus	blue bastard

Binomial	Common Name
Plectorhinchus chaetodonoides	spotted sweetlips
Plectorhinchus flavomaculatus	goldspotted sweetlips
Plectorhinchus gibbosus	brown sweetlips
Plectorhinchus lineatus	oblique-banded sweetlips
Plectorhinchus multivittatus	manyline sweetlips
Plectorhinchus pica	dotted sweetlips
Plectorhinchus polytaenia	ribbon sweetlips
Plectorhinchus unicolor	sombre sweetlips
Plectorhinchus vittatus	oriental sweetlips
Pomadasys argenteus	silver javelin
Pomadasys kaakan	barred javelin
Pomadasys maculatus	blotched javelin
Hemigaleidae	
Hemigaleidae sp.	
Hemigaleus australiensis	weasel shark

Binomial	Common Name
Hemipristis elongata	fossil shark
Istiophoridae	
Istiophoridae sp.	
Istiompax indica	black marlin
Istiophorus platypterus	sailfish
Kajikia audax	striped marlin
Makaira nigricans	blue marlin
Tetrapturus angustirostris	shortbill spearfish
Labridae	
***Bodianus* sp.**	
Bodianus axillaris	coral pigfish
Bodianus bilunulatus	saddleback pigfish
Bodianus mesothorax	eclipse pigfish
Bodianus perditio	goldspot pigfish
Bodianus solatus	sunburnt pigfish
***Cirrhilabrus* sp.**	
Cirrhilabrus cyanopleura	blueside wrasse
Cirrhilabrus temminckii	peacock wrasse
***Coris*sp.**	
Coris aygula	redblotched wrasse
Coris caudimacula	spot-tail wrasse
Coris dorsomacula	pinklined wrasse
Coris pictoides	pixy wrasse
***Suezichthys* sp.**	
Suezichthys cyanolaemus	bluethroat rainbow wrasse
Suezichthys devisi	Australian rainbow wrasse
Suezichthys soelae	soela wrasse
Labridae sp.	

Binomial	Common Name
Achoerodus gouldii	western blue groper
Anampses caeruleopunctatus	diamond wrasse
Anampses geographicus	scribbled wrasse
Anampses lennardi	blue-and-yellow wrasse
Anampses melanurus	blacktail wrasse
Calotomus carolinus	star-eye parrotfish
Calotomus spinidens	spinytooth parrotfish
Cheilinus chlorourus	floral Māori wrasse
Cheilinus trilobatus	tripletail Māori wrasse
Cheilio inermis	sharpnose wrasse
Choerodon anchorago	anchor tuskfish
Choerodon cauteroma	bluespotted tuskfish
Choerodon cephalotes	purple tuskfish
Choerodon cyanodus	blue tuskfish
Choerodon jordani	dagger tuskfish

Binomial	Common Name
Choerodon monostigma	darkspot tuskfish
Choerodon schoenleinii	blackspot tuskfish
Choerodon sugillatum	wedgetail tuskfish
Choerodon vitta	redstripe tuskfish
Choerodon zamboangae	eyebrow tuskfish
Epibulus insidiator	slingjaw wrasse
Gomphosus caeruleus	Indian bird wrasse
Gomphosus varius	birdnose wrasse
Halichoeres biocellatus	false-eyed wrasse
Halichoeres chloropterus	pastel-green wrasse
Halichoeres hartzfeldii	orangeline wrasse
Halichoeres margaritaceus	pearly wrasse
Halichoeres marginatus	dusky wrasse
Halichoeres melanochir	orangefin wrasse
Halichoeres melanurus	Hoeven's wrasse
Halichoeres nebulosus	cloud wrasse
Halichoeres nigrescens	bubblefin wrasse
Halichoeres trimaculatus	threespot wrasse
Hemigymnus fasciatus	fiveband wrasse
Hemigymnus melapterus	thicklip wrasse
Hologymnosus annulatus	ringed slender wrasse
Hologymnosus doliatus	pastel slender wrasse
Hologymnosus rhodonotus	red slender wrasse
Iniistius dea	leaf wrasse
Iniistius jacksonensis	keelhead razorfish
Iniistius pavo	blue razorfish
Labrichthys unilineatus	oneline wrasse
Labroides bicolor	bicolor cleanerfish

Binomial	Common Name
Labroides dimidiatus	common cleanerfish
Leptojulis cyanopleura	shoulderspot wrasse
Leptoscarus vaigiensis	marbled parrotfish
Macropharyngodon meleagris	leopard wrasse
Macropharyngodon negrosensis	black leopard wrasse
Macropharyngodon ornatus	ornate leopard wrasse
Oxycheilinus bimaculatus	little Māori wrasse
Oxycheilinus digramma	violetline Māori wrasse
Oxycheilinus orientalis	oriental Māori wrasse
Pseudocheilinus evanidus	pinstripe wrasse
Pseudodax moluccanus	chiseltooth wrasse
Pteragogus cryptus	cryptic wrasse
Pteragogus enneacanthus	cockerel wrasse
Pteragogus flagellifer	cocktail wrasse
Stethojulis bandanensis	redspot wrasse

Binomial	Common Name
Stethojulis interrupta	brokenline wrasse
Stethojulis strigiventer	silverstreak wrasse
Stethojulis trilineata	three-ribbon wrasse
Thalassoma amblycephalus	bluehead wrasse
Thalassoma hardwicke	sixbar wrasse
Thalassoma lunare	moon wrasse
Thalassoma lutescens	green moon wrasse
Thalassoma purpureum	surge wrasse
Xenojulis margaritacea	pinkspeckled wrasse
Lethrinidae	
***Gymnocranius* sp.**	
Gymnocranius elongatus	swallowtail seabream
Gymnocranius euanus	paddletail seabream
Gymnocranius grandoculis	Robinson's seabream
Gymnocranius griseus	grey seabream
Gymnocranius microdon	bluespotted seabream
***Lethrinus* sp.**	
Lethrinus amboinensis	Ambon emperor
Lethrinus atkinsoni	yellowtail emperor
Lethrinus erythracanthus	orangespotted emperor
Lethrinus erythropterus	longfin emperor
Lethrinus genivittatus	threadfin emperor
Lethrinus harak	thumbprint emperor
Lethrinus laticaudis	grass emperor
Lethrinus lentjan	redspot emperor
Lethrinus microdon	smalltooth emperor
Lethrinus miniatus	redthroat emperor
Lethrinus nebulosus	spangled emperor
Lethrinus olivaceus	longnose emperor
Lethrinus ornatus	ornate emperor
Lethrinus punctulatus	bluespotted emperor
Lethrinus ravus	drab emperor
Lethrinus rubrioperculatus	spotcheek emperor
Lethrinus semicinctus	blackblotch emperor
Lethrinus variegatus	variegated emperor
Lethrinidae sp.	
Gnathodentex aureolineatus	goldspot seabream
Monotaxis grandoculis	bigeye seabream
Lutjanidae	
***Lutjanus* sp.**	
Lutjanus adetii	hussar
Lutjanus argentimaculatus	mangrove jack
Lutjanus bitaeniatus	Indonesian snapper

Binomial	Common Name
Lutjanus bohar	red bass
Lutjanus carponotatus	stripey snapper
Lutjanus decussatus	checkered snapper
Lutjanus erythropterus	crimson snapper
Lutjanus fulviflamma	blackspot snapper
Lutjanus fulvus	blacktail snapper
Lutjanus johnii	golden snapper
Lutjanus kasmira	bluestriped snapper
Lutjanus lemniscatus	darktail snapper
Lutjanus lutjanus	bigeye snapper
Lutjanus malabaricus	saddletail snapper
Lutjanus monostigma	onespot snapper
Lutjanus quinquelineatus	fiveline snapper
Lutjanus russellii	Moses' snapper
Lutjanus sebae	red emperor
Lutjanus vitta	brownstripe snapper
Microdesmidae	
***Ptereleotris* sp.**	
Ptereleotris evides	arrow dartgoby
Ptereleotris hanae	thread-tail dartgoby
Ptereleotris microlepis	greeneye dartgoby
Ptereleotris monoptera	lyretail dartgoby
Microdesmidae sp.	
Gunnellichthys curiosus	curious wormfish
Parioglossus formosus	yellowstriped dartfish
Monacanthidae	
***Aluterus* sp.**	
Aluterus monoceros	unicorn leatherjacket
Aluterus scriptus	scrawled leatherjacket
Monacanthidae sp.	
Anacanthus barbatus	bearded leatherjacket
Brachaluteres taylori	Taylor's pygmy leatherjacket
Cantherhines dumerilii	barred leatherjacket
Cantherhines fronticinctus	spectacled leatherjacket
Cantherhines pardalis	honeycomb leatherjacket
Chaetodermis penicilligerus	tasselled leatherjacket
Colurodontis paxmani	Paxman's leatherjacket
Eubalichthys caeruleoguttatus	bluespotted leatherjacket
Eubalichthys mosaicus	mosaic leatherjacket
Monacanthus chinensis	fanbelly leatherjacket
Oxymonacanthus longirostris	harlequin filefish
Paraluteres prionurus	blacksaddle filefish

Binomial	Common Name
Paramonacanthus choirocephalus	pigface leatherjacket
Paramonacanthus filicauda	threadfin leatherjacket
Paramonacanthus oblongus	Japanese leatherjacket
Paramonacanthus pusillus	Sinhalese leatherjacket
Pervagor janthinosoma	gillblotch leatherjacket
Pseudomonacanthus elongatus	fourband leatherjacket
Thamnaconus hypargyreus	yellowspotted leatherjacket
Mullidae	
***Parupeneus* sp.**	
Parupeneus barberinoides	bicolour goatfish
Parupeneus chrysopleuron	rosy goatfish
Parupeneus ciliatus	diamondscale goatfish
Parupeneus cyclostomus	goldsaddle goatfish
Parupeneus heptacantha	opalescent goatfish
Parupeneus indicus	yellowspot goatfish
Parupeneus multifasciatus	banded goatfish
Parupeneus pleurostigma	sidespot goatfish
Parupeneus spilurus	blacksaddle goatfish
Mullidae sp.	
Mulloidichthys flavolineatus	yellowstripe goatfish
Muraenidae	
***Gymnothorax* sp.**	
Gymnothorax buroensis	latticetail moray
Gymnothorax cephalospilus	headspot moray
Gymnothorax cribroris	sieve moray
Gymnothorax eurostus	stout moray
Gymnothorax fimbriatus	fimbriate moray
Gymnothorax flavimarginatus	yellowmargin moray
Gymnothorax javanicus	giant moray
Gymnothorax longinquus	long moray
Gymnothorax mccoskeri	manyband moray
Gymnothorax melatremus	dwarf moray
Gymnothorax minor	lesser moray
Gymnothorax mucifer	kidako moray
Gymnothorax prasinus	green moray
Gymnothorax pseudothyrsoideus	highfin moray
Gymnothorax thyrsoideus	greyface moray
Gymnothorax undulatus	undulate moray
Muraenidae sp.	
Echidna nebulosa	starry moray
Uropterygius marmoratus	marbled snake moray
Myliobatidae	

Binomial	Common Name
***Mobula* sp.**	
Aetobatus ocellatus	whitespotted eagle ray
Mobula alfredi	reef manta
Mobula birostris	giant manta
Mobula kuhlii	shortfin devilray
Mobula thurstoni	bentfin devilray
Nemipteridae	
***Nemipterus* sp.**	
Nemipterus bathybius	yellowbelly threadfin bream
Nemipterus celebicus	celebes threadfin bream
Nemipterus furcosus	rosy threadfin bream
Nemipterus nematopus	yellowtip threadfin bream
Nemipterus peronii	notched threadfin bream
Nemipterus tambuloides	fiveline threadfin bream
Nemipterus virgatus	golden threadfin bream
Nemipterus zysron	slender threadfin bream
***Pentapodus* sp.**	
Pentapodus emeryii	purple threadfin bream
Pentapodus nagasakiensis	Japanese threadfin bream
Pentapodus paradiseus	paradise threadfin bream
Pentapodus porosus	northwest threadfin bream
Pentapodus vitta	western butterfish
***Scolopsis* sp.**	
Scolopsis affinis	bridled monocle bream
Scolopsis bilineata	two-line monocle bream
Scolopsis lineata	lined monocle bream
Scolopsis meridiana	redspot monocle bream
Scolopsis monogramma	rainbow monocle bream

Binomial	Common Name
Scolopsis taenioptera	lattice monocle bream
Scolopsis taeniopterus	redspot monocle bream
Scolopsis xenochrous	oblique-bar monocle bream
Nemipteridae sp.	
Parascolopsis inermis	redbelt monocle bream
Parascolopsis rufomaculata	yellowband monocle bream
Parascolopsis tanyactis	longray monocle bream
Scaevius milii	coral monocle bream
Nomeidae	
***Psenes* sp.**	
Psenes arafurensis	banded driftfish
Psenes cyanophrys	freckled driftfish
Psenes pellucidus	bluefin driftfish
Nomeidae sp.	
Cubiceps baxteri	black fathead

Binomial	Common Name
Cubiceps capensis	cape fathead
Cubiceps kotlyari	Kotlyar's cubehead
Cubiceps pauciradiatus	bigeye cigarfish
Cubiceps whiteleggii	shadow driftfish
Nomeus gronovii	man-of-war fish
Octopoda	
***Octopus* sp.**	
Octopus cyanea	day octopus
Octopus superciliosus	frilled pygmy octopus
Octopoda sp.	
Ameloctopus litoralis	banded stringarm octopus
Amphioctopus exannulatus	plain-spot octopus
Amphioctopus marginatus	veined octopus
Callistoctopus dierythraeus	red-spot night octopus
Hapalochlaena lunulata	greater blue-ringed octopus
Hapalochlaena maculosa	southern blue-ringed octopus
Ophichtidae	
Ophichthidae sp.	
Callechelys catostoma	blackstriped snake eel
Callechelys marmorata	marbled snake eel
Leiuranus semicinctus	saddled snake eel
Ophichthus altipennis	blackfin snake eel
Ophichthus rutidoderma	olive snake eel
Ophisurus serpens	serpent eel
Phyllophichthus xenodontus	flappy snake eel
Pisodonophis cancrivorus	burrowing snake eel
Scolecenchelys gymnota	slender worm eel
Scolecenchelys macroptera	narrow worm eel

Binomial	Common Name
Ophiuroidea	
Ophiuroidea sp.	
Amphioplus (Lymanella) depressus	-
Amphipholis squamata	brooding brittle star
Amphiura (Amphiura) bidentata	-
Amphiura (Amphiura) duncani	-
Amphiura (Amphiura) leucaspis	-
Amphiura (Amphiura) maxima	-
Amphiura (Amphiura) microsoma	-
Amphiura (Amphiura) velox	-
Amphiura (Fellaria) octacantha	-
Dictenophiura stellata	-
Macrophiothrix belli	-
Macrophiothrix caenosa	-

Binomial	Common Name
Macrophiothrix callizona	-
Macrophiothrix koehleri	-
Macrophiothrix lineocaerulea	-
Macrophiothrix longipeda	-
Macrophiothrix megapoma	-
Macrophiothrix paucispina	-
Macrophiothrix variabilis	-
Ophiacantha indica	-
Ophiactis brevis	-
Ophiactis fuscolineata	-
Ophiactis luteomaculata	-
Ophiactis macrolepidota	-
Ophiactis modesta	-
Ophiactis savignyi	-
Ophiarachnella gorgonia	-
Ophiarachnella infernalis	-
Ophiocentrus dilatatus	-
Ophiochaeta hirsuta	-
Ophiochasma stellata	-
Ophiocnemis marmorata	-
Ophiocoma dentata	-
Ophiocomella sexradia	-
Ophioconis cincta	-
Ophiodaphne formata	-
Ophiodyscrita acosmeta	-
Ophiogymna pulchella	-
Ophiolepis cincta	-
Ophiolepis unicolor	-

Binomial	Common Name
Ophiomastix mixta	-
Ophiomastix variabilis	-
Ophiomaza cacaotica	-
Ophionereis dubia	-
Ophionereis semoni	-
Ophioplocus imbricatus	-
Ophiopsammus yoldii	-
Ophiopteron elegans	-
Ophiothela danae	-
Ophiothrix (Keystonea) martensi	-
Ophiothrix (Keystonea) smaragdina	-
Ophiothrix (Ophiothrix) ciliaris	-
Ophiothrix (Ophiothrix) exigua	-
Ophiothrix (Ophiothrix) foveolata	-

Binomial	Common Name
Ophiothrix (Ophiothrix) plana	-
Ophiothrix (Placophiothrix) lineocaerulea	-
Ophiothrix (Placophiothrix) melanosticta	-
Ostraciidae	
Ostraciidae sp.	
Lactoria cornuta	longhorn cowfish
Lactoria diaphana	roundbelly cowfish
Lactoria fornasini	thornback cowfish
Ostracion cubicus	yellow boxfish
Ostracion meleagris	black boxfish
Ostracion nasus	shortnose boxfish
Ostracion rhinorhynchos	horn-nose boxfish
Rhynchostracion nasus	shortnose boxfish
Tetrosomus gibbosus	humpback turretfish
Tetrosomus reipublicae	smallspine turretfish
Paguridae	
Paguridae sp.	
Pylopaguropsis zebra	-
Spiropagurus fimbriatus	-
Palinuridae	
Palinuridae sp.	
Panulirus ornatus	ornate spiny lobster
Panulirus versicolor	painted spiny lobster
Pinguipedidae	
***Parapercis* sp.**	
Parapercis alboguttata	bluenose grubfish
Parapercis clathrata	spothead grubfish
Parapercis haackei	wavy grubfish
Parapercis multiplicata	doublestitch grubfish
Parapercis nebulosa	pinkbanded grubfish
Parapercis rubricaudalis	redtail sandperch
Parapercis rubromaculata	redspot sandperch
Parapercis snyderi	Snyder's grubfish
Parapercis xanthozona	peppered grubfish
Pinguipedidae sp.	
Ryukyupercis gushikeni	rosy grubfish
Polycheata	
Polycheata sp.	
Ceratonereis australis	-
Ceratonereis mirabilis	-
Ceratonereis singularis	-

Binomial	Common Name
Diopatra amboinensis	-
Diopatra gigova	-
Diopatra maculata	-
Eunice afra	-
Eunice antennata	-
Eurythoe complanata	-
Harmothoe dictyophora	-
Hololepidella nigropunctata	-
Iphione muricata	-
Iphione ovata	-
Leonnates indicus	-
Leonnates stephensoni	-
Lepidonotus carinulatus	-
Lysidice ninetta	-
Marphysa bifurcata	-
Neanthes cricognatha	-
Neanthes dawydovi	-
Neanthes unifasciata	-
Nereis bifida	-
Nereis denhamensis	-
Nereis heirissonensis	-
Onuphis holobranchiata	-
Palola siciliensis	-
Perinereis amblyodonta	-
Perinereis helleri	-
Perinereis nigropunctata	-
Perinereis obfuscata	-
Perinereis suluana	-

Binomial	Common Name
Perinereis vancaurica	-
Platynereis antipoda	-
Platynereis polyscalma	-
Platynereis uniseris	-
Pseudonereis anomala	-
Pseudonereis trimaculata	-
Pomacanthidae	
***chaetodontoplus* sp.**	
Chaetodontoplus duboulayi	scribbled angelfish
Chaetodontoplus mesoleucus	vermiculate angelfish
Chaetodontoplus personifer	yellowtail angelfish
***Pomacanthus* sp.**	
Pomacanthus imperator	emperor angelfish
Pomacanthus semicirculatus	blue angelfish
Pomacanthus sexstriatus	sixband angelfish

Binomial	Common Name
Pomacanthidae sp.	
Apolemichthys trimaculatus	threespot angelfish
Centropyge tibicen	keyhole angelfish
Pomacentridae	
***Chromis* sp.**	
Chromis atripectoralis	blackaxil puller
Chromis chrysura	stoutbody puller
Chromis cinerascens	green puller
Chromis fumea	smoky puller
Chromis margaritifer	whitetail puller
Chromis opercularis	doublebar chromis
Chromis viridis	blue-green puller
Chromis weberi	Weber's puller
Chromis westaustralis	West Australian puller
Pomacentridae sp.	
Abudefduf bengalensis	bengal sergeant
Abudefduf septemfasciatus	banded sergeant
Abudefduf sexfasciatus	scissortail sergeant
Abudefduf sordidus	blackspot sergeant
Abudefduf vaigiensis	Indo-Pacific sergeant
Amblyglyphidodon curacao	staghorn damsel
Amblyglyphidodon ternatensis	ternate damselfish
Amblypomacentrus breviceps	blackbanded damsel
Amphiprion clarkii	Clark's anemonefish
Amphiprion perideraion	pink anemonefish
Amphiprion rubrocinctus	Australian anemonefish
Cheiloprion labiatus	biglip damsel
Chrysiptera cyanea	blue demoiselle
Chrysiptera tricincta	threeband damselfish
Dascyllus aruanus	banded humbug
Dascyllus reticulatus	headband humbug
Dascyllus trimaculatus	threespot humbug
Dischistodus darwiniensis	banded damsel
Dischistodus perspicillatus	white damsel
Dischistodus prosopotaenia	honeyhead damsel
Hemiglyphidodon plagiometopon	lagoon damsel
Neoglyphidodon melas	black damsel
Neoglyphidodon nigroris	scarface damsel
Neopomacentrus azysron	yellowtail demoiselle
Neopomacentrus cyanomos	regal demoiselle
Neopomacentrus filamentosus	brown demoiselle
Neopomacentrus taeniurus	freshwater demoiselle
Plectroglyphidodon dickii	Dick's damsel

Binomial	Common Name
Plectroglyphidodon johnstonianus	Johnston damsel
Plectroglyphidodon lacrymatus	jewel damsel
Plectroglyphidodon leucozona	whiteband damsel
Pomacentrus alexanderae	alexander's damsel
Pomacentrus amboinensis	Ambon damsel
Pomacentrus coelestis	neon damsel
Pomacentrus limosus	muddy damsel
Pomacentrus milleri	Miller's damsel
Pomacentrus moluccensis	lemon damsel
Pomacentrus nagasakiensis	blue-scribbled damsel
Pomacentrus nigromanus	goldback damsel
Pomacentrus pavo	peacock damsel
Pomacentrus vaiuli	princess damsel
Pristotis obtusirostris	gulf damsel
Stegastes apicalis	yellowtip gregory
Stegastes fasciolatus	pacific gregory
Stegastes nigricans	dusky gregory
Stegastes obreptus	western gregory
Stegastes punctatus	bluntsnout gregory
Priacanthidae	
***Priacanthus* sp.**	
Priacanthus blochii	glasseye
Priacanthus hamrur	lunartail bigeye
Priacanthus macracanthus	spotted bigeye
Priacanthus tayenus	purplespotted bigeye
Pseudochromidae	
***Congrogadus* sp.**	
Congrogadus spinifer	spiny eel blenny
Congrogadus subducens	carpet eel blenny
***Pseudochromis* sp.**	
Pseudochromis fuscus	dusky dottyback
Pseudochromis howsoni	Howson's dottyback
Pseudochromis marshallensis	marshall dottyback
Pseudochromis quinquedentatus	spotted dottyback
Pseudochromis reticulatus	reticulate dottyback
Pseudochromis wilsoni	yellowfin dottyback
Rhinidae	
***Rhynchobatus* sp.**	
Rhynchobatus australiae	bottlenose wedgefish
Rhynchobatus palpebratus	eyebrow wedgefish
Rhynhcobatus laevis	smoothnose wedgefish
Rhinidae sp.	

Binomial	Common Name
Rhina ancylostoma	bowmouth guitarfish
Scaridae	
Chlorurus sp.	
Chlorurus bleekeri	Bleeker's parrotfish
Chlorurus capistratoides	pink-margined parrotfish
Chlorurus microrhinos	steephead parrotfish
Chlorurus oedema	knothead parrotfish
Chlorurus rhakoura	raggedfin parrotfish
Chlorurus sordidus	greenfin parrotfish
Scarus sp.	
Scarus chameleon	chameleon parrotfish
Scarus dimidiatus	bluebridle parrotfish
Scarus flavipectoralis	yellowfin parrotfish
Scarus forsteni	whitespot parrotfish
Scarus frenatus	sixband parrotfish
Scarus ghobban	bluebarred parrotfish
Scarus globiceps	violetline parrotfish
Scarus niger	swarthy parrotfish
Scarus oviceps	darkcap parrotfish
Scarus prasiognathos	greencheek parrotfish
Scarus psittacus	palenose parrotfish
Scarus rivulatus	surf parrotfish
Scarus rubroviolaceus	blackvein parrotfish
Scarus schlegeli	Schlegel's parrotfish
Scaridae sp.	
Hipposcarus longiceps	longnose parrotfish
Scombridae	
***Sarda* sp.**	

Binomial	Common Name
Sarda australis	Australian bonito
Sarda orientalis	striped bonito
***Scomberomorus* sp.**	
Scomberomorus commerson	Spanish mackerel
Scomberomorus munroi	spotted mackerel
Scomberomorus queenslandicus	school mackerel
Scombridae sp.	
Acanthocybium solandri	wahoo
Auxis thazard	frigate tuna
Cybiosarda elegans	leaping bonito
Euthynnus affinis	mackerel tuna
Grammatorcynus bicarinatus	shark mackerel
Grammatorcynus bilineatus	scad mackerel
Gymnosarda unicolor	dogtooth tuna
Katsuwonus pelamis	skipjack tuna

Binomial	Common Name
Rastrelliger kanagurta	mouth mackerel
Thunnus orientalis	northern bluefin tuna
Scyphozoa	
Scyphozoa sp.	
Aurelia aurita	moon jellyfish
Catostylus mosaicus	blue blubber
Cephea cephea	-
Chrysaora kynthia	-
Chrysaora pentastoma	-
Crambione mastigophora	-
Cyanea annaskala	-
Cyanea buitendijki	-
Cyanea mjobergi	-
Pelagia noctiluca	mauve stinger
Phyllorhiza pacifica	-
Phyllorhiza punctata	brown jellyfish
Sepiidae	
***Sepia* sp.**	
Sepia elliptica	ovalbone cuttlefish
Sepia latimanus	broadclub cuttlefish
Sepia papuensis	papuan cuttlefish
Sepia pharaonis	pharaoh cuttlefish
Sepia smithi	smith's cuttlefish
Serranidae	
***Cephalopholis* sp.**	
Cephalopholis argus	peacock rockcod
Cephalopholis boenak	brownbarred rockcod
Cephalopholis cyanostigma	bluespotted rockcod
Cephalopholis miniata	coral rockcod
Cephalopholis sonnerati	tomato rockcod
Cephalopholis urodeta	flagtail rockcod
***Epinephelus* sp.**	
Epinephelus amblycephalus	banded grouper
Epinephelus areolatus	yellowspotted rockcod
Epinephelus bilobatus	frostback rockcod
Epinephelus bleekeri	duskytail grouper
Epinephelus chlorostigma	brownspotted grouper
Epinephelus coeruleopunctatus	whitespotted grouper
Epinephelus coioides	goldspotted rockcod
Epinephelus corallicola	coral grouper
Epinephelus fasciatus	blacktip rockcod
Epinephelus fuscoguttatus	flowery rockcod
Epinephelus lanceolatus	Queensland groper

Binomial	Common Name
Epinephelus latifasciatus	striped grouper
Epinephelus macrospilos	snubnose grouper
Epinephelus maculatus	highfin grouper
Epinephelus malabaricus	blackspotted rockcod
Epinephelus merra	birdwire rockcod
Epinephelus multinotatus	Rankin cod
Epinephelus ongus	specklefin grouper
Epinephelus polyphekadion	camouflage grouper
Epinephelus quoyanus	longfin rockcod
Epinephelus rivulatus	chinaman rockcod
Epinephelus sexfasciatus	sixbar grouper
Epinephelus tauvina	greasy rockcod
Epinephelus tukula	potato rockcod
***Plectropomus* sp.**	
Plectropomus areolatus	passionfruit coral trout
Plectropomus laevis	bluespotted coral trout
Plectropomus leopardus	common coral trout
Plectropomus maculatus	barcheek coral trout
Serranidae sp.	
Anyperodon leucogrammicus	whitelined rockcod
Caprodon longimanus	longfin perch
Caprodon schlegelii	sunrise perch
Chromileptes altivelis	barramundi cod
Diploprion bifasciatum	barred soapfish
Pseudanthias rubrizonatus	lilac-tip basslet
Pseudogramma polyacanthus	honeycomb podge
Rainfordia opercularis	rainfordia
Triso dermopterus	oval rockcod

Binomial	Common Name
Variola albimarginata	white-edge coronation trout
Variola louti	yellowedge coronation trout
Siganidae	
***Siganus* sp.**	
Siganus argenteus	forktail rabbitfish
Siganus canaliculatus	whitespotted rabbitfish
Siganus corallinus	coral rabbitfish
Siganus doliatus	bluelined rabbitfish
Siganus fuscescens	black rabbitfish
Siganus javus	Java rabbitfish
Siganus lineatus	goldlined rabbitfish
Siganus punctatissimus	finespotted rabbitfish
Siganus punctatus	spotted rabbitfish
Siganus trispilos	threespot rabbitfish
Siganus virgatus	doublebar rabbitfish

Binomial	Common Name
Siganus vulpinus	foxface
Sphyraenidae	
***Sphyraena* sp.**	
Sphyraena acutipinnis	sharpfin barracuda
Sphyraena barracuda	great barracuda
Sphyraena forsteri	blackspot barracuda
Sphyraena helleri	Heller's barracuda
Sphyraena jello	pickhandle barracuda
Sphyraena novaehollandiae	snook
Sphyraena obtusata	yellowtail barracuda
Sphyraena pinguis	striped barracuda
Sphyraena putnamae	military barracuda
Sphyraena qenie	blackfin barracuda
Synodontidae	
***Saurida* sp.**	
Saurida argentea	shortfin saury
Saurida filamentosa	threadfin saury
Saurida gracilis	gracile saury
Saurida grandisquamis	grey saury
Saurida longimanus	longfin saury
Saurida nebulosa	clouded saury
Saurida undosquamis	largescale saury
Saurida wanieso	wanieso saury
***Synodus* sp.**	
Synodus binotatus	twospot lizardfish
Synodus dermatogenys	banded lizardfish
Synodus hoshinonis	blackshoulder lizardfish
Synodus indicus	Indian lizardfish
Synodus jaculum	tailspot lizardfish
Synodus macrops	triplecross lizardfish
Synodus sageneus	fishnet lizardfish
Synodus similis	streaky lizardfish
Synodus variegatus	variegated lizardfish
Synodontidae sp.	
Trachinocephalus trachinus	painted grinner
Tetraodontidae	
***Arothron* sp.**	
Arothron caeruleopunctatus	bluespotted puffer
Arothron hispidus	stars-and-stripes puffer
Arothron manilensis	narrowlined puffer
Arothron mappa	scribbled puffer
Arothron nigropunctatus	blackspotted puffer
Arothron reticularis	reticulate toadfish

Binomial	Common Name
Arothron stellatus	starry puffer
***Canthigaster* sp.**	
Canthigaster axiologus	crowned toby
Canthigaster callisternus	clown toby
Canthigaster rivulata	ocellate toby
***Lagocephalus* sp.**	
Lagocephalus inermis	smooth golden toadfish
Lagocephalus lunaris	rough golden toadfish
Lagocephalus sceleratus	silver toadfish
Lagocephalus spadiceus	brownback toadfish
lagocephalus suezensis	
Tetraodontidae sp.	
Feroxodon multistriatus	ferocious puffer
Tetractenos hamiltoni	common toadfish
Torquigener pallimaculatus	rusty-spotted toadfish
Torquigener parcuspinus	yelloweye toadfish
Torquigener pleurogramma	weeping toadfish
Torquigener tuberculiferus	fringe-gill toadfish

Binomial	Common Name
Tylerius spinosissimus	finespine pufferfish
Teuthida	
Teuthida sp.	
Sepioteuthis lessoniana	northern calamari
Uroteuthis (Photololigo) chinensis	mitre squid
Uroteuthis (Photololigo) edulis	swordtip squid
Triglidae	
Triglidae sp.	
Chelidonichthys kumu	red gurnard
Lepidotrigla argus	eye gurnard
Lepidotrigla grandis	little red gurnard
Lepidotrigla russelli	smooth gurnard
Lepidotrigla vanessa	butterfly gurnard
Pterygotrigla elicryste	dwarf gurnard