2017

The Ecology of Temperate Soft Sediment Fishes: Implications for Fisheries Management and Marine Protected Area Design

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Recommended Citation

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2017

School of Biological Sciences

This thesis is presented as part of the requirements for the award of the Degree of Doctor of Philosophy of the University of Wollongong

UNIVERSITY OF WOLLONGONG

ABSTRACT

Marine protected areas (MPAs) are an increasingly common management approach to assist in conserving marine biodiversity by limiting, avoiding or removing anthropogenic activities such as pollution, habitat destruction and fishing. Globally, a considerable proportion of the area under protection in MPAs comprises soft sediments. Research on rocky reefs and coral reefs has demonstrated that when MPAs are well designed and implemented, the abundance and biomass of targeted fish species can increase. However, demersal fish on marine soft sediments have been poorly studied and it remains unclear whether they respond in the same ways to protection as fish on other habitats. In this thesis, I aimed to assess (i) whether MPA protection in south-east Australia has affected the species composition, abundance and size of demersal marine soft sediment fishes among management zones and (ii) the degree of long-term residency shown by a key recreationally and commercially targeted species in relation to MPA size and zoning.

First, I used baited remote underwater videos (BRUVs) to sample the fish assemblages and test hypotheses about the effects of MPA management and implementation. My results revealed that in, shallow (10 m), deep (20 m) and offshore (50–60 m) waters, the demersal soft sediment fish assemblages were characterised by a few frequently occurring species. At all depths sampled the most common species were flathead (*Platycephalus caeruleopunctatus* & *Platycephalus grandispinis*). Shallow- and deep-water BRUV sampling was carried out between May and June in 2011, 2013 and 2015, within Jervis Bay Marine Park. At the assemblage level, no impact of MPA zoning was detected at either depth. There was also no difference between zones in total relative abundance (abundance of all species combined) or species richness at either depth. Abundances of individual species (those appearing on $\geq 25\%$ of BRUVS samples) were also compared between zones; In shallow-water, there was a 32% greater abundance of *Platycephalus* spp. in no-take zones (NTZs) compared to partially protected areas (PPAs) over the study. In addition, abundances were more stable in NTZs across time. In shallow-water, Eastern fiddler ray (*Trygonorrhina fasciata*) and shovelnose ray (*Aptychotrema rostrata*), also had higher abundances in NTZs compared to PPAs in 2015. In deep-water there were no differences between zones for any individual species. There were no differences in length of flathead between zones at either depth. Offshore comparisons were carried out between August

and December in 2015, within Jervis Bay Marine Park, Batemans Marine Park and open access (OA) areas outside the two MPAs. Assemblages showed clear differences among NTZ, PPA, and fished OA areas. At the species level, on average, larger individuals of longspine flathead (*P. grandispinis*) were observed in NTZs than in both PPAs and OAs. There were also substantially higher abundances of ocean jackets (*Nelusetta ayraudi*) in NTZs. In offshore water there were no differences in abundances among zones for any other species or in species richness and total relative abundance.

Second, I tested the assumption that fish on soft sediments are unlikely to show residency by evaluating the movement patterns of the bluespotted flathead (*P. caeruleopunctatus*) in Jervis Bay Marine Park. Bluespotted flathead were acoustically tagged within a NTZ in spring 2014 (n=25), autumn 2015 (n=15), and summer 2015 (n=6). I then monitored the tagged fish for 625 days. Bluespotted flathead exhibited small-scale and long-term residency within the NTZ. Over the first 108 days post tagging most fish (74%) remained within a ~200 ha area of NTZ and were detected frequently. I observed residency of up to 600 days. Although close to two thirds of the tagged fish were only detected within Jervis Bay, the remainder were detected moving up to 155 km from where they were tagged. Generally, these fish had a prolonged period of site residency before making these large-scale movements. Importantly, range testing confirmed that acoustic tags in this habitat were detected with a high degree of confidence and reliability.

My findings demonstrate that temperate demersal fishes found on marine soft sediments can be influenced by protection within MPAs at a number of spatial scales. However, the response is highly variable among species with the majority showing no response, a relatively small effect size for those that do show a response and assemblage wide responses occurring in offshore waters but not within nearshore waters. In conclusion, marine soft sediments are an extensive habitat that harbour a unique demersal fish community. This habitat supports an important component of marine biodiversity and represents a rich fishery resource. This study provides a rare example of MPA effects on demersal soft sediment assemblages and presents substantial evidence of long-term residency by a demersal soft sediment associated fish within an NTZ.

TABLE OF CONTENTS

CERTIFICATION

I, Lachlan Clement Fetterplace, declare that this thesis, submitted in partial fulfilment of the requirements of the award of Doctor of Philosophy, in the School of Biological Sciences, University of Wollongong, is wholly my own work unless referenced or acknowledged. The document has not been submitted for qualifications at any other academic institution.

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31th August 2018

ACKNOWLEDGEMENTS

I wish to acknowledge the people of Dharawal Country and of Yuin Country, the owners of the lands on which I study, live and carried out the research in this thesis. To my supervisors Prof Andy Davis and Dr Nathan Knott, I will be forever grateful for the opportunity to work with you and I feel particularly fortunate to have had the extensive advice and support you have both provided. Thank you. I would also like to acknowledge and thank; my advisor Dr Matt Taylor, Dr Matt Rees, Kye Adams, Ian Osterloh, Margie Andréason, Pearse Buchanan, Declan Fetterplace, Cameron Fetterplace, Evelyn Fetterplace, Mark Fackerell, Matt Carr, Joe Neilson, Frances Clements, the staff at Jervis Bay Marine Park, Batemans Marine Park and at Booderee National Park, and the numerous other people that have assisted me along the way. Thank you to my family. My siblings and their partners have helped me in numerous ways over the last few years. Without the extensive assistance and unfaltering support from my parents, this PhD journey would have been a lot more difficult.

Most importantly of all, to Sara and Tovah (and Eldfrid, who appeared right in the middle of the final touches), who were equal parts help and hindrance in this whole endeavour—Thanks for putting up with me through all of this. I am lucky to have you with me and I wouldn't have got through it all without you by my side.

13

synchronisation between each receiver location. A) test download and first download (empty line through data is receiver changeover and download when only one receiver was in the water). Note that receivers 17–20 were added to the VPS at a later date in March 2015). B) Second download of receivers

LIST OF TABLES

LIST OF PLATES

Chapter 1 General Introduction

Marine soft sediments harbour a unique fish community that globally represents a considerable component of coastal biodiversity. Fish on this habitat also comprise an important commercial and recreational fishery. Marine protected areas (MPAs) are a management strategy implemented to conserve marine biological diversity. Large areas of soft sediment habitat worldwide are encompassed within MPAs and in many MPAs, soft sediments are the dominant habitat type. However, there has been little investigation of soft sediment fish assemblages in relation to their response to MPA implementation and appropriateness of MPA design. As a result, the effect of protection on marine soft sediment demersal fish communities is virtually unknown. In this chapter, I briefly review the current status of research on MPAs, and highlight the need for monitoring and research to understand the impacts of protection on soft sediment associated demersal fishes.

1.1 Marine Protected Areas

Marine protected areas (MPAs) are areas that are set aside primarily to protect biodiversity. To achieve this, anthropogenic disturbance is removed or limited (Gell and Roberts 2003). The term 'marine protected area' encompasses a wide range of management and conservation methods with differing levels of protection [see, Day et al. (2012); Guidelines for applying the IUCN Protected Area Management Categories to Marine Protected Areas; Horta e Costa et al. (2016) for an alternative regulation-based classification system; and Dudley et al. (2017) for further discussion of MPA definitions].

However, for practical purposes, MPAs can be grouped into two broad categories (Sala and Giakoumi 2017). The first, no-take MPAs, refers to any area in which all forms of fishing, extractive harvesting, mining and other habitat destruction are prohibited (no-take MPA is analogous with marine reserve, no-take zone, sanctuary zone and green zone). The second category, partially protected areas (PPAs), includes areas with less restrictive regulations than no-take MPAs and generally allow some forms of fishing and harvesting (e.g. they may allow recreational

fishing but prohibit commercial fishing, or they may limit certain types of fishing gear). An MPA can be completely no-take or completely PPA, or in many cases MPAs are made up of multiple management zones.

In most instances, the objective of the MPA is to conserve biodiversity. For example, in Australia, MPAs have primarily been set up following the 'CAR' principles, which are intended to provide comprehensive, adequate and representative protection of habitats, species and biodiversity. MPAs are also often considered to complement the broader spatial management of the marine environment (IUCN 1994, ANZECC 2001, NRSMPA 2011). The public's perception on the objective of MPAs is mixed but often the expectation is that the priority of MPAs is to increase the numbers and size of fish (Pomeroy et al. 2005). It has also been suggested that by maintaining populations that are not under fishing pressure, MPAs can also be used as insurance and a buffer against potential fisheries management mistakes (Allison et al. 1998, Lauck et al. 1998). In some cases well-managed MPAs may also mitigate other anthropogenic stresses and the impacts of climate change (Roberts et al. 2017).

At the most basic level, populations of fish that are targeted by fishing, are expected to suffer lower mortality rates and increase in abundance when protected in an MPA. If the MPA is appropriately designed for the life histories of the harvested fish species, eventually, population densities should return to levels that were present before fishing commenced (Tetreault and Ambrose 2007). In addition, as a result of decreased mortality, individuals would be likely to have a greater chance of growing to larger sizes, survive longer and have increased reproductive potential (Barrett et al. 2007). In theory, these changes in abundances and size will vary across a gradient of fishing pressure, with larger fish in the centre of a reserve and fewer and smaller fish in heavily fished areas outside MPAs (Kramer and Chapman 1999). Many species, however, are not targeted but may be affected indirectly (e.g. as bycatch). These species may also experience similar changes inside MPAs as the targeted species (Byers and Noonburg 2007). On the other hand, there may be reductions in prey species as predator numbers increase (Harasti et al. 2014). An additional benefit provided by MPAs is a reduction in habitat degradation by fishing gear (Allison et al. 1998, Byers and Noonburg 2007).

As well as conserving species within their boundaries, MPAs are also predicted to potentially benefit surrounding fisheries in a number of ways: First, 'spill-over' of adults and juveniles across borders (Roberts et al. 2001). Second, the dispersal of pelagic eggs and larvae which would result in increased settlement of juveniles outside the boundary (Botsford et al. 2003, Gell and Roberts 2003) thereby replenishing fished stocks outside the MPA. It is particularly difficult to show the effects of spill-over, especially of eggs and larvae that are small and may be taken long distances by currents and this is perhaps why there are few studies showing this effect or even testing this prediction (Roberts 1997, Halpern et al. 2009).

Finally, MPAs are predicted to benefit surrounding fisheries by providing reference areas for research, where fishing is not permitted. No-take areas that are working effectively 'represent human predator exclusion plots within a matrix of fished coasts' (Edgar et al. 2014). Likewise, Breen (2007) argues that multiple zone MPAs that have a gradient of fishing pressure are 'ideal to test and refine hypotheses about marine ecosystems and their management'. This is particularly useful as outside MPAs there are almost no areas in the ocean that are now un-fished and that can be used as controls in studies of fishing impacts.

1.2 Assessment and evidence for the effects of Marine Protected Areas.

Following the implementation of MPAs there is the need to assess and quantify whether an MPA is meeting its objectives. In many cases this is in response to public and scientific interest and in others there are legislative requirements to make assessments (Kelleher and Kenchington 1992, NSW MPA 2009). Assessments undertaken to meet legislative requirements often focus on determining if the MPA is effective in achieving comprehensive, adequate and representative (CAR) protection of species and habitats within its borders. The public are often more interested in how protection has impacted upon assemblages and whether there are changes in population structure and abundances.

In many areas, there has been extensive research into understanding and quantifying the effects of MPAs on the assemblages found within their boundaries. It is common for studies to test the response of fish communities (or individual species) to the implementation of MPAs by comparing fish assemblages within the un-fished boundaries of no-take MPAs to fished areas outside (e.g. Barrett et al. 2007, Abecasis et al. 2013, Kelaher et al. 2014, Whitmarsh et al. 2014, Ferguson et al. 2016, Malcolm et al. 2016). As a result, there is increasing scientific evidence validating predictions that the abundance, diversity and length of targeted fish species can increase inside MPAs when compared to areas that continue to be fished (Willis et al. 2003, Alcala et al. 2005). In the largest and most comprehensive assessment of the ecological impacts of MPAs to date, protected fish populations increased relative to fish populations in 71 percent of the 218 MPAs studied (Gill et al. 2017). On average, positive responses occurred in both no-take MPAs and PPAs, although response ratios were almost two times greater in no-take MPAs (Gill et al. 2017). However, the majority of the research on the effects of protection has been undertaken on reefs (Caveen et al. 2013). As a result, much of the clearer evidence for the effectiveness of MPAs have been provided by studies on rocky reef (Babcock et al. 1999, Barrett et al. 2007, Tetreault and Ambrose 2007, Babcock et al. 2010, Aburto-Oropeza et al. 2011) or coral reefs (Evans and Russ 2004, Alcala et al. 2005, Harvey et al. 2012).

MPAs are by no means all effective in meeting their objectives or resulting in benefits to fish populations (Mora and Sale 2011). Gill et al.(2017) reported that the level of recovery was strongly linked to the management of the sites, with shortfalls in staffing and funding the greatest barrier to recovery of fish populations in MPAs. Similarly, in another global study of reefs in 87 MPAs, Edgar et al. (2014) found that the measurable benefits of the impact of fishing removal was strongly influenced by the five NEOLI (no-take, enforced, old, large and isolated) planning and management features. Those MPAs that had elevated biomass of targeted fish species compared to fished areas, scored highly with multiple NEOLI features. Those MPAs only meeting one or two of the features fared poorly and were rarely ecologically distinguishable from areas that continued to be fished (Edgar et al. 2014).

In contrast to other habitats, minimal research has examined fish responses on marine soft sediments to the removal of fishing in MPAs (Caveen et al. 2012). Positive responses to protection by fish assemblages were reported by Gill et al. (2017) in almost all regions and habitats, however there was insufficient data available to make an assessment on soft sediments. Those studies on fish in MPAs that have included soft sediment habitats, generally looked at soft sediments only in comparison or close proximity to nearby habitats such as reef, seagrass, or kelp (e.g. Roberson et al. 2015). The soft sediment areas immediately surrounding reef are within a 'halo' of reef influence and assemblage with 100's of metres of reef are closer to reef assemblages than those on soft sediments outside this halo (Langlois et al. 2005, Schultz et al. 2012). As a result, these areas are not representative of the majority of soft sediment habitat. In some cases, there is some research on soft sediment fish prior to MPA implementation (e.g. Sousa 2011, Hill et al. 2014, Abecasis et al. 2014a) but it is almost completely absent once MPAs are established (though see; Fetterplace 2011, Sousa 2011, Abecasis et al. 2014a, Adams 2016). The net outcome of this lack of data, which has been highlighted in a number of reviews, is that the effect of protection on demersal fish communities inhabiting marine soft sediments is largely unknown (Lester et al. 2009, Bloomfield et al. 2012, Caveen et al. 2012, Caveen et al. 2013, Sciberras et al. 2013). With the recent movement towards multiple use MPAs which incorporate NTZ and PPA, there is now the opportunity to assess the impact of fishing pressures across a gradient of fishing pressure (no-take MPA vs various PPA with differing levels of restrictions vs fished areas outside MPAs). Again, previous research utilising multiple use parks to test across gradients of fishing has focused on reef (e.g. McKinley et al. 2011, Kelaher et al. 2014), with no studies to my knowledge taking this approach on marine soft sediments.

1.3 Fish Movement and Marine Protected Areas

If an MPA is to provide effective protection for a target species, an understanding of the species home range and movement is an essential component of effective MPA design (Kramer and Chapman 1999, Moffitt et al. 2009). Once an MPA is in place, effective management at the species and population level requires not only

an understanding of abundances, the species present and their size, but also knowledge of patterns of fish mobility (Zeller and Russ 1998, Pittman and McAlpine 2003, Topping et al. 2005, Afonso et al. 2011) and of habitat requirements and habitat distribution (Abecasis et al. 2014b). Understanding movement patterns (when and where fishes move and how much time they spend in particular areas) aids in identification of preferred fish habitat, aggregation locations and spawning grounds (Hindell 2007). As a result, such information can improve estimates on how much time fish are likely to spend inside an MPA (Grüss et al. 2011). To be effective, MPAs must be large enough to encompass the movements of species they are designed to protect or encompass key life stages (Kramer and Chapman 1999) and contain adequate suitable habitat (Abecasis et al. 2014b). Excessive movement of animals outside reserve boundaries is a main reason that many MPAs fail to meet objectives (Edgar et al. 2014). On the other hand, an intermediate level of adult movement is desired if MPAs are to benefit fisheries outside the reserve (Botsford et al. 2003). Without movement data, it is difficult to predict the best configuration or how large an MPA needs to be in relation to fish movement to be most effective.

There are numerous examples of studies on fish movement and the number of studies is increasingly rapidly, with the majority of marine tracking research now using acoustic and satellite telemetry (Box 1). Understanding the movement patterns of populations that are the target of protection is particularly important (Grüss et al. 2011) and a considerable subset of tracking studies have looked at fish movement directly in relation to MPAs (e.g. Zeller and Russ 1998, Willis et al. 2001, Parsons et al. 2003, Bellquist et al. 2008, Pastor et al. 2009, March et al. 2010, Afonso et al. 2011, Abecasis et al. 2015, Lee et al. 2015, Harasti et al. 2015a, Ferguson et al. 2016). These studies demonstrate that many species, particularly generalist and reef associated species, spend a large amount of time in relatively restricted areas – consistent with the size of many MPAs globally. Hence, it is likely that for many of these species one would expect to see differences in abundance and size between no-take and fished areas, if fishing outside the MPA was having a large impact. Alternatively, if the fish were regularly moving distances larger than the reserve sizes, then even if the fishery was having a sizeable impact it would be difficult to detect differences in abundance and size of fishes between fished and un-fished areas.

Although we have a good understanding of fish movement in some habitats, this is not the case on marine soft sediment, where relatively few studies have investigated movement of demersal fishes (Box 1). In addition, only a small number of these look at the movement of demersal fish on open coastal soft sediments and rarely in relation to MPAs (though see Fetterplace 2011, Fetterplace et al. 2016 on *Platycephalus caeruleopunctatus*, and Abecasis et al. 2014a on *Solea senegalensis*) . Despite this lack of data, there is often an assumption that fish on this habitat have little reason to show site attachment (Caveen et al. 2012), This assumption is often based on the lack of structure and the overall homogeneous appearance of marine soft sediments; habitat characteristics that do not appear to provide a reason for fish to be resident to a particular area for very long (Lowe and Bray 2006, Caveen et al. 2012, Fetterplace et al. 2016).

1.4 Marine Protected Areas in New South Wales, Australia

The research in this thesis is undertaken in the state of New South Wales (NSW) on the East Coast of Australia. In NSW, five coastal marine protected areas were established between 1998 and 2006 with the aim of protecting biodiversity and maintaining ecological processes (Marine Parks Act 1997). Each MPA is a mix of strictly no-take zones, and partially protected zones which allow recreational fishing and prohibit commercial trawling, long lining, mining and dredging (Read and West 2010). The MPAs were designed following CAR principles (comprehensiveness, adequacy, representativeness), a process that seeks to ensure that 1) all ecosystems in a bioregion are included, 2) that the area under protection is sufficient to ensure ecological viability and integrity of populations, species and communities and 3) that the areas included represent the biological diversity present in an ecosystem (ANZECC 2001).

Box 1 - Marine Fish Movement: A Systematic Review by Habitat, of Species Tracked Using Acoustic and Satellite Telemetry.

Acoustic and Satellite Telemetry: Acoustic and Satellite telemetry are now the main methods for studying the movement patterns of marine animals (Hussey et al. 2015), and globally have been used extensively (Fig. 1.1). Both methods involve the attachment of transmitters on animals. These transmitters (or tags) then transmit or store a location signal that allows an estimate of the animal's position. This technology has revolutionised the study of fish movement by allowing tagged individuals to be tracked for long periods with relatively high spatio-temporal resolution (DeCelles and Zemeckis 2014, Hussey et al. 2015). Unlike traditional mark-recapture techniques where only release and re-capture locations can be obtained, acoustic and satellite tags provide data across multiple time-points and fish do not need to be re-caught to collect the data (Dudgeon et al. 2015).

Research Effort: The number of studies using acoustic and satellite telemetry to monitor fish movement is increasing rapidly (Hussey et al. 2015). Although the technology has been reviewed previously (e.g. Arnold and Dewar 2001, Heupel et al. 2006, Rutz and Hays 2009, Hussey et al. 2015), there have been no reviews examining telemetry effort by habitat. A systematic review of published telemetry literature which I have carried out reveals two clear patterns; (1) there is clear concentration of research effort by location (Fig. 1.1), with North American and to a lesser extent Europe having the bulk of research effort (Fig. 1.1), and (2) there has been a focus on fish associated with particular habitat types (Fig. 1.1). Pelagic and reef associated species are relatively well studied globally. In contrast, the movement of demersal fishes associated with soft sediments has had considerably less research effort (6% of the reviewed studies) and a large portion of the research that has been undertaken on this habitat has been in the United States. The 51 studies on soft sediments were on 24 species, with sturgeons (*Acipenser* spp.) accounting for 11 studies. Only 3 of the demersal soft sediment studies were in marine protected areas and of these 2 were in estuaries. Given the extent of marine soft sediment habitats and the unique associated fish species the small amount of research effort is surprising. It is also worth noting that there were almost no studies of species inhabiting waters deeper than 200 m in any habitats, including soft sediments.

Figure 1.1: Global distribution of acoustic and satellite telemetry studies tracking marine and estuarine fish (bony and cartilaginous). Studies are categorised and plotted by the habitat type the tracked species are associated with. I identified the 729 studies mapped by undertaking a systematic search of the published literature & creating a list of studies on fish movement using acoustic and satellite telemetry. Each study was then assigned to one of the following broad habitat categories [adapted from FishBase (2017)]: pelagic, benthopelagic, demersal generalist, demersal soft sediment associated, reef associated. Two categories bathypelagic & bathydemersal (below 200 m depth) had no studies. For detailed review methods & reference list see appendix B. Maps were created using Python (Python Software Foundation, www.python.org), the matplotlib package (Hunter 2007), & the Iris and Cartopy packages from the UK Met Office (www.scitools.org.uk).
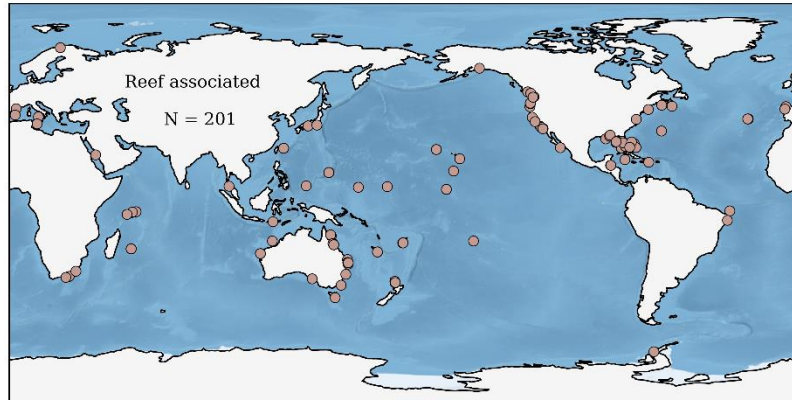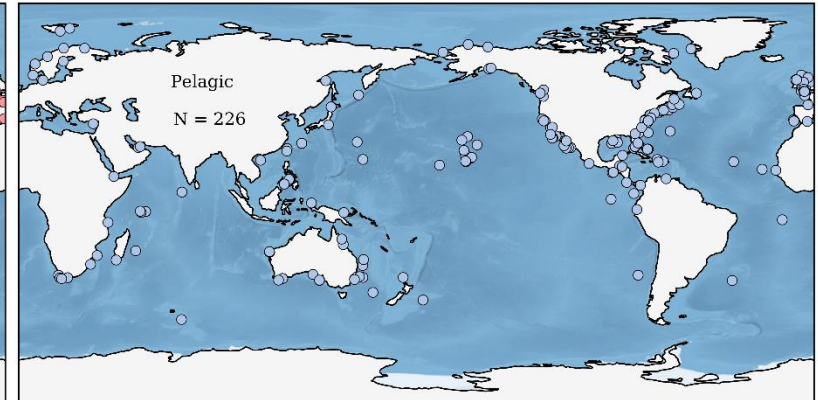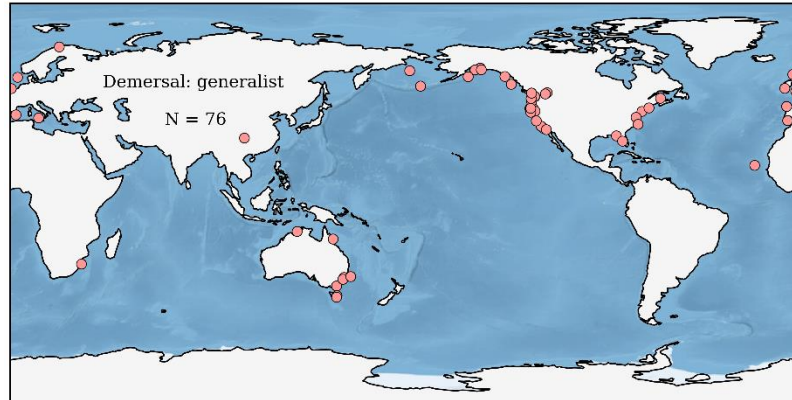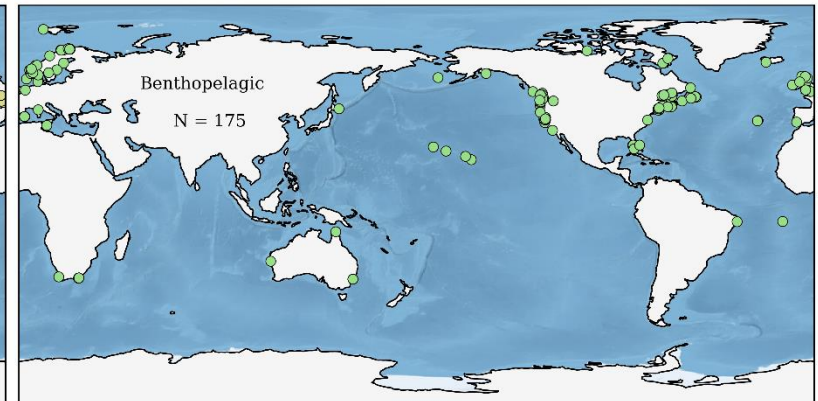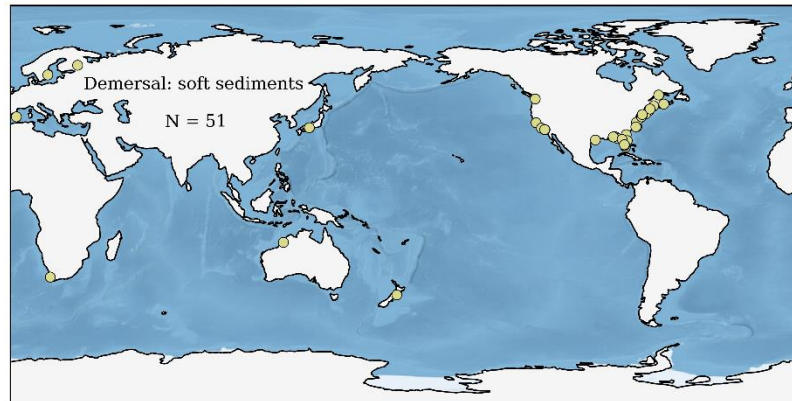
Habitat Categories - *Pelagic*: occurring mainly in the water column, not feeding on benthic organisms; *benthopelagic*: living and/or feeding on or near the bottom, as well as in midwater, between 0–200 m; *demersal generalist*: utilises or found on multiple habitat types, living and/or feeding on or near the bottom, between; 0–200 m*; demersal soft sediments;* living and/or feeding on or near the bottom on soft sediments, between 0–200 m; *reef-associated*: living and/or feeding on or near reefs, between 0–200 m.

In all five of these coastal NSW MPAs, soft sediments are the most extensive habitat type. The amount of soft sediments protected appears to have been largely serendipitous and a result of logistics e.g. to protect widely spaced areas of reef within a single MPA, the large areas of soft sediments between them were included in protection. However, the proportion of soft sediments is representative of the dominance of this habitat in the states coastal waters generally (MEMA 2017). Although the MPAs in NSW have been in place for a relatively long period (the first two MPAs established, zoning and regulations come into effect in 2002), there has been no research evaluating impacts of protection or zoning type on the fish communities occupying marine soft sediment habitats. The need for data on subtidal marine soft sediments has been identified as a key knowledge gap that is hindering the implementation of effective evidence based MPA management in NSW (Brooks et al. 2013).

Recently, there has been zoning changes[1] in NSW MPAs that opened some no-take zones to recreational fishing (Brooks et al. 2013). These changes centred on the idea that fish on soft sediments do not show residency and therefore the no-take areas on soft sediments provide little conservation value. The idea that fish on marine soft sediments habitats are all highly mobile has been put forward by some researchers (e.g. Kearney 2007 in relation to ocean beaches) and repeated by politicians across the political spectrum (e.g. 'beaches do not have resident populations of fish; fish move about and come and go': The Hon. R, Brown in Marine Parks Amendment Moratorium Bill 2013 and 'Fish do not live in cages or adhere to lines' The Hon. D, Gay: Marine Parks Amendment 2007). This may be true for many of the pelagic species, such as taylor and Australian salmon, used as examples. However, the majority of demersal fish associated with marine soft sediments in NSW (or anywhere) have no movement information available (Box 2). Consequently, it is currently impossible to say whether they show residency, migrate, have specific spawning or aggregation locations.

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[1] These zone changes did not impact on the areas assessed in this thesis.

Box 2 - Fish Movement in Temperate Australia: In temperate Australian waters, the research on bony fish and cartilaginous fish movement largely follows the global trends. Movements of bony fish are relatively well studied in habitats such as estuaries and on rocky reef. For example, in estuaries; black bream (*Acanthopagrus butcheri*; Hindell et al. 2008, Sakabe and Lyle 2010), yellowfin bream (*Acanthopagrus australis*; Payne et al. 2013), dusky flathead *(Platycephalus fuscus*; Hindell 2008) and mulloway (*Argyrosomus japonicus*; Taylor et al. 2006); on rocky reef; eastern blue groper (*Achoerodus viridis*; Lee et al. 2015), western blue grouper (*Achoerodus gouldii*; Bryars et al. 2012), luderick (*Girella tricuspidata*; Ferguson et al. 2013, Ferguson et al. 2016) and drummer (*Girella elevata*; Stocks et al. 2015). Large apex species or species found in tropical and subtropical regions tend to be the focus for shark movement research at a global scale (Chapman et al. 2015, Bass et al. 2017) and rays in general have been poorly studied (Le Port et al. 2012, Vaudo and Heithaus 2012). In a similar manner, in temperate Australia, large pelagic or wide-ranging apex shark species have received much of the research attention. For example, white shark (*Carcharodon carcharias*; Bruce et al. 2006, Bruce and Bradford 2012, Bruce et al. 2013, Harasti et al. 2017, McAuley et al. 2017), sevengill (*Notorynchus cepedianus*; Barnett et al. 2010, Barnett et al. 2011), tiger shark (Galeocerdo cuvier; Holmes et al. 2014) and whaler sharks (*Carcharhinus spp.*; Rogers et al. 2013, Heupel et al. 2015). Temperate demersal species that have movement data tend to be reef associated or are habitat generalists. For example, wobbegong shark (*Orectolobus halei*; Huveneers et al. 2006), draughtboard shark *(Cephaloscyllium laticeps*; Awruch et al. 2012, Bruce et al. 2018) and gummy shark (*Mustelus antarcticus*; Barnett et al. 2010, Bruce et al. 2018). And for most of these the data is limited to one location and/or a small number of tagged individuals. The Port Jackson shark is one of the few demersal shark species in temperate Australian waters with movement data from a number of studies and locations using electronic tags (e.g. Powter and Gladstone 2009, Bass et al. 2017, Keller et al. 2017) to build on earlier work using mark recapture tagging methodologies (e.g. O'Gower and Nash 1978, Powter and Gladstone 2009).

For the majority of demersal marine species associated with soft sediment in temperate Australia, there are no data on their movements. The short-term tracking of blue-spotted flathead on marine soft sediments (Fetterplace 2011, Fetterplace et al. 2016) is one of the few exceptions and provided the impetus for the current study.

1.5 Thesis Aims and Structure

There continues to be rapid increases in the amount of area under protection globally (Worm 2017) and a large proportion of this area covers marine soft sediments despite our limited knowledge of MPA efficacy on this habitat. In contrast to fish on other habitat types, demersal fish on soft sediments are poorly studied and it remains to been seen whether they respond in the same ways to the removal of fishing pressure as those on other habitats. There is a clear need for research focusing specifically on the effects of MPAs on these assemblages. In addition, acquiring data on fish movement and behaviour is essential to effective MPA design and management on soft sediments. This will allow an understanding and assessment of the benefits of protection on this habitat and increase the information available for informed management. In this thesis, I examined the ecology of demersal soft sediment fishes in two temperate Marine Protected Areas (MPAs) in south-east Australian waters; Jervis Bay Marine Park (JBMP) and Batemans Marine Park (BMP), with two main aims; to assess (1) how protection impacted on these assemblages, and (2) the degree of long-term residency shown by a key species in the assemblage in relation to MPA size and zoning.

The specific aims addressed by each chapter are:

Chapter 2 — In this chapter I aimed to determine if there were differences in fish assemblages, abundances and size of fish, among no-take and partially protected fished zones on near shore soft sediments in JBMP. I hypothesised that the removal of fishing pressure on soft sediments in JBMP no-take zones will result in changes to fish abundance and size of recreationally and commercially targeted species. Specifically, I tested the following three predictions: (1) That abundances of targeted and bycatch species would be greater in no-take zones; (2) that targeted species of fish in no-take zones would be larger than those in fished zones; and (3) there would be greater diversity in no-take zones compared to fished zones. I tested for these predicted effects of no-takes zones multiple times between 2011 and 2015. In relation to time, I predicted that the patterns outlined above would either be stable through time (i.e. indicating effects had already taken place and were stable) or that these patterns would be developing through time.

Chapter 3 — In this study in open coastal waters on the South-East coast of New South Wales, Australia, I aimed to determine if there were differences in soft sediment demersal fish assemblages across a gradient fishing pressure. The management zones within the two MPAs in the region which include both no-take zones and partially protected areas, and the dominance of soft sediments generally, provided an excellent opportunity to carry out such an assessment. I hypothesised that the differing levels of fishing pressure would result in differences in fish assemblages and that the differences would be relative to the amount of fishing pressure. More specifically I predicted that 1) Abundances of targeted and bycatch species would be greatest in no-take zones, then partially protected areas and lowest in open access areas outside the parks; (2) Size of targeted species would follow the same pattern, with the largest fish in no-take zones; and (3) There would be greater diversity in no-take zones compared to partially protected areas and open access areas would have the lowest diversity.

Chapter 4 — Passive acoustic tracking has become the most common form of monitoring marine fish movement patterns. To effectively undertake tracking of fish in a given location, design a tracking array and understand the results obtained, an *in-situ* understanding of equipment functionality over space and time is required. In this chapter, I undertook acoustic range testing over an extended 70-day period on soft sediments in Jervis Bay Marine Park. The main aim was to determine detection probabilities (how reliably I could detect a tagged fish at varying distances from a receiver) to use in the design of a large passive tracking array on soft sediments.

Chapter 5 — The aim of this chapter was to determine the length and degree of residency shown by bluespotted flathead (*Platycephalus caeruleopunctatus*) in relation to MPA size and zoning. This species is the most common targeted demersal fish in the soft sediment assemblage from 0 to ~60 m depth off the South-Eastern coast of Australia. In the current study, in collaboration with Dr Nathan Knott (DPI Fisheries NSW), I firstly developed and deployed a large passive acoustic tracking system on soft sediments in JBMP. I then used this array to comprehensively assess and quantify the short- and long-term movement patterns and residency of bluespotted flathead within the Jervis Bay Marine Park (New South Wales, Australia). The main aim of this study was to determine how to what degree and over what time-frame bluespotted flathead show residency within an area comparable to NSW Marine Park no-take zones.

Plate 2.1: A southern eagle ray (*Myliobatis tenuicaudatus*) and eastern fiddler ray (*Trygonorrhina fasciata*) baited remote underwater video system deployed on soft sediments in Hare Bay No-Take Zone, Jervis Bay Marine Park.

2.1 Introduction

Marine soft sediments are the most common habitat on earth (Wilson 1991, Dutkiewicz et al. 2015) and are heavily exploited by both commercial and recreational fishers. Demersal fishes, that is those living or feeding on the seafloor, comprise approximately one third of the global fish catch and much of this is caught on soft sediments such as marine sand (AERL 2011, FAO 2016). Soft sediments are the major near-shore and continental shelf environments (Caveen et al. 2012) and almost all marine soft sediments shallower than 1200 m are fished, apart from no-take marine protected areas (MPAs) (Handley et al. 2014).

Marine protected areas are an increasingly common management approach to assist in conserving marine biodiversity (Lubchenco and Grorud-Colvert 2015, White et al. 2017). They use spatial management of a range of human activities by limiting, avoiding or removing anthropogenic activities such as pollution, habitat destruction and fishing (Wells et al. 2016). They are primarily implemented to conserve biodiversity (Wells et al. 2016) but can have potential utility for fisheries management (Botsford et al. 2003, Gladstone 2007). When MPAs are well designed, implemented and human pressures are sufficient[1], the abundance, diversity, and length of targeted fish species can increase (Barrett et al. 2007, Edgar and Stuart-Smith 2018). Almost all of the assessments of MPAs effects on fish have, however, been on rocky reefs and coral reefs; rarely have soft sediments been assessed (Caveen et al. 2012).

Despite MPAs being dominated by unvegetated soft sediments, we have very little knowledge on the effects of MPA implementation on the unique communities associated with this habitat (Caveen et al. 2012). MPAs on marine soft sediments are often put in place without knowing if they will protect the fish diversity in this habitat and monitoring of these assemblages, to assess ecological changes, rarely occurs. A good example is the Great Barrier Reef Marine Park in which soft sediments comprise 95% of the seafloor (Caveen et al. 2012), yet there is extremely little information available on the biological effects of management zones on non-reef habitats in the park (McCook et al. 2010). Studies testing the effect of no-take MPAs on fishes are numerous but have largely ignored soft sediments or were limited in nature. Studies on MPA effects that have included marine soft sediment habitats were generally in very shallow water or only

33

[1] For example, if there was no or minimal fishing occurring before MPA implementation then there is unlikely to be any response in a fish population when fishing is prohibited. In the same way, if there is little or no fishing occurring outside an MPAs borders then differences in fish populations between inside and outside the MPA will not be a result of fishing pressure.

looked at soft sediment habitats in comparison to nearby rocky reefs, sea grass or coral (e.g. Cappo et al. 2007). Those including sites within hundreds of metres of reef are likely to be sampling a 'halo' fish assemblage which may be closer in composition to reef assemblages than those on soft sediments (e.g. Langlois et al. 2005, Schultz et al. 2012). As a result, and as continually highlighted in a number of reviews, the effect of protection on marine soft sediment demersal fish communities is effectively unknown and remains unassessed (Lester et al. 2009, Bloomfield et al. 2012, Caveen et al. 2012, Caveen et al. 2013, Sciberras et al. 2013). Understanding how fish on marine soft sediments respond to MPA implementation appears to be a major gap in our understanding of this worldwide conservation approach.

There continues to be rapid increases in the amount of marine soft sediments under protection globally despite our limited knowledge of MPA efficacy on this habitat. This increase is being driven by exponential growth in the number of MPAs generally (Worm 2017) and also the trend towards more 'vast' MPAs (e.g. The Papahānaumokuākea and the Pacific Remote Islands Marine National Monuments) which cover large areas of deep water soft sediments. Soft sediments are often included in MPAs almost accidentally, as they cover areas between other habitats or sites of specific interest, and some are protected in response to CAR (comprehensive, adequate, representative reserves) approaches to spatial planning (Coleman et al. 2013). Beyond their inclusion, seemingly little thought is put into threats to or conservation of these areas, hence the lack of assessment generally or specific hypotheses or goals proposed for these conservation areas. The amount of marine soft sediment habitat being protected in MPAs has far outpaced research on the ecological impacts of MPAs on this habitat. Whether protection of marine soft sediments assemblages can result in the similar outcomes (e.g. more fish and/or larger fish) as can occur on other habitats, such as coral and rocky reefs, has rarely been explored.

The need for data on subtidal marine soft sediments have been identified as a key knowledge gap that is hindering the implementation of effective evidence based MPA management in the Australian state of New South Wales (Brooks et al. 2013). Jervis Bay Marine Park is one of six MPAs in the state. The park zoning came into effect on the 1st of October 2002, however, as in most MPAs, the potential impact of no-take zoning on demersal soft sediment fish abundances and diversity has not been assessed. Jervis Bay

is dominated by soft sediments and its waters are largely devoid of major human impacts like pollution and modification. Recreational fishing occurs in the majority of Jervis Bay with the exception of its no-take sanctuary zones, where no forms of fishing are permitted. These no-take zones are distributed haphazardly in relation to rocky reefs and seagrasses and replicated across the park. As a result, soft sediments are also well represented in fished and no-take areas. The dominance of soft sediment substrate, its pristine waters and replicated fished and unfished soft sediment areas means that Jervis Bay Marine Park provides a useful opportunity to gauge the impact of fishing and MPA implementation on demersal soft sediment fishes generally.

I hypothesised that the removal of fishing pressure on soft sediments in JBMP no-take zones will result in changes to fish abundance and size of fished species. More specifically, I tested the following three predictions: (1) That abundances of targeted and bycatch species would be greater in no-take zones; (2) that targeted species of fish in no-take zones would be larger than those in fished zones; and (3) there would be greater diversity in no-take zones compared to fished zones. I tested for these predicted effects of no-takes zones multiple times between 2011 and 2015. In relation to time, I predicted that the patterns outlined above would either be stable through time (i.e. indicating effects had already taken place and were stable) or that these patterns would be developing through time. As far as I am aware this is the first long-term study to test for the effects of no-take MPAs on marine soft sediment demersal fish assemblages across multiple years and multiple NTZs.

2.2 Methods

Jervis Bay Marine Park (JBMP) is located on the South-East coast of Australia (Fig. 2.1), covers an area of ~21,000 ha and includes most of the waters of Jervis Bay and a large area of open coast outside the Bay. A small section in the south of Jervis Bay is covered by the Commonwealth Waters of Booderee National Park. JBMP is a multiple zone reserve and is divided into several zones which came into effect on 1 October 2002 (Lynch 2006). No-take sanctuary zones (hereafter NTZs; IUCN category II - also equivalent to 'marine reserves' and 'no-take MPAs') in which all forms of fishing and extractive harvesting are prohibited make up approximately 20% of the park (4,253

hectares). Habitat protection zones (IUCN category IV) make up 72% (15,600 hectares) of the park, while general use zones (IUCN category VI) cover 8% (1,618 hectares). Recreational fishing and some very limited forms of commercial fishing (e.g. beach meshing and purse seining for pelagic bait species) are permitted in these 'fished zones' (FZ, these zones equivalent to partially protected areas in Chapter 1 & 3). Recreational fishing is now the main fishery in Jervis Bay and size and bag limits apply for most targeted species. Over 70% of the seafloor within Jervis Bay is covered by soft-sediments (Dames and Moore 1985) mostly in the form of sandy substrata (Fig. 2.1). Approximately 19% of these soft sediment habitats in the marine park are contained within no-take sanctuary zones (NSW MPA 2009).

Soft sediment associated flatheads (*Platycephalus* spp.) are the main species targeted and caught in large numbers by recreational fishers on soft sediments in JBMP. Several other species found on soft sediments are likely to make up a small but sizable proportion of the recreational catch. These include; shovelnose ray (*Aptychotrema rostrata*) which are taken in large quantities by recreational fishers state-wide (Rowling et al. 2010), eastern fiddler ray (*Trygonorrhina fasciata*) which are often caught but mostly discarded and Port Jackson shark (*Heterodontus portusjacksoni)* which are generally not targeted but are regularly taken as bycatch and released. The fishing effort in Jervis Bay was assessed prior to JBMP zoning implementation and on soft sediments was found to be relatively spatially homogenous (Lynch 2006). Post zoning implementation, fishing effort appears to have declined considerably across JBMP at a considerably higher level than would be predicted based on the displacement of fishing effort by the no-take zones put in place (Lynch 2014). However there has been no investigation of fishing effort inside Jervis Bay since 2009.
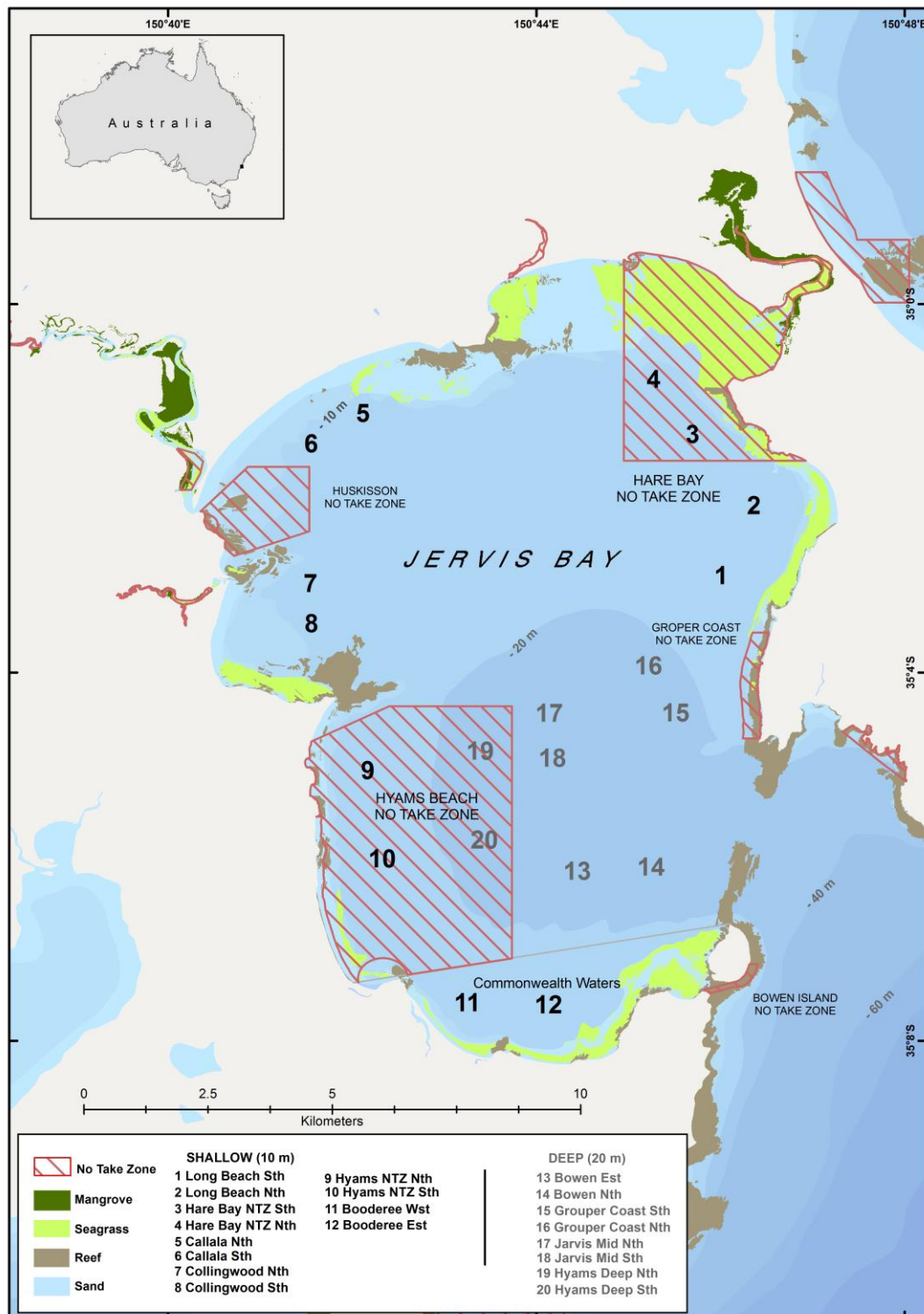
Figure 2.1: Map of study area in Jervis Bay Marine Park; including no-take zones, BRUV sampling sites and major habitat types. Subtidal features digitised preferentially from swath bathymetry, LADS and ADS40 aerial imagery. Sources: NSW DPI, NSW OEH, Geoscience Australia.

Baited remote underwater videos (BRUVs) were deployed to visually survey demersal fish assemblages found on marine soft sediments in Jervis Bay. A colour depth sounder, coastal charts and a drop camera were used to select unvegetated soft sediment habitat. Sampling was carried out in shallow water (10 m ± 2) in 2011, 2013 and 2015. Deeper waters (20 m ± 2) were sampled in 2011 and 2015. In 2011, half of the video samples were taken using single camera BRUV and the other half using stereo-BRUV (Fig. 2.2). In the following years, all samples were taken using stereo BRUV (hereafter BRUV refers to both single and stereo BRUV unless specified). The configuration of zones and distance to habitat other than soft sediments (to avoid halo effects) in Jervis Bay dictated the two depths and number of sites sampled. For example, it was only possible to sample two NTZ sites at 20 m and there are no NTZ covering sufficient soft sediments in the waters deeper than 30 m so no comparisons were made at those depths (Fig. 2.1). All sampling was carried out in May and June of each year across all tides. BRUVs were not deployed within an hour of dusk or dawn. Where possible, deployments that failed were repeated (i.e. landed facing the surface or seafloor, turned off during deployment or where visibility was very poor).

Each BRUV unit consisted of a galvanized steel frame with either one or two water-proof housings (Fig. 2.2) holding either a Canon HG21 or Canon HFG10 video camera. The stereo-BRUV had two cameras which were offset at an 8-degree inward angle and are separated by 0.8 m. This optimizes the field of view overlap between the cameras and allows accurate measurements to be taken within 9 m of the cameras for objects greater than 500 mm and within 5 m for objects less than 50 mm (Harvey et al. 2010). A detachable drop camera with a live feed to the surface was attached to the BRUV, to confirm habitat type and to check the BRUV was level, before being pulled free and retrieved (Fig. 2.2). A horizontal bait arm with bait bag containing 500 g of crushed pilchards (*Sardinops sagax*) as bait was attached to the BRUV when deployed. Bait was replaced on each deployment.
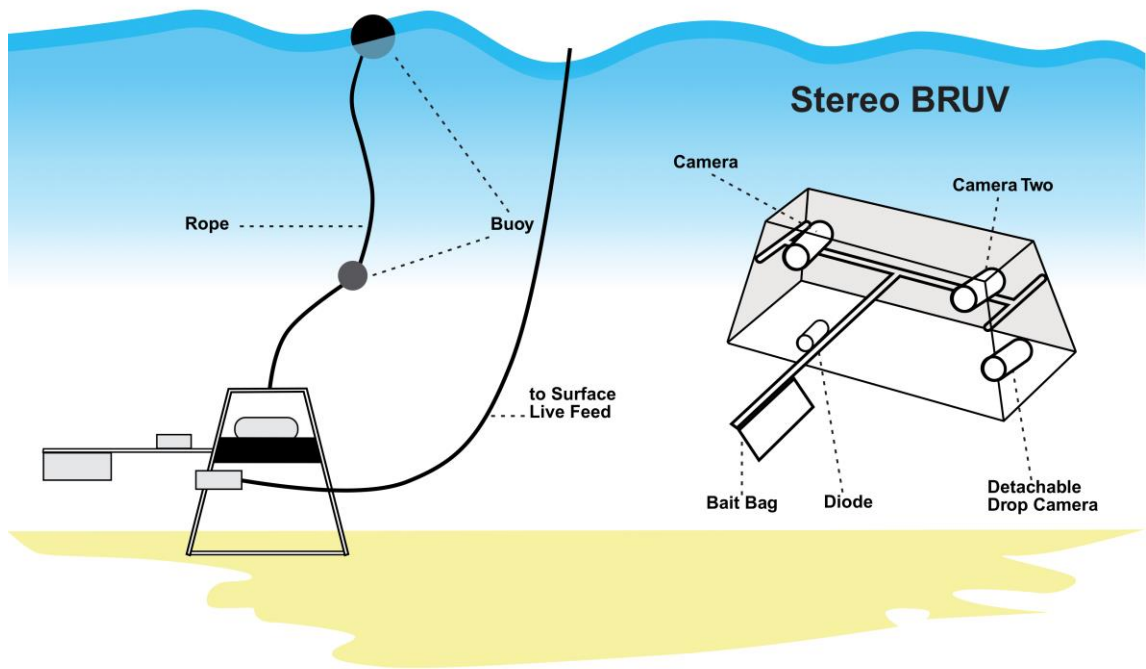
Figure 2.2: Left to right; Baited remote underwater video (BRUV) deployed on the seafloor, a stereo BRUV with diode for frame synchronisation between cameras (Figure from Fetterplace and Rees 2017, CC-BY).

At each sample site, the BRUV was deployed for a bottom "soak time" of 35 minutes. Each deployment was a minimum of 200 m from reefs to reduce the chances of sampling reef associated fishes i.e. to avoid halo effects around reefs where assemblages may be closer to reef assemblages than soft sediment assemblages (Schultz et al. 2012). Four BRUV units at each location were deployed within 5–10 minutes of each other and a minimum distance of 200 m was kept between replicates. This separation distance and soak time is consistent with the BRUV literature (Whitmarsh et al. 2017). Furthermore, Jervis Bay typically has very low flow rates of <1.5 cm $^{s-1}$ (Holloway 1995) and bait plumes at this speed would likely only travel <30 m (< 1.5 cm $^{s-1}$ * 60 secs * 30 mins = 2700/100 = 27 m) in a 30 minute deployment. Given that currents would need to be more than 7x this speed to disperse the bait plume 200 m, I considered a conservative 200 m separation distance to be more than adequate to achieve replicate independence.

In the laboratory, video footage from each BRUV deployment was processed using Event Measure software (Seager 2011). Sampling was conducted with two different camera types; Canon HG21s and HFG10s. HFG10 Canon cameras have a larger field of

view than the HG21's and we standardised the field of view by reducing the HFG10s to closely match that of the HG21s. To do this, the field of view was reduced to 81% of the original in EventMeasure for all HFG10 videos being analysed. Only fish that were within 4 m of the camera were included in counts in order to standardise depth of view across samples. This distance was measured in EventMeasure using the epipolar function. By synchronizing the right and left cameras in each stereo BRUV deployment and calibrating regularly using the CAL program (Seager 2011), distances and fish lengths were also able to be accurately estimated in EventMeasure. For the few single BRUVs in 2011 where visibility was greater than 4 m, the distance was estimated based on the known length of the bait arm.

Analysis of each deployment started from the time the BRUV landed on the sea floor (settlement time) and lasted for 30 minutes. Thirty minutes was selected as a number of studies have found that the peak number of fish recorded is between 20–30 min (e.g. Willis and Babcock 2000, Stobart et al. 2007), and for demersal fish Misa *et al.* (2016) suggested that a set time of 15 minutes was the shortest set length able to capture reliable stereo video metrics. We opted for the upper end of this time frame both because this time was consistent with other studies on reef in the study region (Malcolm et al. 2007, Wraith et al. 2013, Coleman et al. 2015), and because previous studies looking at set time have focused predominately on reef species rather than soft sediments; a more conservative set time was considered prudent. Increasing analysis times beyond thirty minutes was unlikely to result in differences in abundance metrics (Willis and Babcock 2000, Harasti et al. 2015b, Misa et al. 2016).

Each species entering the field of view was identified and recorded. Relative abundance, in this case the maximum number of each individual species in a frame at one time (MaxN), (Cappo et al. 2003) and frequency of occurrence were also recorded (percentage of replicates each species was recorded on). A total MaxN combining all species for each drop was also calculated by summing the MaxN from each species (Willis and Babcock 2000). A number of studies have found that the relative abundance measured by MaxN correlated with fish abundance (Ellis and DeMartini 1995, Willis et al. 2001) and although a conservative approach, ensures that fish are not repeatedly counted (Willis and Babcock 2000, Cappo et al. 2004). The total length (TL, from the tip

of snout to centre of the caudal fin) of individual fish were measured using Event Measure. Where possible, sizes were measured from close to the MaxN frame to ensure that fish were not sized more than once.

It can be difficult to consistently differentiate between some fish species (or life stages of different species) using underwater video alone. In this study, it was often impossible to separate juvenile bluespotted flathead (*Platycephalus caeruleopunctatus*) and longspine flathead (*Platycephalus grandispinis*) with certainty, so a genus level *Platycephalus* spp. MaxN (MaxN at frame with most flathead of any *Platycephalus* species) was taken. Both of these species were present in the study area as confirmed by line fishing and some occasions on BRUV when differentiation was clear. Genus level length measurements were taken at the *Platycephalus* spp. MaxN. Above ~20 cm *P. caeruleopunctatus* are clearly identifiable on video based on tail markings and therefore a MaxN and separate length measurement for adult *P. caeruleopunctatus* was also recorded.

Experimental design

Comparisons were made in shallow water (10 m ± 2 m) and deep water (20 m ± 2 m) and an asymmetrical sampling design was used at both depths (Fig. 2.3 and 2.4). Glasby (1997) proposed using asymmetrical analyses for examining post-impact data from a single disturbed location(s) and multiple undisturbed controls. The use of these fully replicated asymmetrical designs reduces problems of spatial confounding where no pre-data is available. This method has been argued to be the most effective means of evaluating species responses to MPAs where multiple fished zones (FZ) are treated as controls and the removal of fishing in a single (or multiple) no-take MPA as the treatment (Hoskin et al. 2011, Caveen et al. 2012).
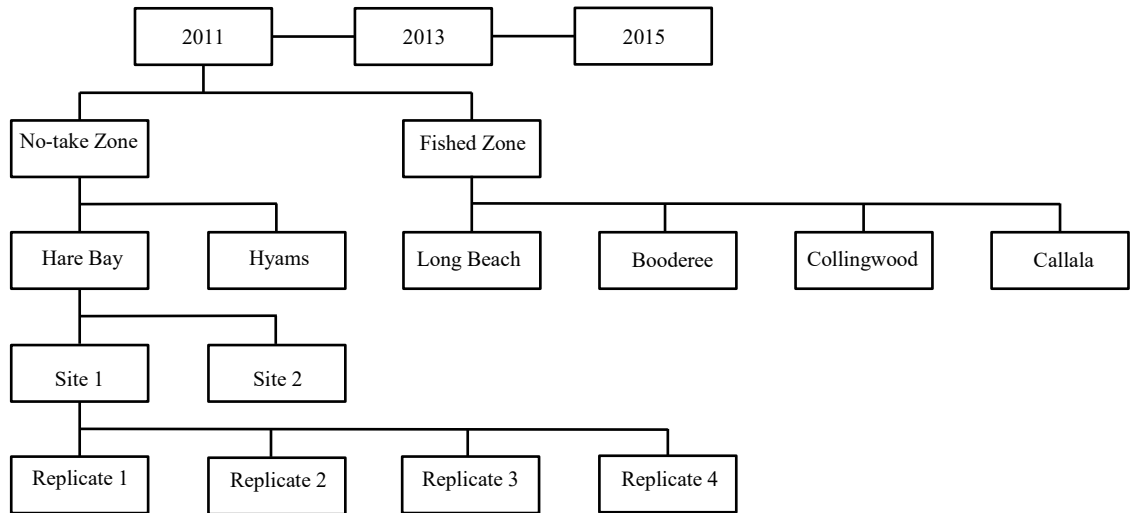
Figure 2.3: Asymmetrical experimental design to assess ecological changes in the diversity, relative abundance and size of soft sediment fishes in Jervis Bay in shallow water (10 m).
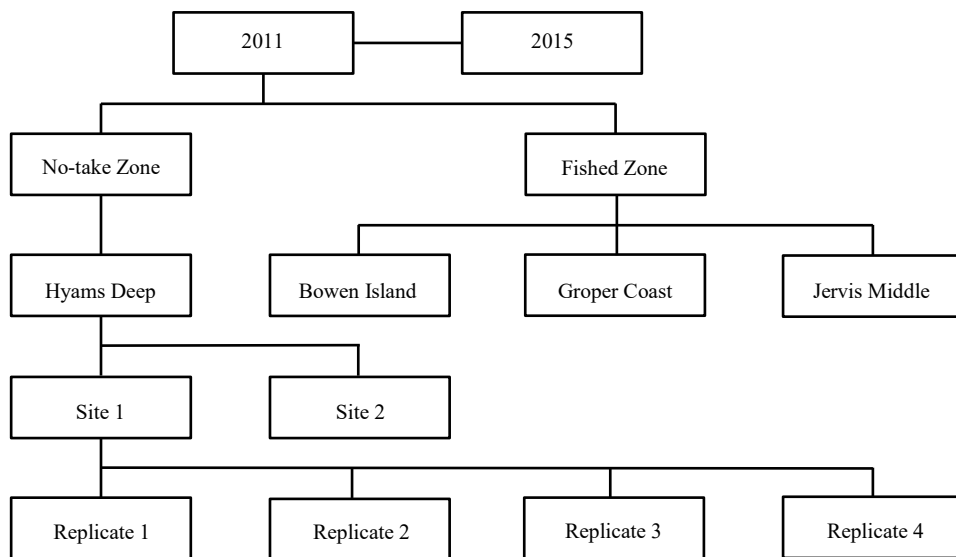
Figure 2.4: Asymmetrical experimental design to assess ecological changes in the diversity, relative abundance and size of soft sediment fishes in Jervis Bay in deep water (20 m).

In the shallow water, the design had four factors: Year (a random orthogonal factor with 3 levels: 2011, 2013 & 2015), Zone (a fixed orthogonal factor with 2 levels, NTZ and FZ), Location (a random nested factor with 2 levels in NTZ and 4 levels in FZ;

with Location nested in Zone), Site (a random nested factor with 2 levels; nested in Location). Each site had 4 replicated BRUV samples. A total of 144 BRUV samples were taken in shallow water, with 48 replicate video samples obtained from each year; 16 within the two NTZ locations in each year and 32 from within three FZ locations in each year (Fig. 2.3). In deep water, the design had four factors; Year (a random orthogonal factor with 2 levels: 2011 & 2015), Zone (a fixed orthogonal factor with 2 levels, NTZ and FZ), Location (a random nested factor with 1 level in NTZ and 3 levels in FZ; with Location nested in Zone), Site (a random nested factor with 2 levels; nested in Location). Each site had 4 replicated BRUV samples. A total 64 BRUV samples were taken in deep water with 32 replicate video samples obtained from each year; 8 within the NTZs in each year and 24 from three FZs in each year (Fig. 2.4).

Of the planned BRUV deployments undertaken, a number failed due to poor visibility, equipment issues or being tipped over mid deployment and I resampled these 'failed' deployments. After resampling, a total of 126 shallow samples (Table 2.1) were deemed successful and analysed. In deep water, only one sample failed (at Hyams Deep Nth) and 63 samples were analysed (Table 2.1). The resulting designs were unbalanced due to missing cells, however by using permutation methods to obtain mean square values and construct appropriate pseudo-f ratios, issues with missing values in the dataset can be effectively overcome (Anderson et al. 2008, Zintzen et al. 2012).

Statistical Analyses

All multivariate and univariate analyses of abundance data were carried out using PERMANOVA analyses (Clarke 1993, Anderson, Gorley et al. 2008) in PRIMER-E v7 using type III sums of squares, 9999 permutations and the design given above. Predictions about multivariate differences in assemblages across management zones were tested using Bray-Curtis dissimilarity values, and a visual indication of assemblage patterns was provided by using Non-Metric Multi-Dimensional Scaling ordination. The two highly abundant pelagic species, yellowtail scad (*Trachurus novaezelandiae*) and slimy mackerel (*Scomber australasicus*), were excluded from multivariate analysis of community composition and univariate Total MaxN analyses as they are not considered benthic species (i.e. soft sediment fishes) and I was concerned that they may have a disproportionate effect on the data set, owing to highly variable numbers (i.e. hundreds

on some BRUV deployments but few or none on most). They were retained in species richness counts. In addition to multivariate comparisons of community composition, I calculated two diversity indices, Shannon Diversity (the exponential of Shannon entrophy, Jost 2006) and Pielou's evenness measure (Jost 2010), using the "Vegan" package in "R" (Oksanen et al. 2018) for each BRUVS replicate (see supporting information 2.5 for details of each index). Euclidean distance was used as the measure of dissimilarity for univariate analyses comparing Total MaxN, species richness, diversity indices and the relative abundance of individual species that met a frequency of occurrence threshold by appearing on 25% or greater of all BRUV samples across years and management zones.

Table 2.1: Site and number of successful BRUV deployments in each year and at shallow (10 m) and deep (20 m) depths. All deployments are stereo camera BRUVs unless "single" camera BRUV is indicated.

10 metres	Location	2011	2013	2015
Fished	LongBeach Nth	4 (2 single)	4	4
	LongBeach Sth	4 (2 single)	4	4
	Booderee Est	4 (2 single)	4	4 (1 Single)
	Booderee Wst	4 (2 single)	4	4
	Collingwood Nth	4 (2 single)	0	4
	Collingwood Sth	3 (2 single)	0	3
	Callala Nth	4 (2 single)	3	4
	Callala Sth	3 (2 single)	2	0
No- Take	Hare Bay Nth	4 (2 single)	4	4
	Hare Bay Sth	4 (2 single)	4	4
	Hyams Nth	4 (2 single)	4	4
	Hyams Sth	4 (2 single)	4	4
Total completed		46/48 (24 single)	37/48	43/48
20 metres	**Location**	**2011**	**2013**	**2015**
Fished	Bowen Island East	4 (2 single)	x	4
	Bowen Island West	4 (2 single)	x	4
	Groper Coast Nth	4 (2 single)	x	4
	Grouper Coast Sth	4 (2 single)	x	4
	Jervis Bay Middle Nth	4 (2 single)	x	4
	Jervis Bay Middle Sth	4 (2 single)	x	4
No- Take	Hyams Deep Nth	3 (2 single)	x	4
	Hyams Deep Sth	4 (2 single)	x	4
Total completed		31/32 (16 single)	x	32/32

To increase the power of the main tests, lower order terms were pooled when p >0.25 (Underwood 1997). For univariate comparisons with significant terms of interest (zone × year or zone effects) PERMDISP was used to test homogeneity of variance, a test that is equivalent to Levene's test for heterogeneity (Anderson 2006, Harvey et al. 2012) and where both significant PERMANOVA and PERMDISP p-values were obtained, the data was fourth root transformed to eliminate or reduce the significant dispersion result and the data reanalysed. Post hoc pairwise comparisons were made on zone × year interaction in the model that were statistically significant in the main PERMANOVA analysis. Monte Carlo random draws were used to obtain p-values where sufficient permutations were not available in pair wise analyses (Anderson et al. 2008).

I also tested whether differences in mean length and shape of the size distribution were different in NTZs compared to FZ. I tested for differences in the cumulative length distribution of flathead across zones using the two sample non-parametric Kolmogorov–Smirnov (KS) test (For a detailed description of the KS test see Langlois et al. 2012). KS test were conducted in R (R Development Core Team 2014) using the 'ks.test' function in the package 'dgof' (Arnold and Emerson 2011). Our data contained no ties which enabled exact p-values to be calculated without the need for bootstrapping (Ogle 2016). In shallow water, lengths for flathead at the genus level from all study years (2011, 2013, and 2015) were aggregated by zone type (FZ vs. NTZ) and for visual comparison of length frequency distributions, 2 cm length intervals were selected. Adult *P. caeruleopunctatus* lengths were compared using the same processes described however only on lengths from 2013 and 2015. In deep water, lengths for flathead at the genus level from both study years (2011 and 2015) were aggregated by zone type (FZ vs. NTZ) and for visual comparison of length frequency distributions, 2 cm length intervals were selected. Adult *P. caeruleopunctatus* lengths were compared using the same processes described however only on lengths from 2015.

2.3 Results

The demersal soft sediment fish assemblages in both shallow (10 m) and deep (20 m) water were characterised by a few frequently occurring species. Flathead species (*Platycephalus* spp.; *P. caeruleopunctatus* & *P. grandispinis*) and eastern fiddler ray

(*Trygonorrhina fasciata*) were the most commonly recorded demersal species in both shallow (Table 2.2) and deep water (Table 2.3). In shallow water, eight demersal species appeared on 25% or more of the total deployments and both NTZ and FZ shared the same most common species (Table 2.2). Similarly, in deep water, six species appeared on 25% or more of the total deployments and the most common species were found in both NTZs and FZs (2.3).

In shallow water, there were 37 fish species observed across the three years sampled (Table 2.4); 19 species were seen in all three sampling years, 5 in two years and 14 in just one of the years. In addition, 15 of these were only encountered in one or two deployments (singletons and doubletons). In deep water, there were a total of 29 species recorded across the two years I sampled (Table 2.5); 16 of which were seen in both sampling years and 13 in only one year. Eleven of the 29 species were only encountered in one or two deployments (Table 2.5). The abundance dataset was also characterized by a few dominant taxa. Eighty two percent of the total abundance was made up by five species in shallow habitat (Table 2.4) and four species in the deeper habitat (Table 2.5).

The demersal fish assemblages were similar between fished zones (FZs) and no-take zones (NTZs) within each year in both in shallow water (Table 2.6, Fig. 2.5), and in deep water (Table 2.6, Fig. 2.6). The lack differences between assemblages among zones were also reflected in frequency of occurrence (Table 2.2 and Table 2.3), species richness, Shannon diversity and Pielou's evenness measure in shallow (Fig. 2.7, Table 2.7) and deep water (Fig. 2.8, Table 2.7). In shallow water, there appeared to be a trend towards increasing total abundances (Total MaxN) through time and the trend appears strongest in the NTZ locations and Long Beach in the FZ, and did not appear to occur in the other FZ locations (Fig. 2.7), never the less the differences were not significant among zones. In deep water, there were no differences among zones and both NTZ and FZ locations showed increased abundances through time (Fig. 2.8).

Table 2.2: Frequency of occurrence (% of samples present) for each species at 10 m depth for all BRUV deployments (n = 126) and when categorised by no-take zone (NTZ, n = 48) and fished zone (FZ, n = 78). Hatched line indicates frequency cut off point for univariate analyses. See Table 2.4 for scientific names.

All		NTZ		FZ	
Flathead	95.2	Flathead	97.9	Flathead	93.6
Fiddler Ray	88.9	Fiddler Ray	91.7	Fiddler Ray	87.2
School Whiting	38.1	Stingaree	50.0	School Whiting	44.9
Port Jackson Shark	38.1	Bluespotted Flathead	45.8	Port Jackson Shark	39.7
Stingaree	35.7	Shovelnose Ray	37.5	Bluespotted Flathead	28.2
Bluespotted Flathead	34.9	Port Jackson Shark	35.4	Shovelnose Ray	28.2
Shovelnose Ray	31.7	School Whiting	27.1	Stingaree	26.9
Ocean Jacket	23.0	Ocean Jacket	22.9	Ocean Jacket	23.1
Southern Eagle Ray	19.0	Southern Eagle Ray	18.8	Weeping Toadfish	20.5
Silver Trevally	15.9	Silver Trevally	10.4	Southern Eagle Ray	19.2
Weeping Toadfish	13.5	Bonito	8.3	Silver Trevally	19.2
Bonito	11.1	Eastern Smooth Boxfish	6.3	Snapper	15.4
Snapper	9.5	Globe Fish	6.3	Bonito	12.8
Eastern Smooth Boxfish	7.1	Eastern Fortescue	4.2	Eastern Smooth Boxfish	7.7
Globe Fish	6.3	Smalltooth Flounder	4.2	Globe Fish	6.4
Australian Goatfish	4.0	Australian Salmon	4.2	Australian Goatfish	5.1
Eastern Fortescue	3.2	Short-tail Stingray	4.2	Spotted Grubfish	3.8
Smalltooth Flounder	2.4	Weeping Toadfish	2.1	Eastern Fortescue	2.6
Spotted Grubfish	2.4	Australian Goatfish	2.1	Tailor	2.6
Tailor	2.4	Tailor	2.1	Eastern Striped Grunter	2.6
Eastern Striped Grunter	1.6	Baitfish	2.1	Australian Mado	2.6
Australian Mado	1.6	Longfin Pike	2.1	Smalltooth Flounder	1.3
Australian Salmon	1.6	Flagtail Flathead	2.1	Baitfish	1.3
Short-tail Stingray	1.6	Estuary Cobbler	2.1	Australian Anchovy	1.3
Baitfish	1.6	Snapper	0.0	Mulloway	1.3
Australian Anchovy	0.8	Spotted Grubfish	0.0	Yellowfin Bream	1.3
Longfin Pike	0.8	Eastern Striped Grunter	0.0	Smooth Toadfish	1.3
Flagtail Flathead	0.8	Australian Mado	0.0	Common Toadfish	1.3
Estuary Cobbler	0.8	Australian Anchovy	0.0	Gummy Shark	1.3
Mulloway	0.8	Mulloway	0.0	Baitfish 2	1.3
Yellowfin Bream	0.8	Yellowfin Bream	0.0	Australian Salmon	0.0
Smooth Toadfish	0.8	Smooth Toadfish	0.0	Short-tail Stingray	0.0
Common Toadfish	0.8	Common Toadfish	0.0	Longfin Pike	0.0
Gummy Shark	0.8	Gummy Shark	0.0	Flagtail Flathead	0.0
Baitfish 2	0.8	Baitfish 2	0.0	Estuary Cobbler	0.0

Table 2.3: Frequency of occurrence (% of samples recorded on) for each species at 20 metres depth for all BRUV deployments (n = 63) and when categorised by no-take zone (NTZ, n = 15) and fished zone (FZ, n = 48). Hatched line indicates frequency cut off point for univariate analyses. See Table 2.5 for scientific names.

All		NTZ		FZ	
Flathead	100.0	Flathead	100.0	Flathead	100.0
Fiddler Ray	88.9	Bluespotted Flathead	100.0	Fiddler Ray	87.5
Bluespotted Flathead	79.4	Fiddler Ray	93.3	Bluespotted Flathead	72.9
School Whiting	57.1	School Whiting	66.7	School Whiting	54.2
Port Jackson Shark	55.6	Port Jackson Shark	60.0	Port Jackson Shark	54.2
Shovelnose Ray	44.4	Shovelnose Ray	33.3	Shovelnose Ray	47.9
Eastern Fortescue	22.2	Eastern Fortescue	26.7	Eastern Fortescue	20.8
Southern Eagle Ray	17.5	Southern Eagle Ray	20.0	Eastern Smooth Boxfish	20.8
Eastern Smooth Boxfish	15.9	Silver Trevally	20.0	Southern Eagle Ray	16.7
Silver Trevally	14.3	Stingaree	20.0	Silver Trevally	12.5
Stingaree	12.7	Tailor	20.0	Ocean Jacket	12.5
Ocean Jacket	11.1	Australian Goatfish	13.3	Stingaree	10.4
Australian Goatfish	9.5	Australian Mado	13.3	Australian Goatfish	8.3
Tailor	7.9	Snapper	13.3	Bonito	8.3
Bonito	7.9	Ocean Jacket	6.7	Gummy Shark	8.3
Gummy Shark	6.3	Bonito	6.7	Tailor	4.2
Short-tail Stingray	4.8	Short-tail Stingray	6.7	Short-tail Stingray	4.2
Australian Mado	3.2	Yellowfin Bream	6.7	Australian Salmon	2.1
Snapper	3.2	Eastern Striped Grunter	6.7	Globe Fish	2.1
Yellowfin Bream	1.6	Eastern Smooth Boxfish	0.0	Rough Flutemouth	2.1
Eastern Striped Grunter	1.6	Gummy Shark	0.0	Smalltooth Flounder	2.1
Australian Salmon	1.6	Australian Salmon	0.0	Samson Fish	2.1
Globe Fish	1.6	Globe Fish	0.0	Weeping Toadfish	2.1
Rough Flutemouth	1.6	Rough Flutemouth	0.0	Australian Mado	0.0
Smalltooth Flounder	1.6	Smalltooth Flounder	0.0	Yellowfin Bream	0.0
Samson Fish	1.6	Samson Fish	0.0	Snapper	0.0
Weeping Toadfish	1.6	Weeping Toadfish	0.0	Eastern Striped Grunter	0.0

Table 2.4: All species recorded at 10 metres depth, their average MaxN per BRUV deployment in each zone (NTZ and FZ) and their total relative abundance (or Total Count) recorded by BRUV per year and total for all three years.

Common Name	Family	Genus	Species	Average MaxN		Total Count			
				FZ	NTZ	Total	2011	2013	2015
Ocean Jacket	Monacanthidae	*Nelusetta*	*ayraudi*	7.96	1.67	701	1	693	7
Flathead*	Platycephalidae	*Platycephalus*	All	4.90	6.48	693	224	242	227
School Whiting	Sillaginidae	*Sillago*	*flindersi*	2.24	3.19	328	16	87	225
Fiddler Ray	Rhinobatidae	*Trygonorrhina*	*fasciata*	1.87	2.58	270	87	70	113
Silver Trevally	Carangidae	*Pseudocaranx*	*georgianus*	1.24	0.31	112	11	78	23
Bluespotted Flathead**	Platycephalidae	*Platycephalus*	*caeruleopunctatus*	0.40	0.73	66	20	19	27
Port Jackson Shark	Heterodontidae	*Heterodontus*	*portusjacksoni*	0.47	0.48	60	28	21	11
Weeping Toadfish	Tetraodontidae	*Torquigener*	*pleurogramma*	0.71	0.04	57	30	3	24
Shovelnose Ray	Rhinobatidae	*Aptychotrema*	*rostrata*	0.32	0.54	51	26	12	13
Stingaree***	Urolophidae spp.	All	All	0.29	0.58	51	15	12	24
Striped Grunter	Terapontidae	*Pelates*	*sexlineatus*	0.51	0.00	40	40	0	0
Southern Eagle Ray	Myliobatidae	*Myliobatis*	*tenuicaudatus*	0.22	0.19	26	4	8	14
Snapper	Sparidae	*Pagrus*	*auratus*	0.33	0.00	26	12	12	2
Bonito	Scombridae	*Sarda*	*australis*	0.19	0.10	20	0	20	0
Australian Mado	Scorpididae	*Atypichthys*	*strigatus*	0.23	0.00	18	13	0	5
Australian Goatfish	Mullidae	*Upeneus*	*sp*	0.06	0.08	9	1	5	3
Smooth Boxfish	Ostraciidae	*Anoplocapros*	*inermis*	0.08	0.06	9	6	2	1
Globe Fish	Diodontidae	*Dicotylichthys*	*punctulatus*	0.06	0.06	8	4	4	0
Eastern Fortescue	Tetrarogidae	*Centropogon*	*australis*	0.03	0.04	4	3	0	1
Australian Anchovy	Engraulidae	*Engraulidae*	*sp*	0.04	0.00	3	3	0	0
Smalltooth Flounder	Paralichthyidae	*Pseudorhombus*	*jenynsii*	0.01	0.04	3	1	1	1
Spotted Grubfish	Pinguipedidae	*Parapercis*	*ramsayi*	0.04	0.00	3	0	2	1
Tailor	Pomatomidae	*Pomatomus*	*saltatrix*	0.03	0.02	3	0	0	3
Australian Salmon	Arripidae	*Arripis*	*trutta*	0.00	0.04	2	0	0	2
Short-tail Stingray	Dasyatidae	*Bathytoshia*	*brevicaudata*	0.00	0.04	2	0	1	1
Baitfish				0.01	0.02	2	0	1	1
Longfin Pike	Dinolestidae	*Dinolestes*	*lewini*	0.00	0.02	1	0	1	0
Flagtail Flathead	Platycephalidae	*Platycephalus*	*endrachtensis*	0.00	0.02	1	1	0	0
Estuary Cobbler	Plotosidae	*Cnidoglanis*	*macrocephalus*	0.00	0.02	1	0	1	0
Mulloway	Sciaenidae	*Argyrosomus*	*japonicus*	0.01	0.00	1	1	0	0
Yellowfin Bream	Sparidae	*Acanthopagrus*	*australis*	0.01	0.00	1	1	0	0
Smooth Toadfish	Tetraodontidae	*Torquigener*	*glaber*	0.01	0.00	1	1	0	0
Common Toadfish	Tetraodontidae	*Tetractenos*	*hamiltoni*	0.01	0.00	1	0	0	1
Gummy Shark	Triakidae	*Mustelus*	*antarcticus*	0.01	0.00	1	1	0	0
Baitfish 2				0.01	0.00	1	1	0	0
			Total per year			2576	551	1295	730
			Total species count[#]			37	27	24	25

* Genus level count including *P. caeruleopunctatus* and all *P. grandispinis*
** Adult blue-spotted flathead only.
*** Family level count; most likely *Trygonoptera testacea* but may include *Urolophus sufflavus, Urolophus kapalensis* and *Urolophus cruciatus.*
[#] Includes *Scomber australasicus* and *Trachurus novaezelandiae* in species richness and total species count.

Table 2.5: All species recorded at 20 metres depth, their average MaxN per BRUV deployment in each zone (NTZ and FZ) and their total relative abundance (or Total Count) recorded by BRUV per year and total for all years.

Common Name	Family	Genus	Species	Average MaxN		Total Count		
				FZ	NTZ	Total	2011	2015
Flathead*	Platycephalidae	*Platycephalus*	All	7.52	6.27	455	207	248
School Whiting	Sillaginidae	*Sillago*	*flindersi*	6.44	7.33	419	34	385
Fiddler Ray	Rhinobatidae	*Trygonorrhina*	*fasciata*	2.31	2.87	154	58	96
Bluespotted Flathead**	Platycephalidae	*Platycephalus*	*caeruleopunctatus*	1.27	1.60	84	48	36
Port Jackson Shark	Heterodontidae	*Heterodontus*	*portusjacksoni*	0.71	0.73	45	26	19
Silver Trevally	Carangidae	*Pseudocaranx*	*georgianus*	0.77	0.40	43	2	41
Shovelnose Ray	Rhinobatidae	*Aptychotrema*	*rostrata*	0.67	0.33	37	24	13
Eastern Fortescue	Tetrarogidae	*Centropogon*	*australis*	0.46	0.47	29	22	7
Australian Mado	Scorpididae	*Atypichthys*	*strigatus*	0.00	1.13	17	0	17
Southern Eagle Ray	Myliobatidae	*Myliobatis*	*tenuicaudatus*	0.19	0.27	13	7	6
Eastern Smooth Boxfish	Ostraciidae	*Anoplocapros*	*inermis*	0.23	0.00	11	10	1
Ocean Jacket	Monacanthidae	*Nelusetta*	*ayraudi*	0.19	0.07	10	5	5
Stingaree***	Urolophidae spp.	All	All	0.10	0.20	8	1	7
Australian Goatfish	Mullidae	*Upeneus*	*sp*	0.08	0.13	6	0	6
Tailor	Pomatomidae	*Pomatomus*	*saltatrix*	0.04	0.20	5	0	5
Bonito	Scombridae	*Sarda*	*australis*	0.08	0.07	5	4	1
Gummy Shark	Triakidae	*Mustelus*	*antarcticus*	0.08	0.00	4	3	1
Short-tail Stingray	Dasyatidae	*Bathytoshia*	*brevicaudata*	0.04	0.07	3	1	2
Yellowfin Bream	Sparidae	*Acanthopagrus*	*australis*	0.00	0.13	2	0	2
Snapper	Sparidae	*Pagrus*	*auratus*	0.00	0.13	2	0	2
Eastern Striped Grunter	Terapontidae	*Pelates*	*sexlineatus*	0.00	0.13	2	0	2
Australian Salmon	Arripidae	*Arripis*	*trutta*	0.02	0.00	1	0	1
Globe Fish	Diodontidae	*Dicotylichthys*	*punctulatus*	0.02	0.00	1	1	0
Rough Flutemouth	Fistulariidae	*Fistularia*	*petimba*	0.02	0.00	1	1	0
Smalltooth Flounder	Paralichthyidae	*Pseudorhombus*	*jenynsii*	0.02	0.00	1	0	1
Samson Fish	Carangidae	*Seriola*	*hippos*	0.02	0.00	1	1	0
Weeping Toadfish	Tetraodontidae	*Torquigener*	*pleurogramma*	0.02	0.00	1	1	0
					Total per year	1360	456	904
					Total species count[#]	29	21	24

* Genus level count including *P. caeruleopunctatus* and all *P. grandispinis*
** Adult bluespotted flathead only.
*** Family level count; most likely *Trygonoptera testacea* but may include *Urolophus sufflavus, Urolophus kapalensis* and *Urolophus cruciatus.*
[#] Includes *Scomber australasicus* and *Trachurus novaezelandiae* in species richness and total species count.

Table 2.6: Results of assemblage comparisons across zones (fished and no-take) at a) 10 metres depth, and b) 20 metres depth, using permutational analysis of variance (PERMANOVA). 10 m depth factors: Year (Ye, random, 3 levels: 2011, 2013 and 2015), Zones (Zo, fixed, 2 levels: fished and no-take), locations (Lo, random,) and sites (Si). 20 m depth factors are the same except Year (Ye, random, 2 levels: 2011 and 2015). Significant P values (<0.05) shown in bold.

Source	df	MS	Pseudo-F	P(perm)	df	MS	Pseudo-F	P(perm)
		a) 10 m			b) 20 m			
Ye	2	9236	2.45	**0.031**	1	17310	5.18	0.072
Zo	1	6208	1.50	0.160	1	992	1.40	0.344
Lo(Zo)	4	4249	0.96	0.545	2	2157	0.68	0.818
YexZo	2	2140	0.58	0.856	1	1032	0.31	0.770
Si(Lo(Zo))	6	2459	1.52	0.065	4	1249	1.19	0.328
YexLo(Zo)**	7	3680	2.27	**0.003**	2	3498	3.27	**0.032**
YexSi(Lo(Zo))**	10	1632	1.44	**0.013**	4	1053	1.27	0.181
Res	93	1133			47	831		
Total	125				62			

** Term has one or more empty cells.
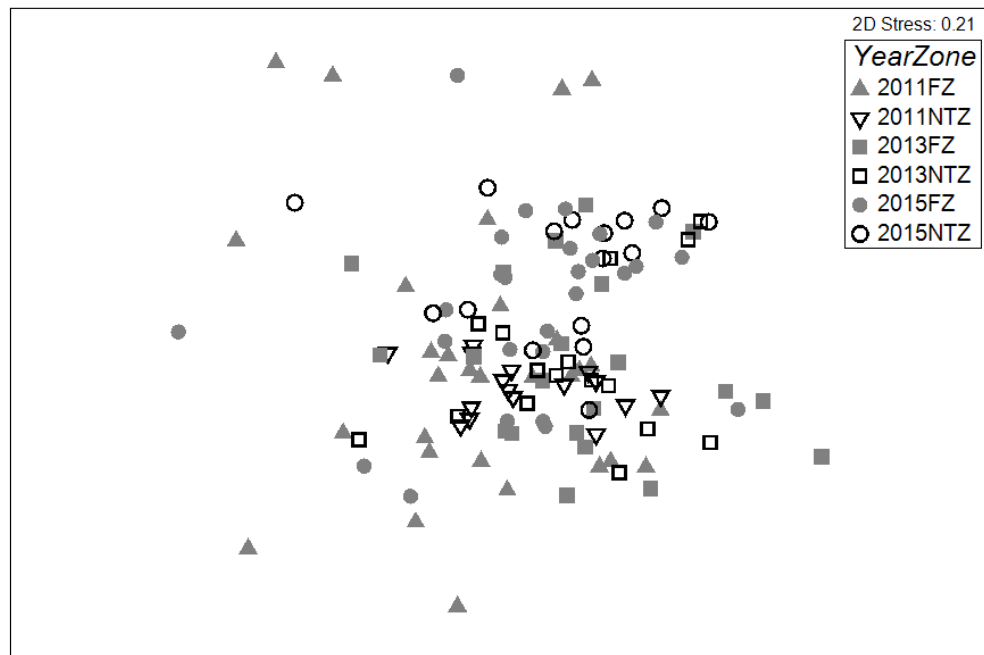
Figure 2.5: Differences in soft sediment fish assemblages between zones in JBMP at 10 m depth shown by nonmetric multidimensional scaling (MDS) plot based on Bray-Curtis similarities. 2011 (n=46), 2013 (n =37), 2015 (n=43).
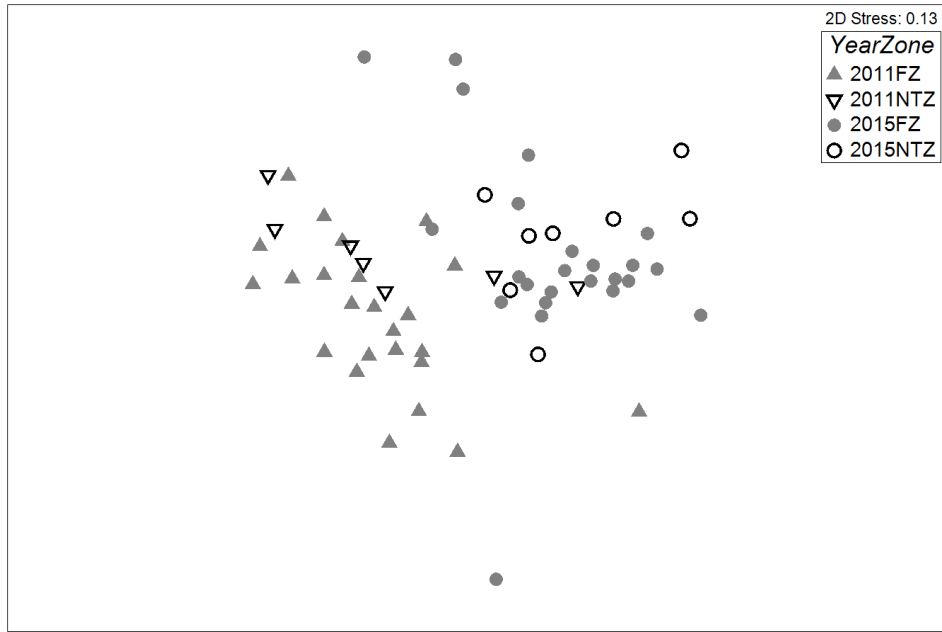
Figure 2.6: Differences in soft sediment fish assemblages between zones in JBMP at 20 m depth shown by nonmetric multidimensional scaling (MDS) plot based on Bray-Curtis similarities. 2011 (n=31) and 2015 (n=32).
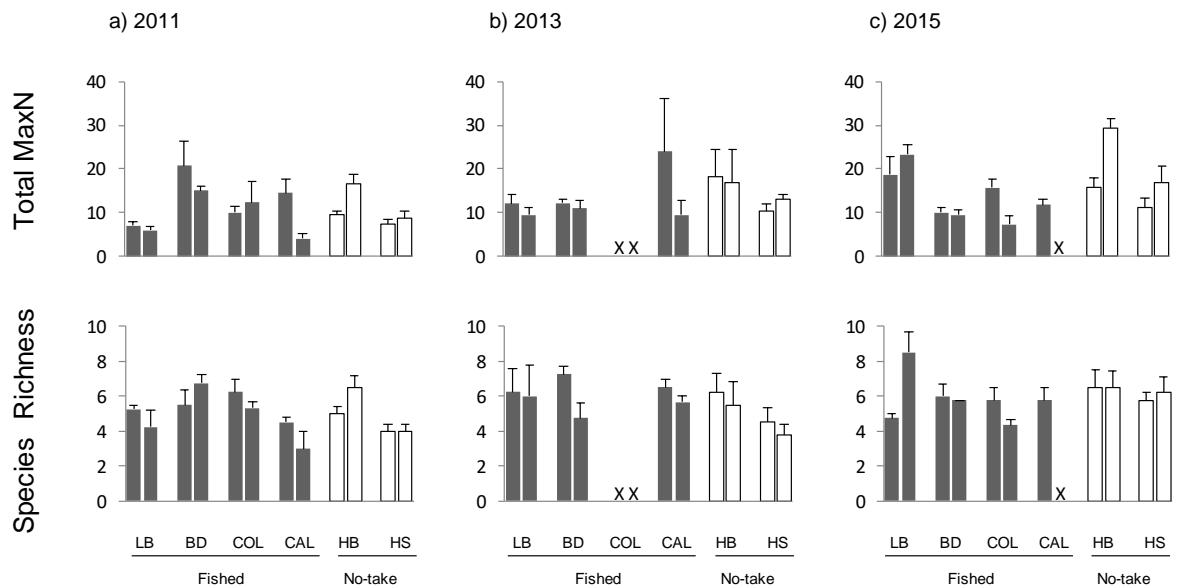
Figure 2.7: Average 10 m total abundance (Total MaxN) and average species richness in a) 2011, b) 2013 and c) 2015 at each site (n= 4 replicates per site). Error bars are SE. No-take sanctuary zones are in white, recreationally fished habitat protection zones in grey. X indicates no data. LB = Long
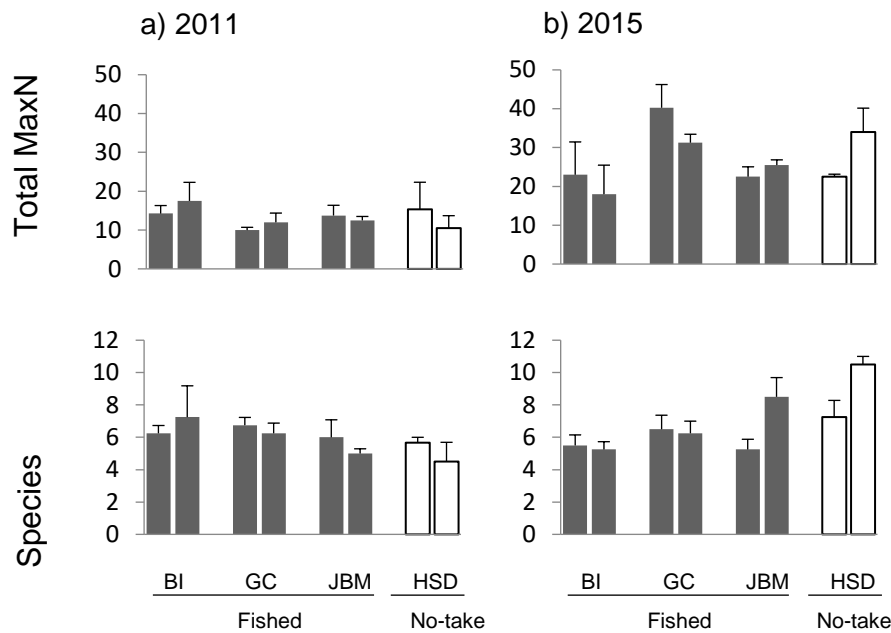
Figure 2.8: Average total abundance (Total MaxN) and average species richness at 20 metres depth in a) 2011 and b) 2015 at each site (n= 4 replicates per site). Error bars are SE. No-take sanctuary zones are in white, recreationally fished habitat protection zones in grey. X indicates no data. BI = Bowen Island, GC = Grouper Coast, JBM = Jervis Bay Mid, HSD = Hyams Sanctuary Deep. Each of these locations has two sites.

In contrast to the overall assemblage, zoning effects were apparent for some individual taxa in shallow water. Abundances of *Platycephalus* spp. showed a significant effect of zone (Table 2.8) and abundances were more stable NTZs across time (Fig. 2.9a). Overall, there was a 32% greater abundance of *Platycephalus* spp. in NTZs compared to FZs. I also detected a significant year x zone interaction for *T. fasciata* and shovelnose ray (*Aptychotrema rostrata*) (Table 2.8). Both species were more abundant in NTZs compared to FZs in 2015 (Fig. 2.9, Table 2.8) with an increase in abundances of 72% for *T. fasciata* and 171% for *A. rostrata* in NTZs compared to FZs that year. There appeared to be a trend towards increasing abundances of *T. fasciata* over the three sampling years in NTZs but not in FZs (Fig. 2.9a). Abundances of stingarees (Urolophidae spp.) were also greater in 2015 compared to earlier years (Fig. 2.9b) and the increase was greatest in NTZs, however the increase was not statistically significant (Table 2.8; Zone: $p = 0.07$).

Abundances of adult *P. caeruleopunctatus*, ocean jacket (*Nelusetta ayraudi*), eastern school whiting (*Sillago flindersi*) and Port Jackson shark (*Heterodontus portusjacksoni*) did not differ between management zones in shallow water (Table 2.8, Fig. 2.9). Adult *P. caeruleopunctatus* were much more patchily encountered in FZs compared to NTZs (Fig. 2.9a); recorded on 28% of BRUV deployments in FZs and 47% of deployments in NTZs (Table 2.2) and average MaxN per deployment was lower in FZs (Table 2.4), the differences in abundances among zones were not statistically significant (Table 2.8). Abundances of *S. flindersi* increased substantially through time in shallow water in both NTZs and FZs (Fig. 2.9a); from a total of 16 (0.4 fish per deployment) counted in 2011 to 250 counted in 2015 (5.2 fish per deployment). In shallow water, *N. ayraudi* had the highest total abundance of any species accounting for 27% of the total count (Table 2.4). Although they appeared on slightly fewer than 25% of our samples (Table 2.2) and therefore under our frequency of occurrence threshold, we present the results here and included them for analysis due to their striking patterns in abundance. In contrast to *Platycephalus* spp. which had a similar Total MaxN and appeared on almost all BRUV deployments, *N. ayraudi* were patchily distributed and were only seen on 29 deployments. Almost all of these were in 2013 when huge numbers turned up on some BRUV deployments; particularly at the two FZ locations closest to the entrance of Jervis Bay; Booderee and Long beach (Fig. 2.9b).

In deep water, no zoning effects were detected for any of the individual taxa. *Platycephalus* spp. abundances were similar at all locations (Fig. 2.10a) and there was no difference by management zone (Table 2.9). Compared to shallow water samples, adult *P. caeruleopunctatus* appeared much more frequently in deeper water where they were recorded at every site in both years (Fig. 2.10a) and appeared on all NTZ deployments and 73% of those in FZs. Abundances of *T. fasciata* increased through time in both NTZs and FZs and for *A. rostrata* the opposite occurred with a decrease in numbers recorded in 2015 compared to 2011 (Fig. 2.10a). In deep water, very few *S. flindersi* were recorded in 2011 (1.1 fish per deployment) and abundances increase substantially in 2015 (12 fish per deployment). This increase occurred in both NTZs and FZs (Fig 2.10b). We also detected no differences across zones for *H. portusjacksoni,* however unlike *S. flindersi* they were present in low numbers in both years (Fig. 2.10b).

Table 2.7: Results of comparisons across zones (fished and no-take) for a) total abundance (TMaxN), b) species richness (SR), c) Shannon diversity and d) Pielou's evenness measure using permutational analysis of variance (PERMANOVA). 10 m depth factors: Year (Ye, random, 3 levels: 2011, 2013 and 2015), Zones (Zo, fixed, 2 levels: fished and no-take), locations (Lo, random,) and sites (Si). 20 m depth factors are the same except Year (Ye, random, 2 levels: 2011 and 2015). Significant *P* values (<0.05) shown in bold.

Source	df	SS	MS	Pseudo-F	P(perm)	Source	df	SS	MS	Pseudo-F	P(perm)
a) MaxN 10 m						**a) MaxN 20 m**					
Ye	2	423.60	211.80	1.00	0.428	Ye	1	2541.20	2541.20	6.53	0.109
Zo	1	140.65	140.65	1.42	0.334	Zo	1	11.29	11.29	2.98	0.225
Lo(Zo)	4	527.85	131.96	0.51	0.504	Lo(Zo)	2	277.04	138.52	0.85	0.578
YexZo	2	224.19	112.10	0.53	0.611	YexZo	1	1.25	1.25	0.00	0.866
Si(Lo(Zo))	6	722.95	120.49	3.43	**0.004**	Si(Lo(Zo))	4	178.65	44.66	0.19	0.928
YexLo(Zo)**	7	1458.10	208.30	5.92	**0.0002**	YexLo(Zo)	2	810.04	405.02	1.68	0.293
Pooled	103	3622.70	35.17			YexSi(Lo(Zo))	4	952.88	238.22	1.70	0.167
Total	125	6810.30				Res	47	6573.70	139.87		
						Total	62	12313.00			
b) Species Richness 10 m						**b) Species Richness 20 m**					
Ye	2	24.81	12.41	4.16	0.065	Ye	1	39.71	39.71	5.42	0.126
Zo	1	0.42	0.42	0.18	0.971	Zo	1	6.35	6.35	0.32	0.739
Lo(Zo)	4	38.33	9.58	2.58	0.059	Lo(Zo)	2	1.17	0.58	1.00	0.503
YexZo	2	18.67	9.33	3.13	0.100	YexZo	1	41.49	41.49	5.66	0.124
Si(Lo(Zo))	6	14.53	2.42	0.59	0.733	Si(Lo(Zo))	4	10.19	2.55	0.27	0.884
YexLo(Zo)**	7	20.50	2.93	0.72	0.669	YexLo(Zo)	2	15.17	7.58	0.80	0.509
YexSi(Lo(Zo))	10	41.22	4.12	1.67	0.102	YexSi(Lo(Zo))	4	37.69	9.42	2.96	**0.027**
Res	93	229.92	2.47			Res	47	149.67	3.18		
Total	125	397.97				Total	62	279.43			
c) Shannon Diversity 10 m						**c) Shannon Diversity 20 m**					
Ye	2	4.21	2.11	1.27	0.335	Ye	1	6.11	6.11	1.05	0.384
Zo	1	0.51	0.51	0.94	0.510	Zo	1	0.56	0.56	0.39	0.749
Lo(Zo)	4	8.71	2.18	1.57	0.213	Lo(Zo)	2	0.41	0.21	0.13	0.830
YexZo	2	0.21	0.11	0.09	0.931	YexZo	1	15.87	15.87	2.72	0.225
Si(Lo(Zo))	6	5.66	0.94	0.50	0.803	Si(Lo(Zo))	4	7.85	1.96	2.28	0.072
YexLo(Zo)**	7	11.49	1.64	0.86	0.570	YexLo(Zo)	2	12.24	6.12	7.13	**0.002**
YexSi(Lo(Zo))	10	19.26	1.93	1.94	**0.049**	Pooled	51	43.80	0.86		
Res	93	92.27	0.99			Total	62	80.83			
Total	125	145.34									
d) Pielou's Evenness 10 m						**d) Pielou's Evenness 20 m**					
Ye	2	0.04	0.02	0.86	0.463	Ye	1	3743.40	3743.40	5.44	0.096
Zo	1	0.05	0.05	1.95	0.209	Zo	1	47.93	47.93	1.62	0.341
Lo(Zo)	4	0.11	0.03	1.12	0.402	Lo(Zo)	2	714.94	357.47	0.81	0.620
YexZo	2	0.01	0.00	0.23	0.812	YexZo	1	112.85	112.85	0.16	0.844
Si(Lo(Zo))	6	0.08	0.01	1.27	0.348	Si(Lo(Zo))	4	391.43	97.86	0.32	0.922
YexLo(Zo)**	7	0.15	0.02	2.09	0.137	YexLo(Zo)	2	1441.30	720.64	2.36	0.176
YexSi(Lo(Zo))	10	0.10	0.01	1.32	0.235	YexSi(Lo(Zo))	4	1204.80	301.20	1.69	0.136
Res	93	0.73	0.01			Res	47	8374.50	178.00		
Total	125	1.25				Total	62	17783.00			

** Term has one or more empty cells. Pooled indicates where P was > 0.25 and post-hoc pooling was done to increase the power of the main tests (Underwood, 1997).
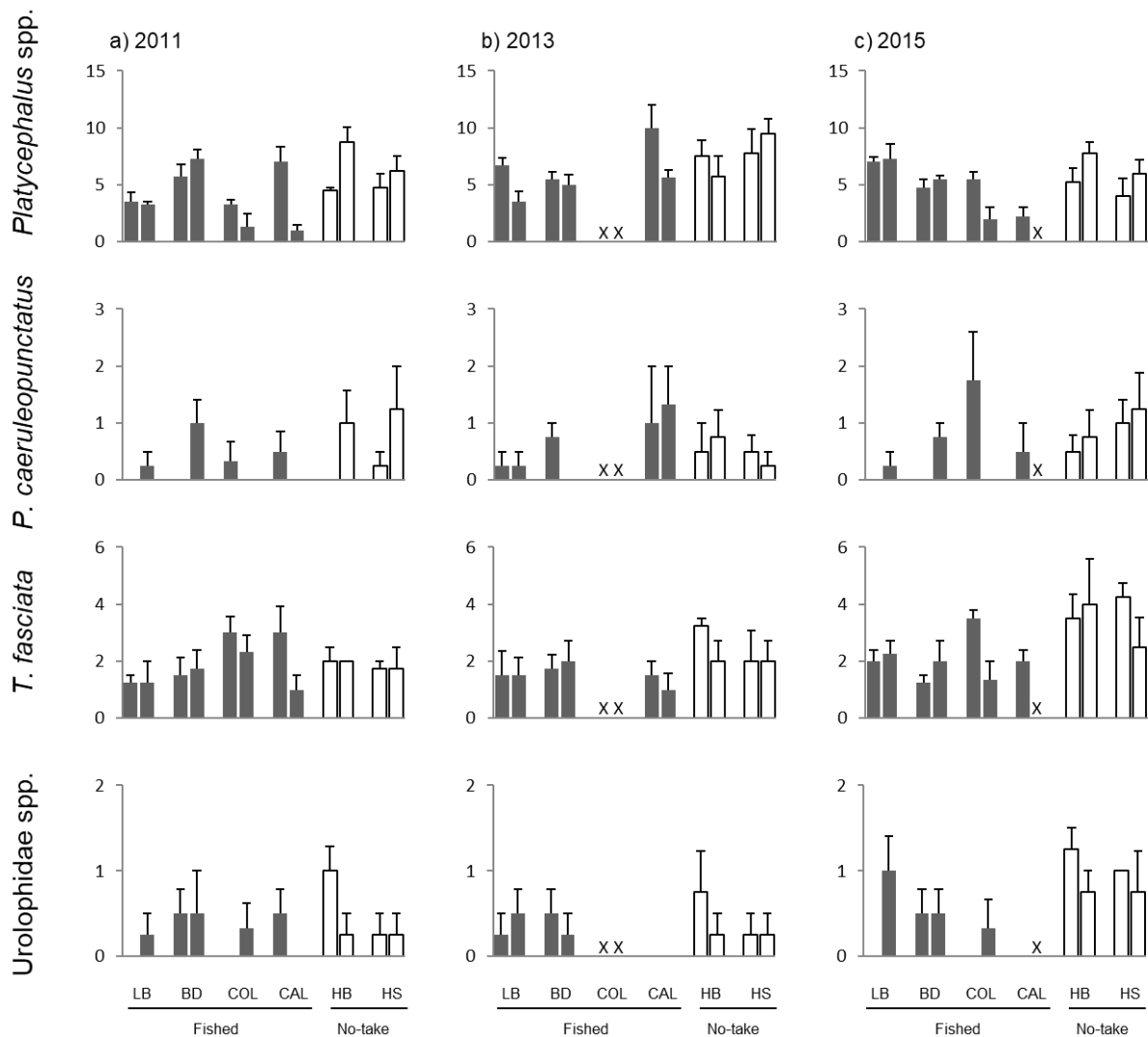
Figure 2.9a: Average abundance (MaxN) at 10 metres depth for each species in a) 2011, b) 2013 and c) 2015 at each site (n = 4 replicates per site) in Jervis Bay Marine Park. Error bars are SE. No-take sanctuary zones are in white, recreationally fished habitat protection zones in grey. X indicates no data. LB = Long Beach, BD = Booderee, COL = Collingwood Beach, CAL = Callala Beach, HB = Hare Bay, HS = Hyams Beach, and each of these locations has two sites. Platycephalus includes *P. grandispinis* and *P. caeruleopunctatus*. Only adults are included in *P. caeruleopunctatus.*
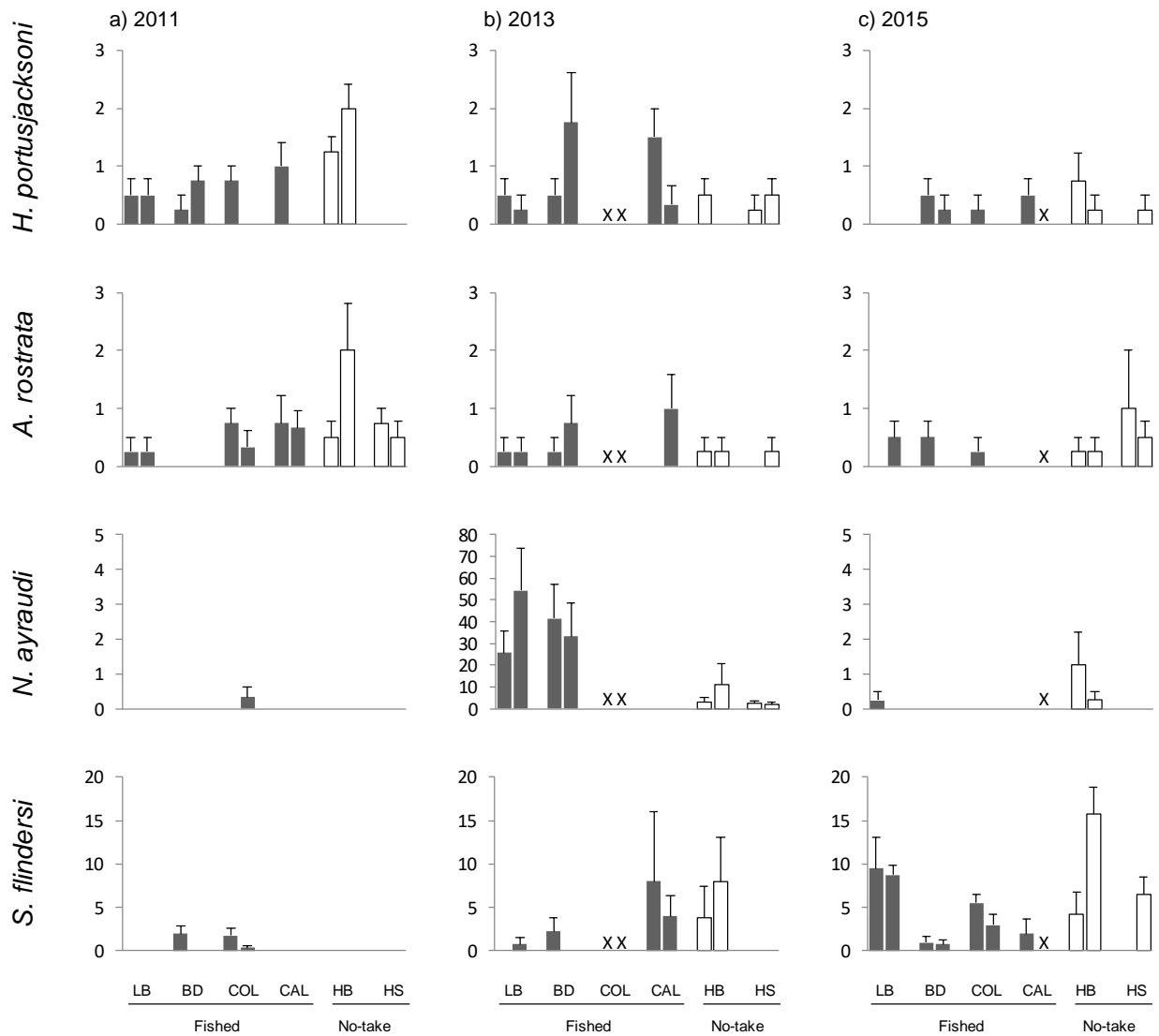
Figure 2.9b: Average abundance (MaxN) at 10 metres depth for each species in a) 2011, b) 2013 and c) 2015 at each site (n = 4 replicates per site) in Jervis Bay Marine Park. Error bars are SE. No-take sanctuary zones are in white, recreationally fished habitat protection zones in grey. X indicates no data. LB = Long Beach, BD = Booderee, COL = Collingwood Beach, CAL = Callala Beach, HB = Hare Bay, HS = Hyams Beach, and each location has two sites. Note the y axis scale for *N. ayraudi* in 2013.

Table 2.8: Results of abundance (MaxN) comparisons at 10 metres depth between fished and unfished zones for individual species using univariate PERMANOVA. Factors: Year (Ye, random, 3 levels: 2011, 2013 and 2015), Zones (Zo, fixed, 2 levels: fished and no-take), locations (Lo, random,) and sites (Si). Significant P values (<0.05) shown in bold.

Source	df	SS	MS	Pseudo-F	P(perm)	Source	df	SS	MS	Pseudo-F	P(perm)
a) *Platycephalus* spp.						b) *P. caeruleopunctatus*					
Ye	2	69.91	34.95	1.18	0.362	Ye	2	1.40	0.70	0.64	0.558
Zo	1	102.52	102.52	9.49	**0.010**	Zo	1	2.10	2.10	1.09	0.433
Lo(Zo)	4	51.38	12.85	0.36	0.956	Lo(Zo)	4	7.81	1.95	1.51	0.233
YexZo	2	0.19	0.10	0.01	0.999	YexZo	2	1.78	0.89	0.81	0.475
Si(Lo(Zo))	6	144.65	24.11	3.99	**0.031**	Si(Lo(Zo))	6	4.86	0.81	0.91	0.528
YexLo(Zo)**	7	202.28	28.90	4.77	**0.012**	YexLo(Zo)**	7	7.64	1.09	1.21	0.373
YexSi(Lo(Zo)	10	60.86	6.09	1.30	0.233	YexSi(Lo(Zo)	10	9.04	0.90	1.58	**0.118**
Res	93	434.58	4.67			Res	93	53.17	0.57		
Total	125	973.50				Total	125	87.43			
c) *T. fasciata*						d) Urolophidae					
Ye	2	17.10	8.55	5.05	**0.008**	Ye	2	2.24	1.12	4.05	0.019
Zo	1	14.18	14.18	1.72	0.122	Zo	1	2.94	2.94	2.46	0.073
Lo(Zo)	4	9.50	2.37	0.91	0.534	Lo(Zo)	4	2.22	0.56	0.83	0.537
YexZo	2	13.61	6.80	4.02	**0.021**	YexZo	2	1.45	0.72	2.62	0.078
Si(Lo(Zo))	6	16.01	2.67	1.58	0.160	Si(Lo(Zo))	6	4.11	0.69	2.48	**0.026**
Pooled	110	186.36	1.69			Pooled	110	30.42	0.28		
Total	125	255.43				Total	125	42.36			
e) *H. portusjacksoni*						f) *A. rostrata*					
Ye	2	3.84	1.92	1.67	0.255	Ye	2	2.62	1.31	2.69	0.067
Zo	1	0.01	0.01	0.36	0.870	Zo	1	1.03	1.03	0.79	0.270
Lo(Zo)	4	6.41	1.60	1.00	0.493	Lo(Zo)	4	0.88	0.22	0.45	0.787
YexZo	2	3.39	1.69	1.48	0.296	YexZo	2	3.50	1.75	3.58	**0.030**
Si(Lo(Zo))	6	5.92	0.99	2.10	0.148	Pooled	116	56.68	0.49		
YexLo(Zo)**	7	7.87	1.12	2.39	0.104	Total	125	64.36			
YexSi(Lo(Zo)	10	4.73	0.47	1.37	0.205	Pairwise Test				t	P(perm)
Res	93	32.17	0.35			Fished = No-take (2011)				1.794	0.169
Total	125	63.43				Fished = No-take (2013)				1.897	0.152*
						Fished ≠ No-take (2015)				1.895	**0.014**
g) *S. flindersi*						h) *N. ayraudi*					
Ye	2	486	243.23	3.60	0.082	Ye	2	5257	2628.40	4.94	0.054
Zo	1	21	20.76	0.89	0.527	Zo	1	1363	1363.30	1.05	0.454
Lo(Zo)	4	274	68.54	0.86	0.593	Lo(Zo)	4	1841	460.14	0.97	0.196
YexZo	2	57	28.60	0.43	0.684	YexZo	2	2615	1307.50	2.46	0.159
Si(Lo(Zo))	6	231	38.58	1.85	0.172	Si(Lo(Zo))	6	610	101.72	0.74	0.626
YexLo(Zo)**	7	462	65.96	3.15	**0.050**	YexLo(Zo)**	7	3635	519.27	3.77	**0.002**
YexSi(Lo(Zo)	10	211	21.09	1.70	0.089	Pooled	103	14200	137.87		
Res	93	1152	12.38			Total	125	35343			
Total	125	2908									

* Monte Carlo *P* Value. ** Term has one or more empty cells. Pooled indicates where *P* was > 0.25 and post-hoc pooling was done to increase the power of the main tests (Underwood, 1997). # 4th root transformed. PERMDISP was non-significant in all cases with the exception of *Trygonorrhina fasciata* ($p = 0.034$), until square root transformed ($p = 0.62$).
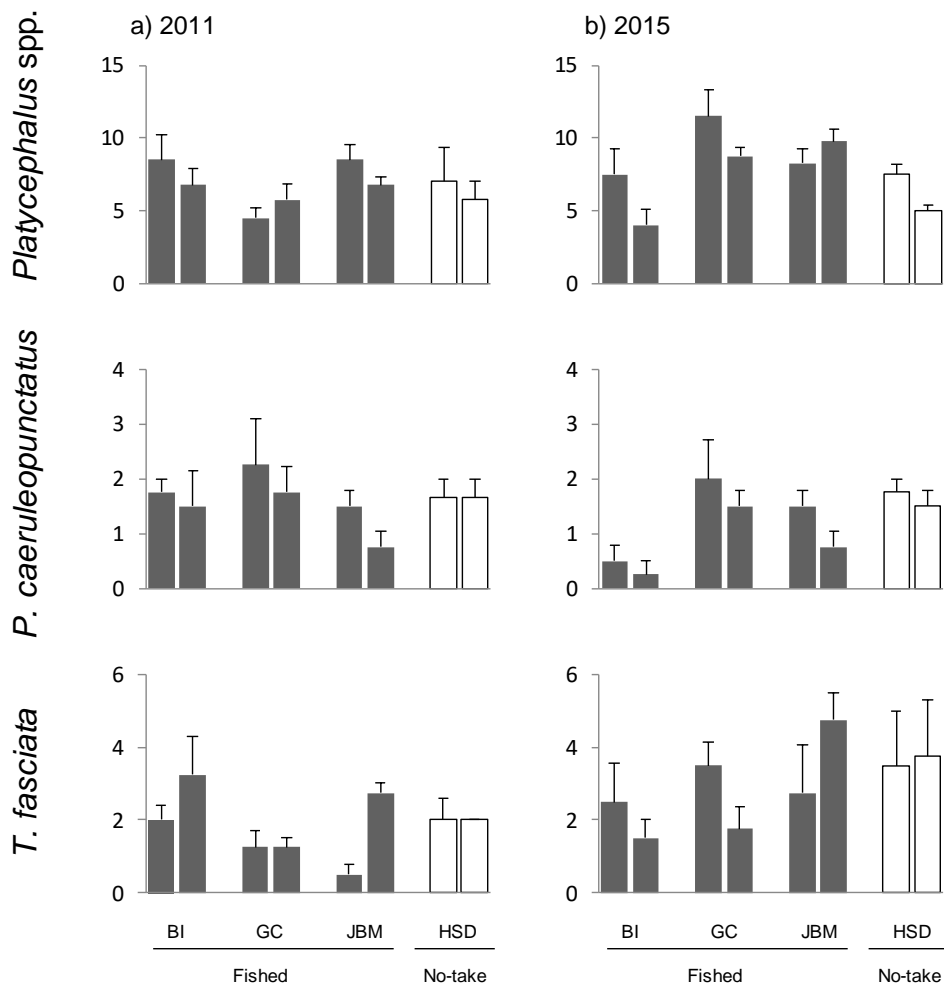
Figure 2.10a: Average abundance (MaxN) at 20 metres depth for each species in a) 2011 and b) 2015 at each site (n = 4 replicates per site). Error bars are SE. No-take sanctuary zones are in white, recreationally fished habitat protection zones in grey. BI = Bowen Island, GC = Grouper Coast, JBM = Jervis Bay Mid, HSD = Hyams Sanctuary Deep. Each of these locations has two sites. *Platycephalus* includes *P. grandispinis* and *P. caeruleopunctatus*. Only adults are included in *P. caeruleopunctatus*.
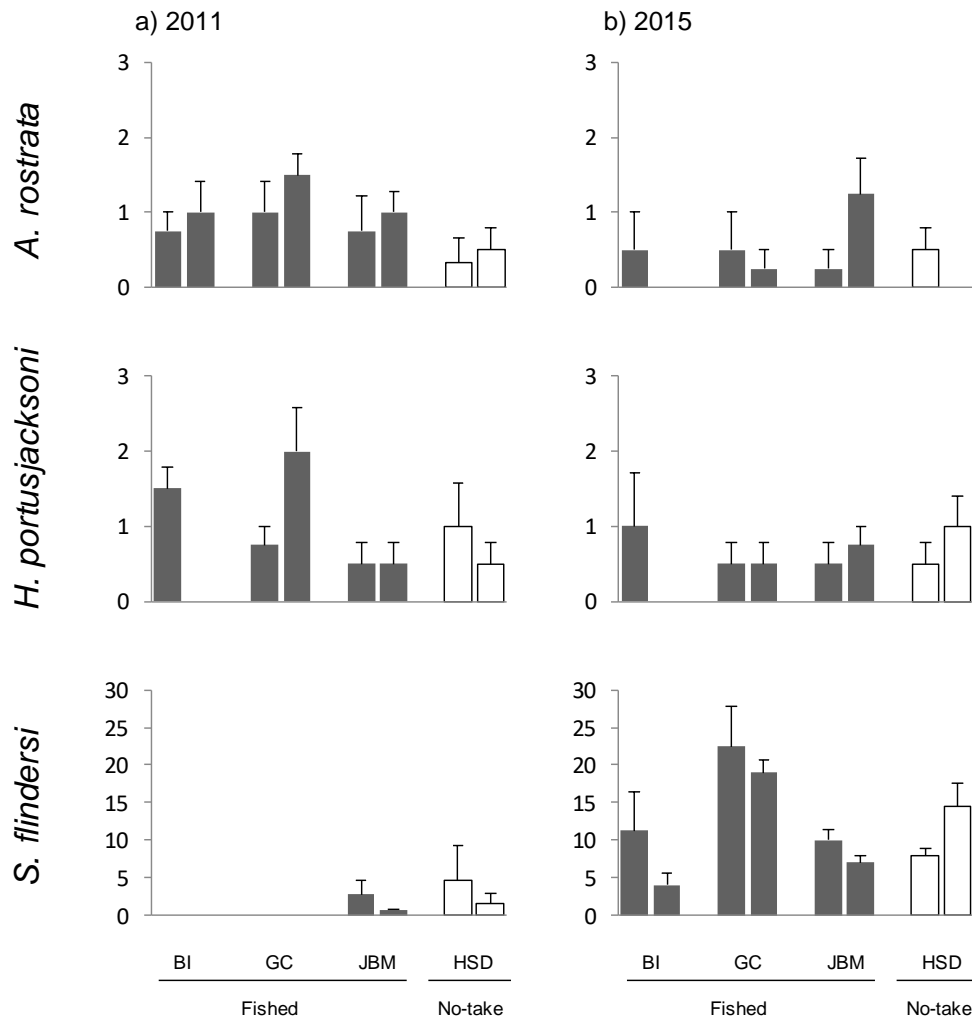
Figure 2.10b: Average abundance (MaxN) at 20 metres depth for each species in a) 2011 and b) 2015 at each site (n = 4 replicates per site). Error bars are SE. No-take sanctuary zones are in white, recreationally fished habitat protection zones in grey. BI = Bowen Island, GC = Grouper Coast, JBM = Jervis Bay Mid, HSD = Hyams Sanctuary Deep. Each of these locations has two sites.

Table 2.9: Results of abundance (MaxN) comparisons at 20 metres depth between fished and unfished zones for individual species using univariate PERMANOVA. Factors: Year (Ye, random, 2 levels: 2011and 2015), Zones (Zo, fixed, 2 levels: fished and no-take), locations (Lo, random,) and sites (Si).

Source	df	SS	MS	Pseudo-F	P(perm)	Source	df	SS	MS	Pseudo-F	P(perm)
a) *Platycephalus* spp.						b) *P. caeruleopunctatus*					
Ye	1	4.69	4.69	0.1	0.786	Ye	1	0.96	0.96	0.83	0.467
Zo	1	15.91	15.91	3.38	0.206	Zo	1	1.25	1.25	0.44	0.725
Lo(Zo)	2	23.04	11.52	0.29	0.636	Lo(Zo)	2	8.79	4.4	3.36	0.093
YexZo	1	7.55	7.55	0.16	0.74	YexZo	1	1.25	1.25	1.08	0.399
Si(Lo(Zo))	4	41.05	10.26	1.78	0.148	Si(Lo(Zo))	4	1.97	0.49	0.39	0.814
YexLo(Zo)	2	98.04	49.02	8.53	**0.001**	YexLo(Zo)	2	2.38	1.19	0.93	0.472
Pooled	51	293.25	5.75			YexSi(Lo(Zo))	4	5.07	1.27	1.74	0.163
Total	62	498.89				Res	47	34.17	0.73		
						Total	62	58.32			
c) *T. fasciata*						d) *H. portusjacksoni*					
Ye	1	19.07	19.07	2.45	0.247	Ye	1	0.25	0.25	0.25	0.68
Zo	1	2.78	2.78	3.04	0.207	Zo	1	0.01	0.01	0.97	0.519
Lo(Zo)	2	4.5	2.25	0.38	0.547	Lo(Zo)	2	1.29	0.65	0.39	0.547
YexZo	1	1.3	1.3	0.17	0.754	YexZo	1	0.39	0.39	0.39	0.605
Si(Lo(Zo))	4	21.26	5.31	1.92	0.124	Si(Lo(Zo))	4	7.88	1.97	3.82	**0.008**
YexLo(Zo)	2	16.17	8.08	2.92	**0.066**	YexLo(Zo)	2	2.04	1.02	1.98	0.143
Pooled	51	141.25	2.77			Pooled	51	26.3	0.52		
Total	62	207.56				Total	62	38.86			
e) *A. rostrata*						f) *S. flindersi*					
Ye	1	1.48	1.48	2.95	0.092	Ye	1	1129.4	1129.4	5	0.095
Zo	1	1.73	1.73	3.19	0.127	Zo	1	7.08	7.08	1.04	0.488
Lo(Zo)	2	0.67	0.33	0.67	0.518	Lo(Zo)	2	394.13	197.06	0.88	0.546
YexZo	1	0.38	0.38	0.75	0.395	YexZo	1	37.96	37.96	0.17	0.74
Pooled	57	28.51	0.5			Si(Lo(Zo))	4	108.13	27.03	0.72	0.616
Total	62	34.6				YexLo(Zo)	2	477.54	238.77	6.21	0.068
						YexSi(Lo(Zo))	4	151.13	37.78	1.67	0.166
						Res	47	1065.2	22.66		
						Total	62	4155.1			

Lengths

I detected no differences in length of either *Platycephalus* spp. or adult *P. caeruleopunctatus* when comparing by zone. In shallow water, a total of 543 *Platycephalus* spp. were measured from the 126 BRUV deployments; 258 in NTZs and 285 in FZs. The shape of the cumulative length frequency distribution obtained for *Platycephalus* spp. at the genus level were similar in both FZs and NTZs (Fig. 2.11) and did not differ significantly between the zones (Table 2.10). The length frequency distribution was unimodal with most of the fish around the 140 to 180 mm length (Fig. 2.11). In deep water, a total of 227 *Platycephalus* spp. were measured from 63 BRUV deployments; 193 in FZs and 34 in NTZs. The shape of the cumulative length frequency distribution obtained for *Platycephalus* spp. at the genus level were similar in both FZs and NTZs (Fig. 2.12) and did not differ significantly between the zones (Table 2.10). The length frequency distribution showed that most fish measured in deep water were around the 160 to 200 mm length (Fig. 2.12). Few larger fish were measured in these estimates of size for *Platycephalus* spp. at either depth as MaxN (where measurements were taken) generally was dominated by juveniles.

Measurements of adult *P. caeruleopunctatus* at both depths appeared to better account for larger fish that were often missed in measurements at the genus level but comparisons were limited by the lower sample sizes obtained, particularly in deep water (Table 2.10). Any comparisons between zones using the results for adult *P. caeruleopunctatus* should therefore be treated with some caution. In shallow water, 48 adult *P. caeruleopunctatus* were measured (from 80 deployments in 2013 and 2015) and the shape of the cumulative length frequency distributions (Fig. 2.13) did not differ significantly between zones, (Table 2.10) although the average length in NTZs was larger at 318 mm compared to 281mm in FZs (Table 2.10). In deep water, 26 adult *P. caeruleopunctatus* were measured (from 32 deployments in 2015) the trend was the opposite with average length in FZs larger at 338 mm compared to 333 mm though again differences were not significant (Fig. 2.14, Table 2.10).

Table 2.10: Average length (+- SE), minimum and maximum length by zone type. Results of Kolmogorov-Smirnov (KS) tests of differences of cumulative length distributions between fished and no-take zones for flathead spp. (*Platycephalus grandispinis* and *P. caeruleopunctatus*) and adult blue-spotted flathead (BSF, *P. caeruleopunctatus*).

	Flathead		Adult BSF	
	Fished	No-take	Fished	No-take
a) 10 m				
Max (mm)	507.7	630.2	507.7	630.2
Min (mm)	55.3	59.4	174.5	181.8
Ave (mm, SE)	170.4 (2.8)	170.6 (2.9)	281.0 (17.6)	317.8 (22.5)
N	285	258	25	23
KS test	D	P	D	P
	0.045	0.95	0.226	0.49
b) 20 m				
Max (mm)	391.3	360.8	545.1	399.6
Min (mm)	85.6	121	296.1	227.9
Ave (mm, SE)	192.9 (3.9)	190.4 (6.8)	338 (17.1)	333 (16)
N	193	34	14	12
KS test	D	P	D	P
	0.131	0.7	0.274	0.62
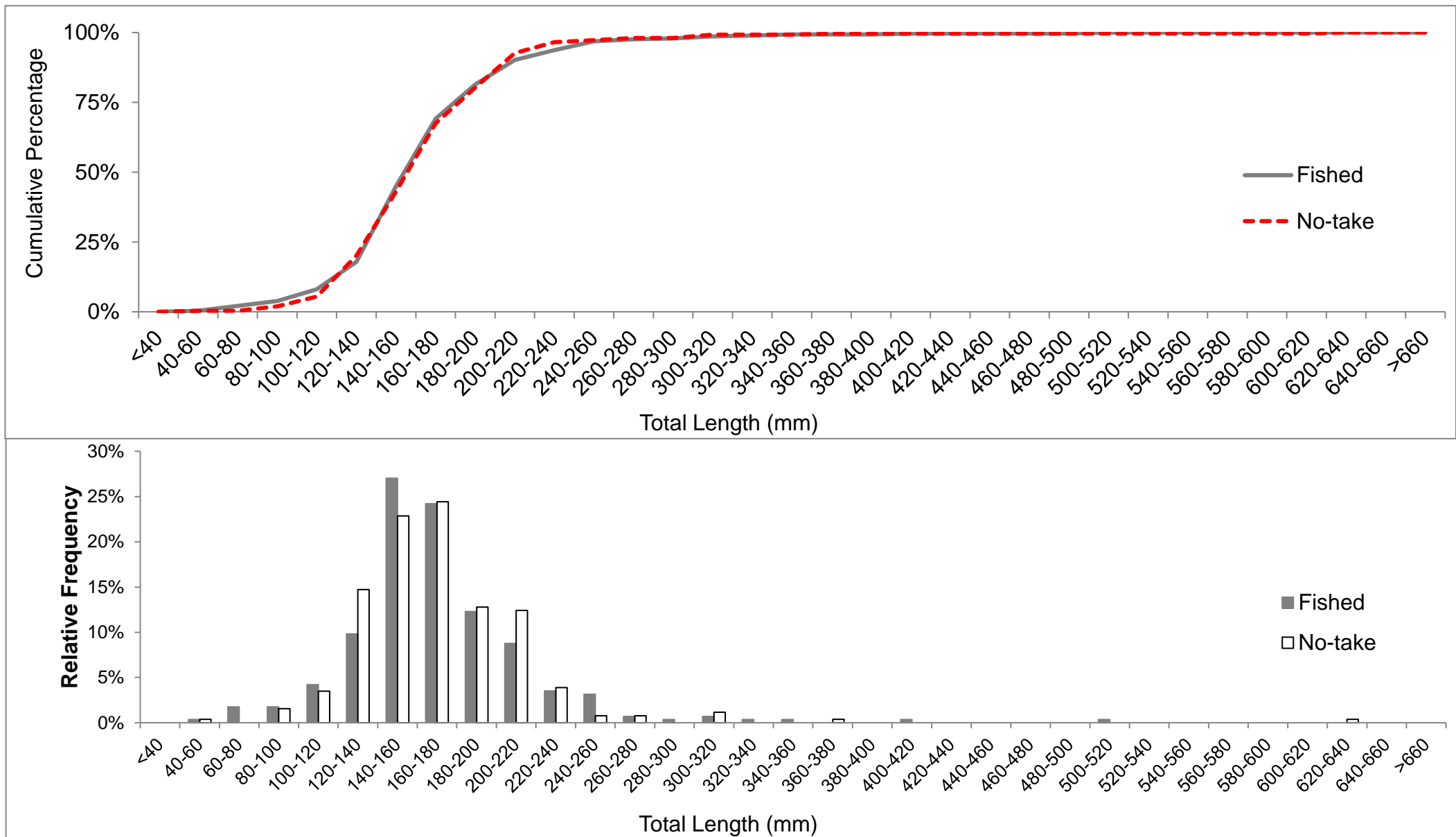
Figure 2.11: 10 metre depth flathead cumulative length frequency distributions (Top) and length frequency distributions (Bottom) in 20 mm length intervals, by no-take zones (N = 258) and fished zone (N = 285) for 2011–2015. Plotted as cumulative percentages as sample sizes are uneven.
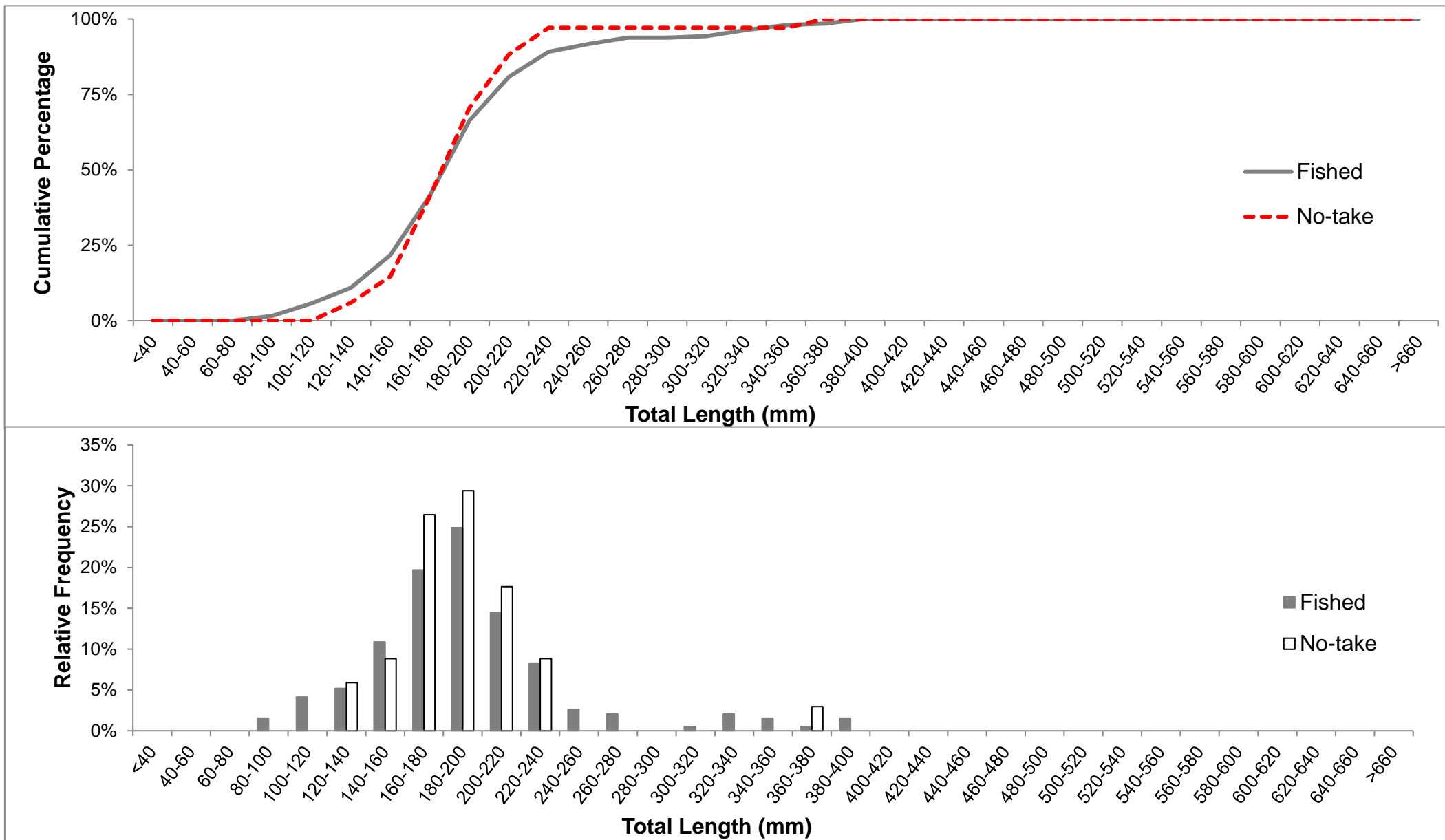
Figure 2.12: 20 metre depth flathead cumulative length frequency distributions (Top) and length frequency distributions (Bottom) in 20 mm length intervals, by no-take zones (N = 34) and fished zone (N = 193) for 2011 and 2015.
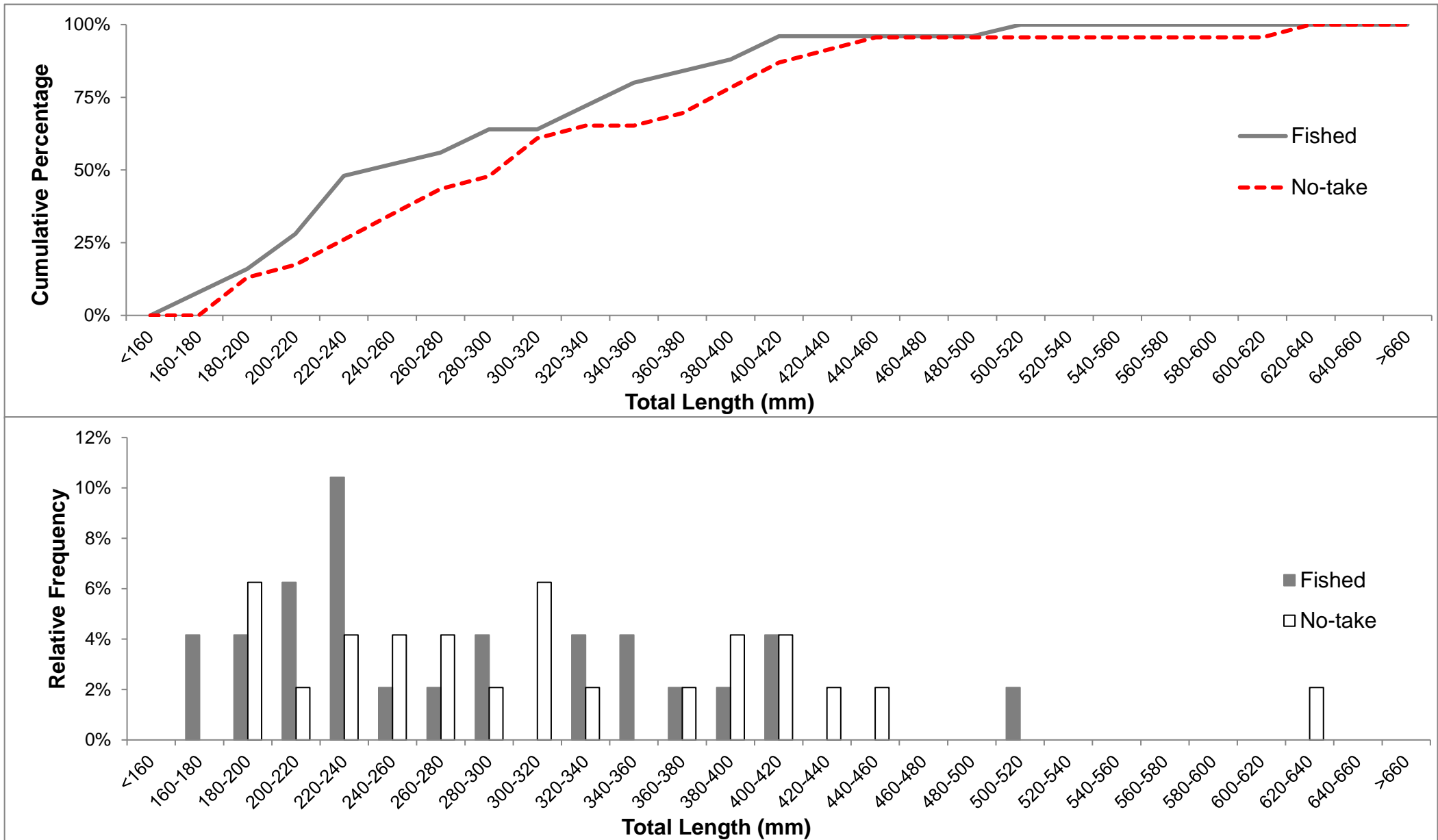
Figure 2.13: 10 metre depth adult blue-spotted flathead; cumulative length frequency distributions (Top) and length frequency distributions (Bottom), in 20 mm length intervals, by no-take zones (N = 23) and fished zone (N = 25) for 2013 and 2015.
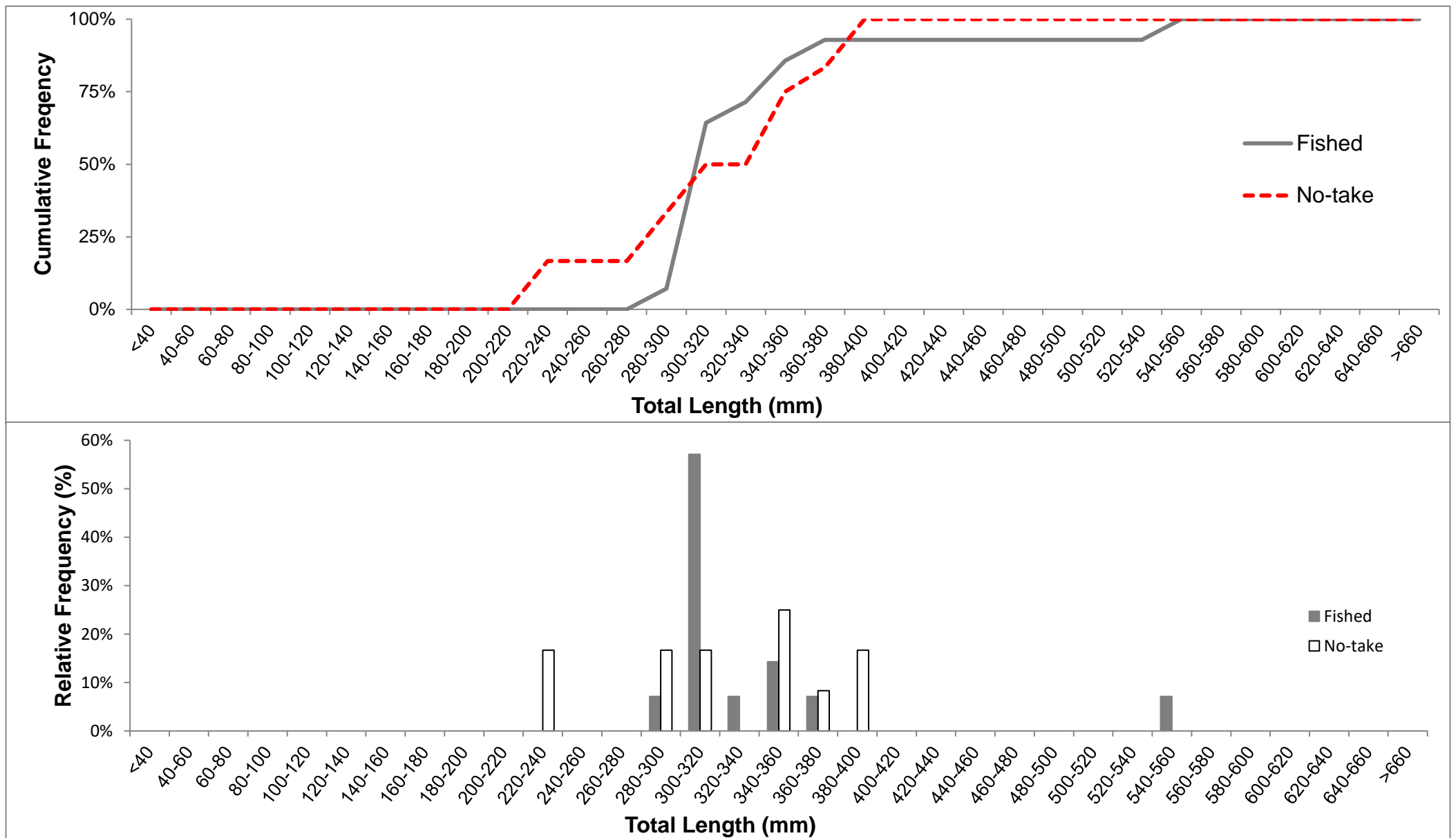
Figure 2.14: 20 metre depth adult blue-spotted flathead; cumulative length frequency distributions (Top) and length frequency distributions (Bottom) in 20 mm length intervals, by no-take zones (N = 12) and fished zone (N = 14) for 2015 only.

2.4 Discussion

Though rarely investigated, assessing the response of marine soft sediment fishes to the removal of fishing is critical in determining the efficacy of marine protected areas (MPAs), particularly given that many MPAs are dominated by soft sediment habitats. I have provided one of the first in-depth assessments within a temperate MPA of demersal soft sediment fish assemblage composition, species richness, relative abundance and lengths across no-take zones (NTZs) and fished zones (FZs) over time. I found no evidence of differences in demersal soft sediment fish assemblage composition or diversity between zones or differences in total relative abundance (all species combined), species richness or size in either shallow (10 m ± 2 m) or deep (20 m ± 2 m) water within Jervis Bay. There were no differences in abundances for any individual species at 20 m or in sizes of flathead at either depth. In shallow water, however, flathead (*Platycephalus* spp*.)*, eastern fiddler ray (*Trygonorrhina fasciata*) and shovelnose ray (*Aptychotrema rostrata*) had higher abundances in NTZs compared to FZs.

Contrary to my predictions that NTZs would support a different assemblage to FZs, there was no difference at the assemblage level between zones, despite the expectation of higher recreational fishing effort in the fished zones. Many of the species in the assemblage are rarely retained by recreational fishers (e.g. Port Jackson sharks - *Heterodontus portusjacksoni*), or are never caught (eastern fortescue - *Centropogon australis*, weeping toadfish - *Torquigener pleurogramma*) so are unlikely to show a direct response differences across management zones and contribute to differences across zones. Nevertheless, I still would have expected sufficient differences in targeted species across zones to result in an assemblage difference being detected. There were considerable differences among sites in shallow water and locations in deeper water, which has been found numerous times in other studies (e.g. Connell and Lincoln-Smith 1999, Hyndes et al. 1999, Sih et al. 2017) and is not unexpected here, particularly as many of the species in the demersal soft sediment assemblage in Jervis Bay are known to appear sporadically across years in large numbers (e.g. ocean jackets - *Nelusetta ayraudi* and eastern school whiting - *Sillago flindersi*). However, these differences across years occurred in both management zones, and as I was most interested in testing effects across zones the differences were not of great relevance. It should be noted that unlike all the

other common species recorded, *S. flindersi* are incredibly hard to detect on video and it is possibly that for this species, poorer visibility in 2011 had some part to play in lower numbers at both depths.

In shallow water, the strongest and most consistent result was greater relative abundance of *Platycephalus* spp. (*P. caeruleopunctatus* and *P. grandispinis*) in NTZs compared to FZs irrespective of sampling year. On average, there were 32% more *Platycephalus* spp. recorded in NTZs compared to FZs. Targeted species often show the earliest response to the cessation of fishing (Babcock et al. 2010), particularly species that are site attached . *Platycephalus* spp. are highly sought after by recreational fishers, and in addition, my research (as reported in this thesis) suggests that a large proportion of the populations of both *P. caeruleopunctatus* and *P. grandispinis* can exhibit long term site attachment (See also Fetterplace et al. 2016). Higher abundances of *Platycephalus* spp. in our study accord with our predictions and are consistent with findings for site attached targeted species on hard substratum habitats in the region. For example; luderick (*Girella tricuspidata*) on shallow subtidal reefs in Jervis Bay were 86% more abundant in NTZs compared to FZs (Ferguson et al. 2016) and similarly, red morwong (*Cheilodactylus fuscus*) were found to be more abundant in NTZs in a number of studies on rocky reef (Edgar and Barrett 2012, Coleman et al. 2013, Malcolm et al. 2016).

My research has also revealed increases in abundance through time in NTZs. In shallow water, two species (*T. fasciata* and *A. rostrata)*, had significantly higher abundances in NTZs in 2015 but not in the earlier two sampling years. Members of the Urolophidae showed a similar though not significant pattern in abundances. All three are commonly caught by recreational fishers, with *A. rostrata* often retained, *T. fasciata* mostly released and Urolophids almost never kept (Authors pers. obs.). Greater abundances in 2015 (13 years 8 months after JBMP zoning was implemented) may reflect a lag in the influence of NTZs for these species (Roberts et al. 2001, Molloy et al. 2009, Babcock et al. 2010, Edgar and Barrett 2012), alternately it may simply relate to temporal variation in abundance. Cyclical peaks and falls in abundances of marine fish are common and there is some evidence suggesting that peaks can be stronger in NTZs. For example, in a 13 year study on shallow reefs, snapper abundances were higher in NTZs in peak years (Malcolm et al. 2015). Numbers stayed higher in the year or two following a strong

peak, before returning to similar abundances to fished areas before the next peak (Malcolm et al. 2015). There are no studies I am aware of comparing demersal marine soft sediment fish assemblages over such extended an extended time frame in MPAs, so it is unclear whether our results for these species will follow a similar pattern. Further research, likely over decadal timescales, is required to establish whether the greater abundances I observed for these species in NTZs continue consistently post 2015 or follow a cyclical pattern.

In deep water, I found no differences in abundances for any species across zones. In stark contrast to our predictions and our results in shallow water, *Platycephalus* spp. abundances did not differ between zones in deep water in any year. Further, in 2015 when a number of species in shallow water had higher abundances, these were not reflected in my deep-water comparisons. Why there was no difference in abundances between zones for these species at the deeper depth is not clear. A possible explanation is that our sample sites in deeper water are close to the edge of the NTZ whilst our shallow water sites are well within the boundaries of the NTZs (Kramer and Chapman 1999). 'Edge' effects where abundances of fish around the perimeter of a no-take MPA are lower than those in the centre can be related to several factors including; fish close to the reserve boundary moving in and out of the reserve exposing them to a risk of greater fishing mortality (Malcolm et al. 2016), and 'fishing the line' where fishers tend to preferentially fish along the outer edge of NTZs exposing fish there to higher fishing mortality levels (Kellner et al. 2007). Further, anecdotally it appears that illegal fishing i.e. "fishing inside the line" in Jervis Bay tends to be along the inside edges of the zones (Authors pers. obs.).

As fishing tends to target larger individuals, biases towards larger size classes and/or increases in mean size in protected areas have commonly been reported (Lester et al. 2009). In contrast, our results indicate that while there were more *Platycephalus* spp. in NTZs in shallow water, there was no apparent bias towards larger length frequencies compared to FZs in either shallow or deep water. A potential explanation for the pattern in our study is that fish suffer higher mortality in fished areas but do not remain in the NTZs long enough for a size effect to appear. Either because larger fish may move over larger scales exposing them to greater fishing mortality or because at some point larger, older fish leave the NTZ (see Chapter 5). A further consideration is that I failed to measure

the largest *Platycephalus* spp. because of fish behaviour and our method of sampling. MaxN often occurred when smaller juveniles gathered around the bait in large numbers. However, over the course of the study I observed that the largest individual *P. caeruleopunctatus* were often alone or in pairs and juveniles left the BRUV field of view more often when large fish were present (Authors pers. obs). The largest *P. caeruleopunctatus* therefore tended not to appear at *Platycephalus* spp. MaxN and as that is when I attempted to take length measurements they were less likely to have been measured. I attempted to overcome this issue by taking a separate MaxN and set of length measurements for adult *P. caeruleopunctatus* and again found no difference in lengths between zones. However, low numbers of measurements for adult *P. caeruleopunctatus* means that these adult comparisons should be treated with some caution. Again, the largest individuals were often not present at MaxN. Future studies should endeavour to increase the number of length measurements for adult *P. caeruleopunctatus,* possibly through increasing the number of replicate BRUV deployments at each site.

The extent of localised fishing pressure will drive any changes in abundance or fish size and the magnitude of any change when fishing is removed (Halpern 2003, Barrett et al. 2007). If fishing effort is low in 'fished areas' then there will be no effect of fishing. Fishing effort was reported to be relatively homogenous on soft sediments in Jervis Bay before the MPAs zoning was put in place, although some areas in the middle of the bay were never observed to be fished (Lynch 2006). Once the JBMP zoning was in place, effort was reported to have declined substantially (Lynch 2014). If effort has remained low then my results, particularly in deeper water where there were no differences across zones, are likely to be explained by lack of fishing effort. There has, however, been no investigation of fishing effort in JBMP since 2009 and in the absence of current data on fishing pressure, interpretation of my results should be cautious, particularly as there are a number of other potential explanations for the patterns observed.

An alternate explanation for the results is that fishing is having an impact but that movement for most species between the two management zones is sufficient to overcome differences in fishing effort. Or simply put, the fish are moving in and out of the zones frequently enough that they are in effect one assemblage experiencing the same level of fishing mortality. Given that we have no or very limited movement data for most of the

71

fish species in the soft sediment assemblage it is difficult to know whether movement of fish is the driver of the patterns detected (though see Chapter 5).

Soft sediments are the major habitat under protection, are potentially costly to enforce protection on and if not effective may divert compliance resources better spent on other habitats as well as needlessly inconveniencing stakeholders. Conversely, they may be an effective conservation and management measure for fish on soft sediments, yet without ecological monitoring we have no way of determining this either way. Understanding how marine soft sediment fish assemblages respond to protection should be an essential component of the assessment of ecological changes within MPAs. Without this information, it is difficult to know whether conservation goals are being achieved or to quantify the value of protecting these habitats. Developing a clearer understanding of both the movement patterns of fish in the assemblage, and of current fishing effort (both outside and inside the line) remains a significant challenge, quantifying these will allow a clearer interpretation of the effects of (or lack of) protection on marine soft sediments fishes.

2.5 Supporting Information

<u>Diversity Indices</u>

Shannon Diversity used here $=$ $\exp(H)$ and Pielou's evenness $= H/\log(S)$.

The Shannon entrophy H was calculated as

$$H = \sum_{i=1}^{S} p_i \, log_b \, p_i$$

where, S = species richness, b is the base of the logarithm (natural log) and p_i is the proportion of species so that $\sum_{i=1}^{S} P_i = 1$.

Limitations with the most common diversity indices include, 1) the output values are not linear with respect to diversity (Jost 2006, Chao et al. 2010). I overcome this issue by

using the exponential of Shannon entropy [$\exp(H)$] so that the values were linear (Jost 2007, Chao et al. 2010, Gaggiotti et al. 2018) and species diversity values could be compared across management zones intuitively. The use of Hill numbers, such as $\exp(H)$, is argued to be the most effective use of diversity indices to measure diversity (see Ellison 2010, Gaggiotti et al. 2018) and can overcome many of the issues abundance based species diversity indices have (Gaggiotti et al. 2018). However, a potentially greater issue is, 2) these indices do not take into account composition of the assemblages and this greatly limits comparisons among assemblages using these measures. For example; assemblages having no co-occurring species can, counter intuitively, have the same diversity values. See Table S2.1 for a simplified hypothetical demonstration; where three assemblages each have the same number of species and same abundance ratios, however the assemblages have no species in common. The diversity indices do not indicate any differences in the assemblages only that their 'diversity' is equivalent. Nevertheless. these indices may be useful as an additional complimentary measure alongside multivariate comparisons that consider species composition, such as PERMANOVA. For an example, see Chapter 3 where multivariate comparisons suggest the assemblages differ in composition significantly and univariate comparisons of diversity indices suggest that the assemblages found in three management zones have equivalent diversity and evenness. In that instance, the addition of diversity indices is useful as it allows further interpretation of the multivariate results i.e. considered together the results suggest that while each zone maintains equivalent levels of diversity (Shannon diversity), this diversity is represented by different species in each zone (multivariate tests).

Table S2.1: Three hypothetical assemblages (NTZ, PPA, and OA), abundance for each species, and diversity indices values for each assemblage.

Species	NTZ	PPA	OA
Anoplocapros inermis	10	0	0
Aptychotrema rostrata	20	0	0
Carcharias taurus	30	0	0
Chelidonichthys kumu	10	0	0
Chyrosophyrs auratus	10	0	0
Eubalichthys bucephalus	10	0	0
Gorgasia	10	0	0
Heterodontus portusjacksoni	10	0	0
Meuschenia flavolineata	10	0	0
Meuschenia freycineti	10	0	0
Meuschenia scaber	0	10	0
Mustelus antarcticus	0	20	0
Myliobatis australis	0	30	0
Nelusetta ayraudi	0	10	0
Nemadactylus douglasii	0	10	0
Platycephalus caeruleopunctatus	0	10	0
Platycephalus longispinis	0	10	0
Platycephalus richardsoni	0	10	0
Pristiophorus cirratus	0	10	0
Pristiophorus nudipinnis	0	10	0
Pseudocaranx dentex	0	0	10
Pseudorhombus jenynsii	0	0	20
Sardinops sagax	0	0	30
Scomber australasicus	0	0	10
Seriola lalandi	0	0	10
Sillago flindersi	0	0	10
Thyrsites atun	0	0	10
Trachurus novaezelandiae	0	0	10
Trygonorrhina fasciata	0	0	10
Upeneichthys vlamingii	0	0	10
Total Abundance	130	130	130
Species Richness	10	10	10
Shannons Entrophy (H)	2.20	2.20	2.20
Shannons Diversity, exp(H)	9.07	9.07	9.07
Pielou's Evenness Measure	0.96	0.96	0.96
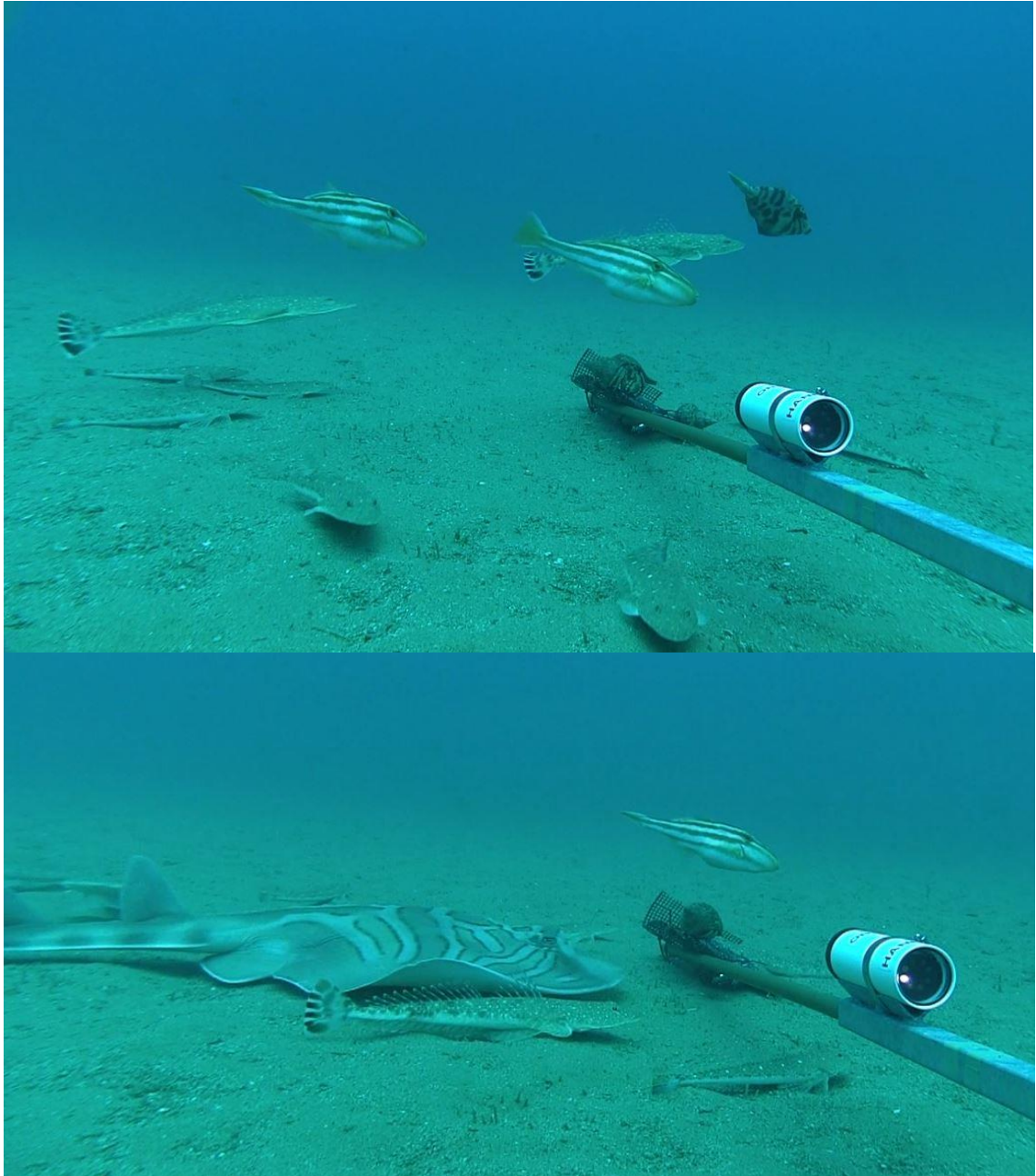
Plate 3.1: Frame grabs taken from a baited remote underwater video system deployed at a depth of 50 metres on soft sediments on the south coast of New South Wales, Australia.

3.1 **Introduction**

Marine soft sediments areas dominate coastal shelf environments (Caveen et al. 2012) and can harbour surprising levels of biodiversity, including many species that only occur on soft sediments. Coastal soft sediment shelf environments are important fishery areas exploited by both commercial and recreational fishing (FAO 2016) and have historically borne the brunt of fishing pressure (Roberts et al. 2003).

The most widespread direct human impact on the seafloor comes from mobile demersal fishing such as trawling and dredging (Halpern et al. 2008, Hughes et al. 2014). The impacts of trawling are well researched and trawling can be highly destructive (Roberts et al. 2003). However, the ecological impacts of trawling and subsequent recovery times after trawling occurs, are highly variable and dependent on numerous factors including habitat type, species life histories and trawling intensity (Hiddink et al. 2017). In contrast to commercial fisheries, there is a poor understanding of the impacts of recreational fisheries (Young et al. 2014), and the combined impact of commercial and recreational fisheries is often underestimated (Cooke and Cox 2004). Although in general, recreational fishing is expected to have a lower impact than commercial fishing (Cooke and Cox 2004), this is dependent on species, gear and fishing effort by each sector. There are for example, demersal species for which landings by recreational fishers are much greater than those by commercial fishers, including red drum (*Sciaenops ocellatus*) and bocaccio (*Sebastes paucispinus)* in some coastal fisheries in the United States (Coleman et al. 2004), and bluespotted flathead (*Platycephalus caeruleopunctatus*) and dusky flathead (*Platycephalus fuscus*) on the east coast of Australia (Stewart et al. 2015).

The direct ecological effects of commercial and recreational fishing on target species are similar (Coleman et al. 2004). Recreational fisheries tend to focus on the top predators and commercial fishing fishes at both upper and lower trophic levels, however in both cases biomass is reduced, size and ages structures are altered and community composition changes (Jennings and Kaiser 1998). All forms of fishing can also have numerous complex indirect impacts, for example removing one species may result in an increase of a prey species or the increase of non-target species in the face of decreased competition (Jennings and Kaiser 1998, Coleman et al. 2004). The net result of both

indirect and direct impacts is that fished and unfished areas would be expected to have different fish communities occupying them. Though this is of course dependent on the degree of movement by fish between the two areas (Gerber et al. 2003, Grüss et al. 2011), the intensity of fishing effort in the fished area (Lester et al. 2009) and age of the MPA at time of sampling (Edgar et al. 2014).

No-take marine protected areas (no-take MPAs) are a form of spatial management where all fishing and other extractive activity is prohibited (Lester et al. 2009). There are few areas of the worlds continental shelves that are unfished (Handley et al. 2014) and therefore no-take MPAs represent some of the few unfished areas of the seafloor. In many cases, on rocky reefs and coral reefs, no-take MPAs have been shown to increase the abundance, size and diversity of fishes compared to areas that continue to be fished (Lester et al. 2009), although these responses are dependent on how well these MPAs are designed, implemented, managed and enforced (Edgar et al. 2014, Gill et al. 2017). Partially protected areas (PPAs) that limit fishing can also increase abundance and size of fish compared to areas that continue to be fully fished, although generally to a lesser degree than in no-take MPAs (Sciberras et al. 2013, Sala and Giakoumi 2017). Recent evidence based on fish biomass data from 218 MPAs (none of which were over soft sediments), shows that on average, fish in PPAs show a positive response to management, although fish biomass in no-take MPAs has a twofold greater response (Gill et al. 2017). However, a number of studies have demonstrated that partially protected areas that allow some fishing can be ineffective (e.g. Malcolm et al. 2015) or may even reduce (McKinley et al. 2011) or nullify (e.g. Denny and Babcock 2004) the overall effect of multiple zone MPAs. Globally, MPAs both no-take and partially protected, cover large areas of coastal soft sediments, yet their impact on demersal fishes on this habitat has rarely been investigated (Caveen et al. 2012).

In the open coastal waters of the South-East coast of Australia, the dominant habitat is soft sediments (Boyd et al. 2004, MEMA 2017). In this region, two multiple zone marine protected areas (MPAs) have been established with the aim of protecting biodiversity and maintaining ecological processes (Marine Parks Act 1997) and both include large areas of soft sediments. Both Jervis Bay Marine Park (JBMP) and Batemans Marine Park (BMP) have a mix of strictly no-take zones, and partially protected zones

which allow recreational fishing and prohibit commercial trawling, long lining, mining and dredging (Read and West 2010). Commercial fishing began on the South-East Coast of Australia in 1915 (Roughley 1951) and the major impacts on the seafloor are from commercial demersal trawling for fishes, prawns and scallops (Evans et al. 2016). Currently, otter trawling and seine fishing are the major commercial types of fishing in use in the region around JBMP and BMP, however line, trap and longlining fisheries are also present. There is also a large active recreational fishing population in the region. Although both MPAs have been in place for a relatively long period (zones came into effect for JBMP in 2002 and BMP in 2007), there has been no research evaluating impacts of MPA zoning on the fish assemblages occupying soft sediment habitats. This is despite soft sediments being the most extensive habitat in both MPAs and the surrounding waters.

In this study, I tested for differences in soft sediment demersal fish assemblages across a gradient of fishing pressure on the southern coast of New South Wales, Australia. The management zones within the two MPAs in the region include both no-take zones and partially protected areas, and given the dominance of soft sediments, provided an excellent opportunity to carry out such an assessment. I hypothesised that the differing levels of fishing pressure would result in (1) differences in fish assemblages among management zones; (2) abundances of targeted and bycatch species being greatest in no-take zones, then partially protected areas and lowest in open access areas outside the parks; (3) Size of targeted species would follow the same pattern, with the largest fish in no-take zones.

3.2 Methods

The study was undertaken on marine soft sediments at depths of 40–60 m on the southern coast of NSW Australia (Fig.3.1). Sampling was undertaken across three levels of management: 1) no-take zones (NTZs) which are strictly no-take marine reserves that prohibit extractive activities; 2) partially protected areas (PPAs), where recreational fishing is permitted and commercial trawling, long lining, mining and dredging are prohibited, 3) open access areas (OAs) where general NSW commercial and recreational fishing rules and regulations apply. All NTZs and PPAs were located within two marine

protected areas (MPAs); Jervis Bay Marine Park (JBMP) and Batemans Marine Park (BMP) (Fig. 3.1). Both parks are multiple zone MPAs and have multiple management zones in place. In this study, all zones in MPAs other than NTZs were treated as PPAs as management on demersal habitats are similar. For a detailed outline of all restrictions and zoning see Read and West (2010). JBMP zoning came into effect on 1 October 2002 (Lynch 2006) and BMP zoning came into effect on June 2007 (Kelaher et al. 2014). All OA locations are outside the two MPAs (Fig. 3.1).
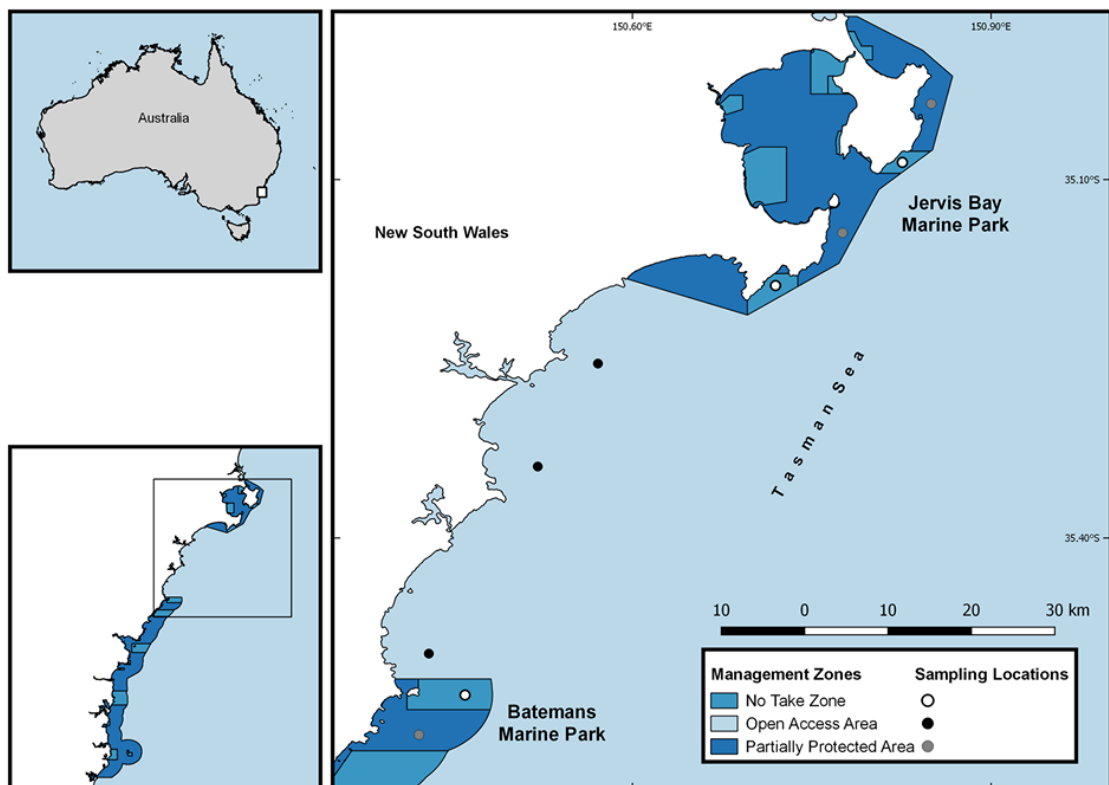
Figure 3.1: Map of study site and sampling locations within Jervis Bay Marine Park, Batemans Bay Marine Park, and surrounding open access areas. Sampling locations from North to South: in Open Access Areas were Cudmirra, Ulladulla and Bawley Point; In Partially Protected Areas were Lamond Head, Bowen Island, Kiola, Point Perpendicular, St Georges Head and Brush Island.

Stereo-Baited Remote Underwater Videos (hereafter BRUVs) were deployed, from the 27 August to 13 December 2013, to survey the demersal soft sediment fish assemblages found in open coastal waters on un-vegetated soft sediments in waters of 40–60 m depth. Eight stereo-BRUVs were deployed at each location with a minimum

spacing of 200 m between replicates. The BRUVs were deployed as in Chapter 2 and the same MaxN, Total MaxN, species richness and frequency of occurrence metrics were recorded from each deployment. In contrast to Chapter 2, where it was difficult to identify juvenile *Platycephalus caeruleopunctatus* (bluespotted flathead) and *Platycephalus grandispinis* (longspine flathead*)* individuals smaller than ~20 cm, greater visibility in the deeper coastal waters in this chapter meant that reliably distinguishing between the two species was possible. There also appeared to be no juvenile *P. caeruleopunctatus* at the depths sampled.   Therefore, a separate MaxN for *P. caeruleopunctatus* and *P. grandispinis* was taken. The total length (TL, from the tip of snout to centre of the caudal fin) of individual *P.* grandispinis was measured at MaxN. Separate length measurements were also taken for *P. caeruleopunctatus* at MaxN (details as per Chapter 2). The size of largest *P. caeruleopunctatus* (the main commercially recreationally targeted species) observed in each video was measured and included in the *P. caeruleopunctatus* length dataset as a complementary measure to assess the differences in largest fish sizes among zones.

Experimental design

The design had two factors: Zone (A fixed orthogonal factor with 3 levels, NTZ, PPA and OA) and Location (a random nested factor with 3 levels in each of NTZ, PPA, and OA; Location nested in Zone). Each location had 8 replicate BRUVs samples planned with 24 from each zone type and a total of 72 BRUV deployments (Fig. 3.2). Of the planned BRUV deployments undertaken, a number failed due to poor visibility, equipment issues or being tipped over in large swell and we resampled these 'failed' deployments a second time.   After resampling, a total of 56 samples (Table 3.1) were deemed successful and analysed.

Statistical analysis

Multivariate PERMANOVA analyses (Clarke 1993, Anderson, Gorley et al. 2008) of assemblage data used PRIMER-E v7 with type III sums of squares, 9999 permutations and the design given above. Multivariate differences in assemblages across management zones were tested using Bray-Curtis dissimilarity values calculated from untransformed data. I visualized assemblage patterns using Non-Metric Multi-Dimensional Scaling ordination. Yellowtail scad (*Trachurus novaezelandiae*), and slimy
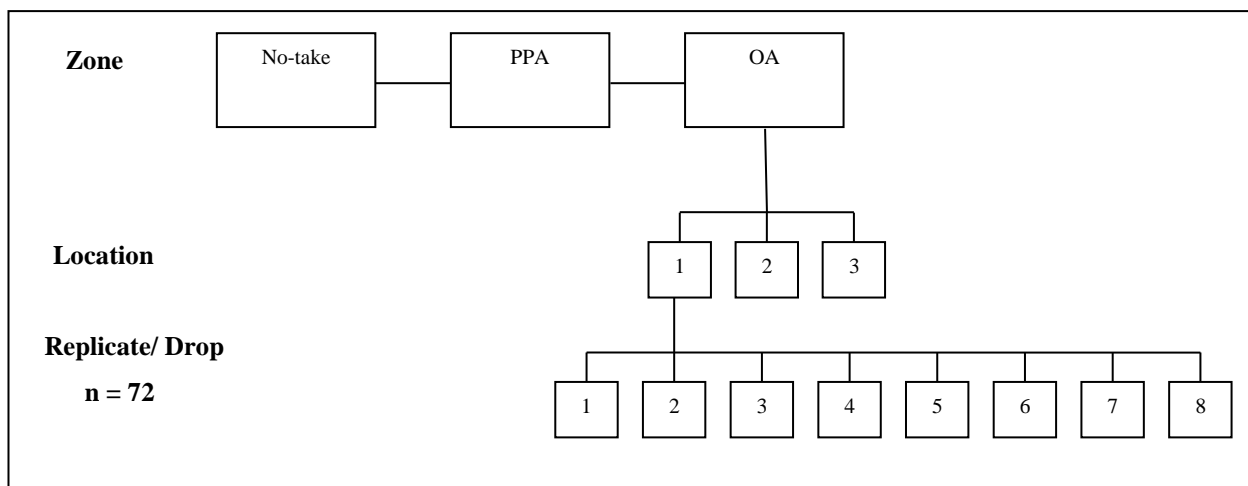
Figure 3.2: Experimental design to assess ecological changes in the diversity, relative abundance and size of soft sediment fishes in no-take zones, partially protected areas (PPA) within two marine protected area (MPAs) and open access (OA) areas outside the MPAs.

Table 3.1. Site and number of successful BRUV deployments at each location. All deployments were stereo camera BRUVs.

Management Type	Location	Deployments
No Take Zone	Point Perpendicular	6
	St Georges Head	5
	Brush Island	8
Partially Protected Area	Lamond Head	5
	Bowen Island	7
	Kiola	8
Open Access Area	Cudmirra	6
	Ulladulla	3
	Bawley Point	8

mackerel (*Scomber australasicus*) were excluded from multivariate analysis of community composition and Total MaxN analyses, as they were not considered benthic species (i.e. soft sediment fishes) and may have a disproportionate effect on the data set owing to their high abundance.

Differences in abundances of individual species that met a frequency of occurrence threshold of appearing on 25% or greater of the BRUVs samples across management zones were analysed using PERMANOVA. Euclidean distance was used as the measure of dissimilarity for univariate analyses. Total MaxN, species richness, Shannon diversity and Pielou's evenness measure (see Chapter 2 for details on indices) were compared in the same manner. PERMDISP, a test that is equivalent to Levene's test for heterogeneity when used with univariate data (Anderson 2006, Harvey et al. 2012), was used to test homogeneity of variance. Where PERMANOVA and PERMDISP both had significant *p*-values, the data was fourth root transformed to eliminate or reduce the significant dispersion and the data reanalysed. Post hoc pairwise comparisons were made by zone type, where zone was found to be significant in the main PERMANOVA analysis.

I tested dissimilarities in the cumulative length distribution of flathead across the three management zones (NTZ vs PPA, NTZ vs OA, and PPA vs OA) using multiple two sample non-parametric Kolmogorov–Smirnov tests (KS tests, see Chapter 2 for full KS test details). The data contained no ties which enabled exact p-values to be calculated without the need for bootstrapping (Ogle 2016). A Benjamini–Hochberg correction to control the false discovery rate in multiple tests was applied to the resulting *p*-values (Benjamini and Hochberg 1995). The correction was applied using p.adjust in the kSamples package in R (R Development Core Team 2014). Lengths for *Platycephalus grandispinis* at the genus level were aggregated by zone type and for visual comparison of length frequency distributions, they were grouped into 2 cm increments. *Platycephalus caeruleopunctatus* lengths were compared using the same processes described above, however the largest specimen on each drop was included in this analysis.

3.3 Results

The demersal fish assemblage on soft sediments was characterised by a small number of frequently observed species. Two species were observed on the majority of video samples; longspine flathead (*Platycephalus grandispinis*) on 88% of video and bluespotted flathead (*Platycephalus caeruleopunctatus*) on 73% (Table 3.2). The ocean jacket (*Nelusetta ayraudi*) and fiddler ray (*Trygonorrhina fasciata)* were also common, appearing on more than 50% of samples (Table 3.2). Nine species in total appeared on 25% or more of the video samples (Table 3.2). There were 31 fish species observed in total and 9 of these were only recorded in one or two deployments (singletons and doubletons, Table 3.3).

The abundance dataset was also characterized by a few dominant taxa. School whiting (*Sillago flindersi*) and silver trevally (*Pseudocaranx georgianus*) were the two most abundant species, and combined with ocean jacket, longspine flathead and bluespotted flathead made up 82% of the total abundance (Table. 3.3). Another 11% of the total was made up by 6 species (Table 3.3). The only species that was commonly encountered but did not appear in the nine most abundant species was the sixspine leatherjacket (*Meuschenia freycineti*), a species that often appear alone. In contrast, the only fish that was in the nine most abundant fish and not among the most common fish was the Australian Sardine (*Sardinops sagax*) which appeared in large numbers on a single video sample.

There was evidence of separation in demersal fish assemblages among open access fished zones (OA), partially protected areas (PPA) and no-take zones (NTZs) in the nMDS plot (Fig. 3.3). Multivariate tests confirmed this separation, showing that zone had a significant effect on the demersal fish assemblage, with a distinct assemblage composition observed among each zone type (Table 3.4). There was no difference in Shannon diversity or Pielou's evenness measure among zones, indicating that diversity was equivalent and that the communities in each were similarly even (Table 3.5). The assemblage differences among zones appeared to be due to the contribution of several species and the species composition overall. Overall, there were on average more species detected in NTZs, however, this difference was not statistically significant as there was

substantial variation among locations within each zone, with the greatest species richness observed at Point Perpendicular (NTZ) and Bawley Point (OA) locations (Fig. 3.4, Table 3.5). There was also no difference across zones in Total MaxN, while differences between locations were highly variable (Table 3.5, Fig. 3.4). Total MaxN can be strongly influenced by abundant schooling species and the Total MaxN data in this study reflected that. I observed the highest Total MaxN at Bawley Point (OA) and Brush Island (NTZ) (Fig. 3.4); both had very large abundances of a single species driving high abundance results (school whiting *S. flindersi* and silver trevally *P. georgianus* respectively, Fig. 3.5).

Table 3.2: Frequency of occurrence (% of samples present) for each species for all BRUV deployments and by zone (NTZ and FZ, PPA and NTZ). Hatched line indicates frequency cut off point for univariate analyses. See Table 3.3 for scientific names.

All		FZ		PPA		NTZ	
Longspine Flathead	87.5	Longspine Flathead	70.6	Longspine Flathead	100.0	Longspine Flathead	89.5
Bluespotted Flathead	73.2	Bluespotted Flathead	70.6	Fiddler Ray	70.0	Bluespotted Flathead	89.5
Ocean Jacket	62.5	Ocean Jacket	70.6	Bluespotted Flathead	60.0	Ocean Jacket	89.5
Fiddler Ray	50.0	Tiger Flathead	58.8	Sixspine Leatherjacket	45.0	Fiddler Ray	68.4
Sixspine Leatherjacket	37.5	Yellowtail Scad	52.9	School Whiting	35.0	Velvet Leatherjacket	68.4
Tiger Flathead	32.1	School Whiting	41.2	Ocean Jacket	30.0	Silver Trevally	47.4
Velvet Leatherjacket	30.4	Sixspine Leatherjacket	29.4	Port Jackson Shark	20.0	Sixspine Leatherjacket	36.8
Silver Trevally	26.8	Silver Trevally	23.5	Red Gurnard	20.0	Red Gurnard	26.3
School Whiting	25.0	Port Jackson Shark	11.8	Tiger Flathead	15.0	Tiger Flathead	26.3
Yellowtail Scad	23.2	Grey Morwong	11.8	Yellowtail Scad	15.0	Barracouta	26.3
Port Jackson Shark	16.1	Slimey Mackerel	11.8	Velvet Leatherjacket	15.0	Shovelnose Ray	21.1
Red Gurnard	16.1	Fiddler Ray	5.9	Barracouta	15.0	Port Jackson Shark	15.8
Barracouta	16.1	Velvet Leatherjacket	5.9	Shovelnose Ray	15.0	Gummy Shark	15.8
Shovelnose Ray	12.5	Barracouta	5.9	Southern Eagle Ray	15.0	Black Reef Jacket	10.5
Gummy Shark	7.1	Stingaree	5.9	Common Sawshark	15.0	Yellowtail Scad	5.3
Black Reef Jacket	5.4	Pink Snapper	5.9	Silver Trevally	10.0	Stingaree	5.3
Grey Morwong	5.4	Smalltooth Flounder	5.9	Stingaree	5.0	Grey Morwong	5.3
Southern Eagle Ray	5.4	Australian Sardine	5.9	Gummy Shark	5.0	Pink Snapper	5.3
Common Sawshark	5.4	Garden Eels	5.9	Black Reef Jacket	5.0	Smalltooth Flounder	5.3
Stingaree	5.4	Eastern SBoxfish	5.9	Yellowstriped Jacket	5.0	Greynurse Shark	5.3
Pink Snapper	3.6	Bluespotted Goatfish	5.9	Southern Sawshark	5.0	Yellowtail Kingfish	5.3
Smalltooth Flounder	3.6	Red Gurnard	0.0	Grey Morwong	0.0	School Whiting	0.0
Slimey Mackerel	3.6	Shovelnose Ray	0.0	Slimey Mackerel	0.0	Southern Eagle Ray	0.0
Australian Sardine	1.8	Gummy Shark	0.0	Pink Snapper	0.0	Common Sawshark	0.0
Garden Eels	1.8	Black Reef Jacket	0.0	Smalltooth Flounder	0.0	Yellowstriped Jacket	0.0
Eastern SBoxfish	1.8	Southern Eagle Ray	0.0	Australian Sardine	0.0	Southern Sawshark	0.0
Greynurse Shark	1.8	Common Sawshark	0.0	Garden Eels	0.0	Slimey Mackerel	0.0
Yellowstriped Jacket	1.8	Greynurse Shark	0.0	Eastern SBoxfish	0.0	Australian Sardine	0.0
Southern Sawshark	1.8	Yellowstriped Jacket	0.0	Bluespotted Goatfish	0.0	Garden Eels	0.0
Yellowtail Kingfish	1.8	Southern Sawshark	0.0	Greynurse Shark	0.0	Eastern SBoxfish	0.0
Bluespotted Goatfish	1.8	Yellowtail Kingfish	0.0	Yellowtail Kingfish	0.0	Bluespotted Goatfish	0.0

Table 3.3: All species recorded and their average MaxN per BRUV deployment in each zone (NTZ and FZ) and their total relative abundance (or total count) recorded by BRUV.

Common Name	Family	Genus	Species	Average MaxN			Total Count
				FZ	PPA	NTZ	
Yellowtail Scad	Carangidae	*Trachurus*	*novaezelandiae*	40.76	0.70	0.11	709
School Whiting	Sillaginidae	*Sillago*	*flindersi*	18.29	3.20	0.00	375
Silver Trevally	Carangidae	*Pseudocaranx*	*georgianus*	1.29	0.25	14.58	304
Ocean Jacket	Monacanthidae	*Nelusetta*	*ayraudi*	1.76	0.60	10.16	235
Longspine Flathead	Platycephalidae	*Platycephalus*	*grandispinis*	2.06	5.80	3.84	224
Bluespotted Flathead	Platycephalidae	*Platycephalus*	*caeruleopunctatus*	2.76	1.00	2.37	112
Australian Sardine	Cluepeidae	*Sardinops*	*sagax*	3.59	0.00	0.00	61
Fiddler Ray	Rhinobatidae	*Trygonorrhina*	*fasciata*	0.06	1.30	1.05	47
Velvet Leatherjacket	Monacanthidae	*Meuschenia*	*scaber*	0.06	0.15	1.79	38
Tiger Flathead	Platycephalidae	*Platycephalus*	*richardsoni*	1.18	0.15	0.32	29
Sixspine Leatherjacket	Monacanthidae	*Meuschenia*	*freycineti*	0.41	0.45	0.53	26
Port Jackson Shark	Heterodontidae	*Heterodontus*	*portusjacksoni*	0.12	0.25	0.21	11
Red Gurnard	Triglidae	*Chelidonichthys*	*kumu*	0.00	0.20	0.32	10
Barracouta	Gemplidae	*Thyrsites*	*atun*	0.06	0.15	0.26	9
Shovelnose Ray	Rhinobatidae	*Aptychotrema*	*rostrata*	0.00	0.15	0.21	7
Garden Eels	Congridae	*Gorgasia*	*spp.*	0.29	0.00	0.00	5
Gummy Shark	Triakidae	*Mustelus*	*antarcticus*	0.00	0.05	0.21	5
Black Reef Jacket	Monacanthidae	*Eubalichthys*	*bucephalus*	0.00	0.05	0.16	4
Grey Morwong	Cheilodactylidae	*Nemadactylus*	*douglasii*	0.12	0.00	0.11	4
Southern Eagle Ray	Myliobatidae	*Myliobatis*	*tenuicaudatus*	0.00	0.15	0.00	3
Common Sawshark	Pristiophoridae	*Pristiophorus*	*cirratus*	0.00	0.15	0.00	3
Stingaree*	Urolophidae spp.	All	All	0.06	0.05	0.05	3
Pink Snapper	Sparidae	*Chrysophrys*	*auratus*	0.06	0.00	0.05	2
Smalltooth Flounder	Paralichtyidae	*Pseudorhombus*	*aenynsii*	0.06	0.00	0.05	2
Slimey Mackerel	Scombridae	*Scomber*	*australasicus*	0.12	0.00	0.00	2
Eastern SBoxfish	Aracanidae	*Anoplocapros*	*inermis*	0.06	0.00	0.00	1
Greynurse Shark	Odontaspididae	*Carcharias*	*taurus*	0.00	0.00	0.05	1
Yellowstriped Jacket	Monacanthidae	*Meuschenia*	*flavolineata*	0.00	0.05	0.00	1
Southern Sawshark	Pristiophoridae	*Pristiophorus*	*nudipinnis*	0.00	0.05	0.00	1
Yellowtail Kingfish	Carangidae	*Seriola*	*lalandi*	0.00	0.00	0.05	1
Bluespotted Goatfish	Mullidae	*Upeneichthys*	*vlamingii*	0.06	0.00	0.00	1

*Family level count; most likely *Trygonoptera testacea* but may include *Urolophus sufflavus, Urolophus kapalensis* and *Urolophus cruciatus.*

The two pelagic species (*Scomber australasicus* and *Trachurus novaezelandiae*) were excluded from multivariate analysis of demersal community composition and univariate Total MaxN analyses and retained in the species richness count.
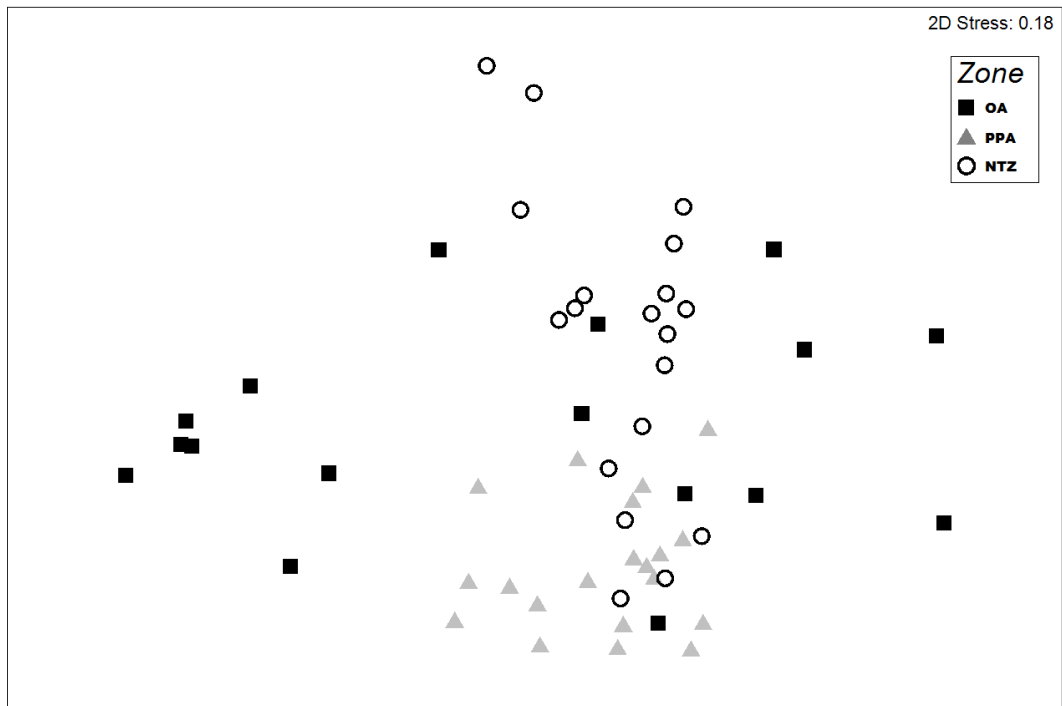
Figure 3. 3: Differences in soft sediment fish assemblages between management zones using a nonmetric multidimensional scaling (nMDS) plot based on Bray-Curtis similarities. NTZ (n = 19), PPA (n = 20), OA (n = 18).

Table 3.4: Results of assemblage comparisons between open access areas (OA), partially protected areas (PPA) and no-take zones (NTZ). Significant P values (<0.05) shown in bold.

Source	df	SS	MS	Pseudo-F	P(perm)
Zo	2	28383	14191	2.7116	**0.0011**
Lo(Zo)	6	32354	5392.3	2.584	**0.0001**
Res	47	98082	2086.8		
Total	55	167000			
		Pairwise Test		t	P(perm)
		OA ≠ NTZ		1.372	**0.050**
		OA ≠ PPA		1.805	**0.021**
		NTZ ≠ PPA		1.805	**0.049**
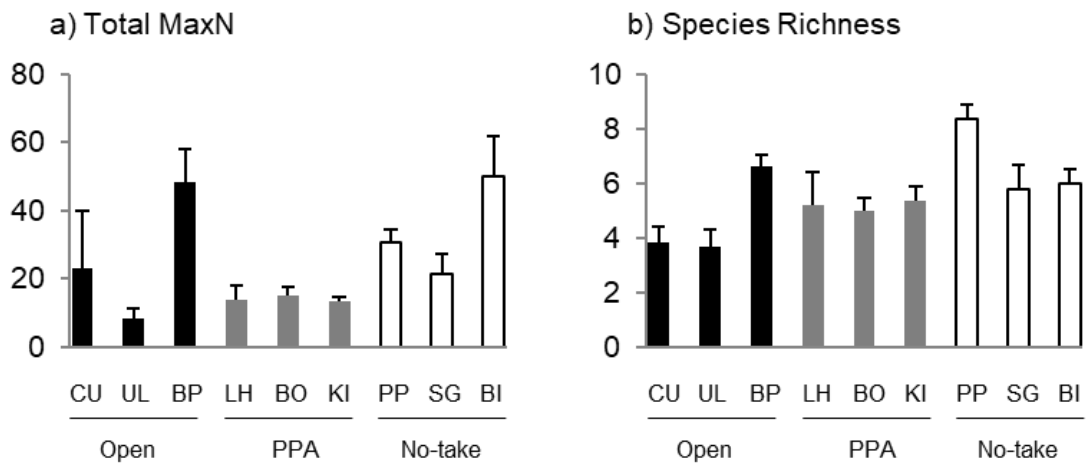
Figure 3. 4: a) Average total abundance (MaxN) and b) species richness, at each location (n= 3–8 replicates per site, see Table 3.1). Error bars are SE. No-take zones are in white, partially protected areas in grey and open access areas in black. CU = Cudmirra, UL= Ulladulla, BP = Bawley Point, LH =Lamond Head, BO = Bowen Island, KI = Kiola, PP = Point Perpendicular, SG = St Georges Head and BI = Brush Island.

Table 3.5: Results of assemblage comparisons across open access areas (OA), partially protected areas (PPA) and no-take zones (NTZ) for a) total abundance (TMaxN), b) species richness (SR), c) Shannon diversity and d) Pielou's evenness measure using permutational analysis of variance (PERMANOVA). Significant P values (<0.05) shown in bold.

Source	df	SS	MS	Pseudo-F	P(perm)	Source	df	SS	MS	Pseudo-F	P(perm)
a) MaxN						**b) Species Richness**					
Zo	2	3774.60	1887.30	1.65	0.281	Zo	2	37.3	18.63	1.97	0.232
Lo(Zo)	6	7048.80	1174.80	2.34	**0.047**	Lo(Zo)	6	58.9	9.81	3.97	**0.003**
Res	47	23587.00	501.85			Res	47	116.2	2.47		
Total	55	36085.00				Total	55	204.2			
c) Shannon Diversity						**d) Pielou's Evenness**					
Zo	2	9.27	4.64	3.29	0.113	Zo	2	0.2	0.08	1.83	0.246
Lo(Zo)	6	8.33	1.39	0.77	0.607	Lo(Zo)	6	0.3	0.04	1.21	0.315
Res	47	85.21	1.81			Res	47	1.7	0.04		
Total	55	103.81				Total	55	2.1			
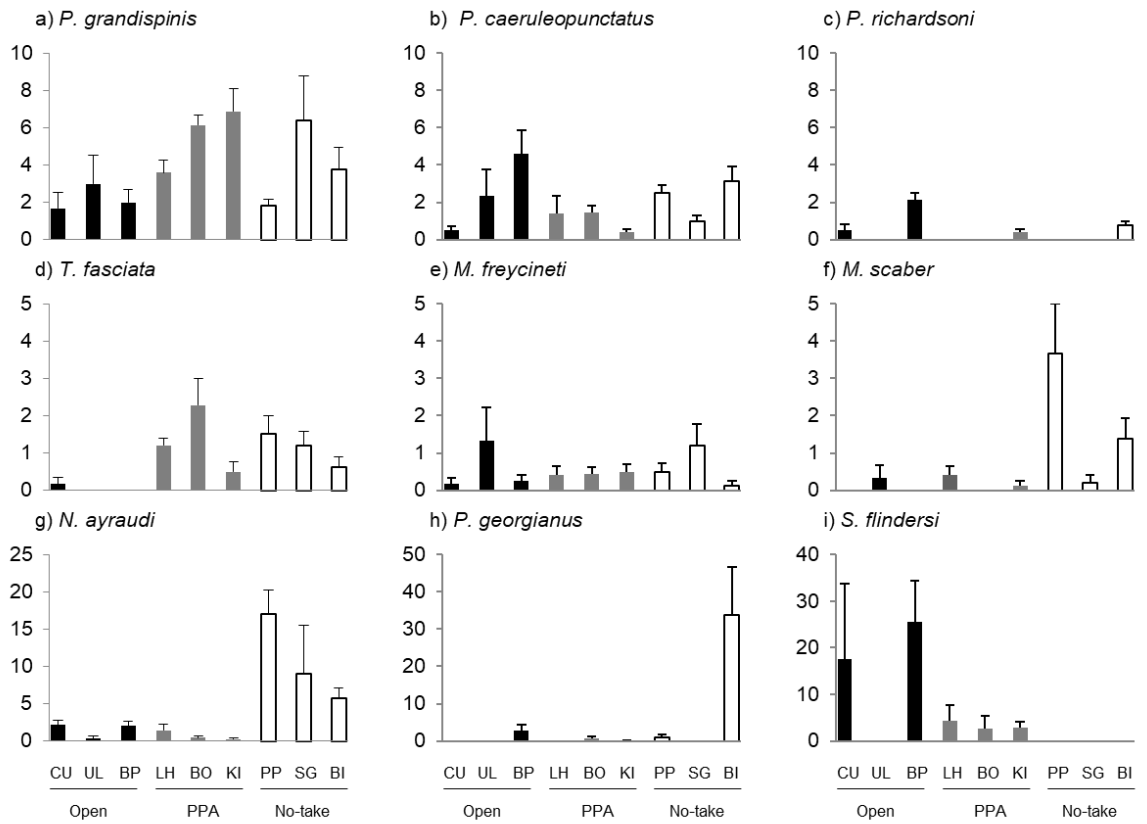
Figure 3.5: Average abundance (MaxN) for each species at each location (n= 3–8 replicates per site) Error bars are SE. No-take zones are in white, partially protected areas in grey and open access areas in black. CU = Cudmirra, UL= Ulladulla, BP = Bawley Point, LH =Lamond Head, BO = Bowen Island, KI = Kiola, PP = Point Perpendicular, SG = St Georges Head and BI = Brush Island.

Table 3.6: Results of abundance (MaxN) comparisons between open access areas (OA), partially protected areas (PPA) and no-take zones (NTZ) for individual species using univariate PERMANOVA. Significant P values (<0.05) shown in bold.

Source	df	SS	MS	Pseudo-F	P(perm)	df	SS	MS	Pseudo-F	P(perm)	df	SS	MS	Pseudo-F	P(perm)
	a) *P. grandispinis*					b) *P. caeruleopunctatus*					c) *P. richardsoni*				
Zo	2	90.80	45.40	2.94	0.102	2	19.94	9.97	0.79	0.519	2	5.09	2.54	0.92	0.510
Lo(Zo)	6	94.87	15.81	2.05	0.076	6	78.27	13.05	3.67	**0.004**	6	17.38	2.90	8.64	**0.0001**
Res	47	361.80	7.70			47	167.21	3.56			47	15.75	0.34		
Total	55	586.00				55	278.00				55	43.98			
	d) *T. fasciata*					e) *M. freycineti*					f) *M. scaber*				
Zo	2	14.60	7.30	3.04	0.139	2	0.29	0.15	0.14	0.942	2	30.36	15.18	2.63	0.083
Lo(Zo)	6	14.85	2.48	2.82	**0.020**	6	6.72	1.12	2.49	**0.033**	6	35.90	5.98	3.76	**0.005**
Res	47	41.24	0.88			47	21.09	0.45			47	74.75	1.59		
Total	55	71.55				55	27.93				55	146.21			
	g) *N. ayraudi*					g) *P. georgianus*					h) *S. flindersi*				
Zo	2	12	6.15	8.96	**0.013**	2	2	0.76	1.22	0.357	2	1789	894.56	3.69	0.106
Lo(Zo)	6	4	0.70	2.18	0.062	6	4	0.64	2.35	**0.044**	6	1446	241.00	0.88	0.511
Res	47	15	0.32			47	13	0.27			47	12817	272.70		
Total	55	33				55	19				55	17646			

Pairwise Tests	t	P(perm)
OA = NTZ	2.48	0.064
OA = PPA	1.36	0.242
FZ ≠ PPA	2.88	**0.045**

Effects of zoning were statistically significant for the abundance of just one individual taxon, *Nelusetta ayraudi* (Table 3.6). Abundances of *N. ayraudi* in all NTZ locations were relatively high and lowest in PPA, however abundances were only significantly greater among NTZs and PPAs (Table. 3.6, Fig. 3.5g).

Abundances of *Platycephalus grandispinis* were relatively even between locations (Fig. 3.5a) and were not influenced by zone (Table 3.6). Likewise, no effect of zone on abundances of *Platycephalus caeruleopunctatus* were detected (Table 3.6, Fig. 3.5b). Interestingly, the PPA locations all had high abundances of *P. grandispinis* and low abundances of *P. caeruleopunctatus*. There did not appear to be any juvenile *P. caeruleopunctatus* present in the sampled locations. I did not observe tiger flathead (*Platycephalus richardsoni)* north of Cudmirrah and abundances were low and patchy across the other locations with no effect of zone evident (Table 3.6, Fig. 3.5c).

The abundances of velvet leatherjacket (*Meuschenia scaber*) and sixspine leatherjacket (*Meuschenia freycineti)* were not affected by zone (Table 3.6). There were very low numbers of *M. scaber* observed, with the exception of Point Perpendicular NTZ where they were found on all video samples and at Brush Island NTZ where patchy numbers were recorded (Fig. 3.5f). Abundances of *M. freycineti* were low relatively consistently at all locations (Fig. 3.5e).

No differences in the abundance of fiddler rays (*Trygonorrhina fasciata)* by zone were detected statistically (Table 3.6). This is a surprising result given that there was only one *T. fasciata* observed in all OA samples and they were present in all other locations (Table 3.2), although in patchy numbers in PPAs and NTZs (Fig. 3.5d, Table 3.3). There were very large numbers of *Pseudocaranx georgianus* observed at Brush Island NTZ and very low abundance elsewhere and consequently there was no effect of zone (Table 3.6, Fig. 3.5) Conversely, *S. flindersi* were not observed in NTZ locations and were present in all PPAs in low numbers and in two OAs areas, Cudmirrah and Bawley point in high numbers. However, in all the PPAs the total abundance at each location came from one sample, almost the entire count of *S. flindersi* at Cudmirrah was made up by one large school on one video sample and at Bawley Point, samples either had zero or very large numbers of fish. All three species, but in particular *S. flindersi* and *P. georgianus,* had highly variable abundances and patchy presence between samples and locations.

A total of 97 *Platycephalus grandispinis* were measured from the 56 BRUV deployments and the length frequency distribution was unimodal with the exception of OA, where it was skewed left and no large fish were measured. Fish length decreased with level of fishing pressure (Fig. 3.6), with the mean length highest in NTZs and lowest in OAs (Table 3.7). The difference in average length between NTZ and OA zones was 24 mm, the difference between NTZ and PPA was 8 mm (Table. 3.7) The median, variance and shape of the cumulative length frequency distribution obtained for *P. grandispinis* at the genus level differed significantly between NTZs and both other zone types (Fig. 3.6, Table 3.7). The median length was greater in PPAs the difference among OA and PPAs was not significant (Fig. 3.6, Table 3.7). The mean length by zone for *Platycephalus caeruleopunctatus* based on measurements of 92 fish also decreased with fishing pressure, with the highest mean in NTZ and lowest in OA (Table 3.7, Fig. 3.7). However,

the shape of the distribution was similar particularly for PPA and NTZs and there was no difference detected by zone in median, variance and shape of the cumulative length frequency distribution obtained (Table 3.7).

Table 3.7: Average length (±SE), minimum and maximum length by zone type. Results of Kolmogorov-Smirnov (KS) tests of differences of cumulative length distributions between NTZ, PPA and OA for longspine flathead (*Platycephalus grandispinis*) and bluespotted flathead (*P. caeruleopunctatus*). All measurements are in millimetres.

	Longspine			Bluespotted		
	OA	PPA	NTZ	OA	PPA	NTZ
Max	248	191	305	546	516	571
Min	179	317	176	300	320	331
Median	224	236	248	382	387	402
Ave (SE)	220 (7)	236 (4)	244 (5)	385 (10)	403 (14)	418 (9)
N	10	43	44	35	17	40
KS test	D	p-value		D	p-value	
NTZ vs PPA	0.292	0.0496		0.262	0.4863	
NTZ vs OA	0.5	0.0496		0.3	0.1642	
PPA vs OA	0.342	0.2383		0.21	0.6170	
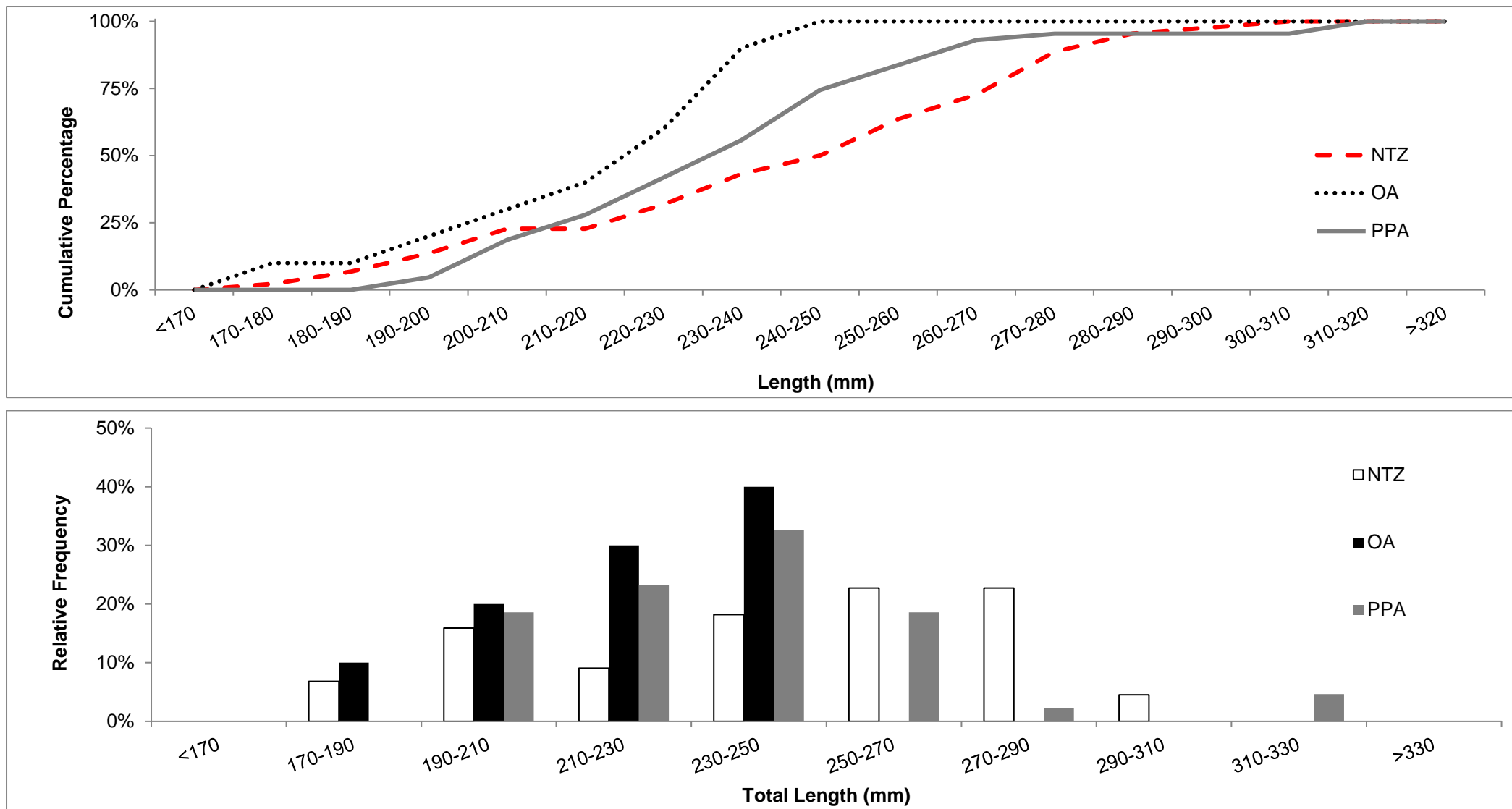
Figure 3.6: *Platycephalus grandispini*s cumulative length frequency distributions by management zone (top) and length frequency distributions in 20 mm increments by management zone (Bottom); No-take zone (NTZ, n=44), Partially protected area (PPA, n =43), Open access area (OA, n=10).
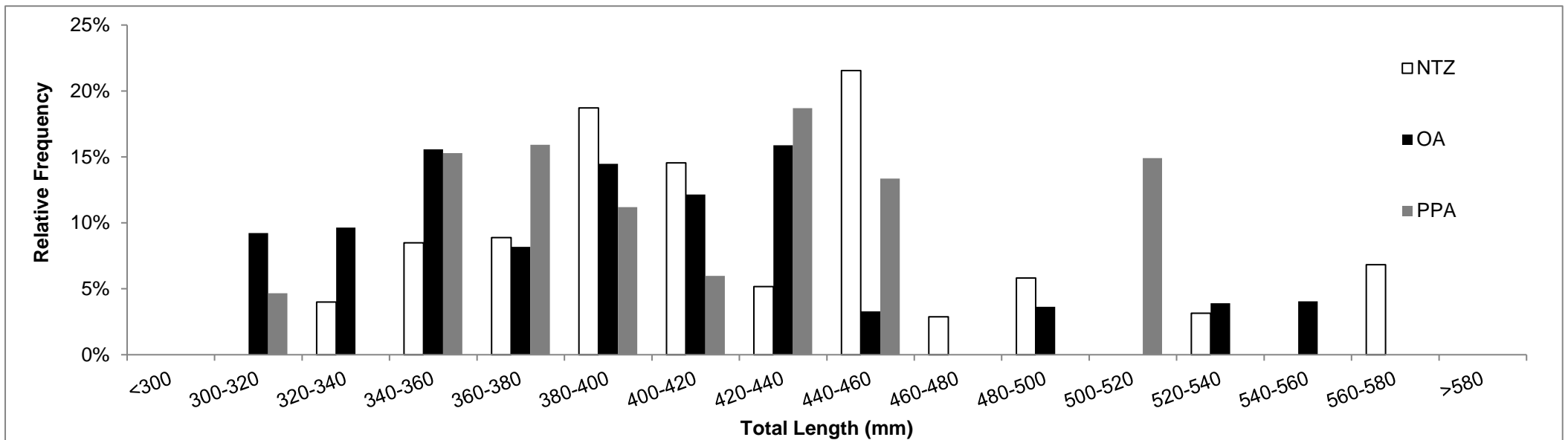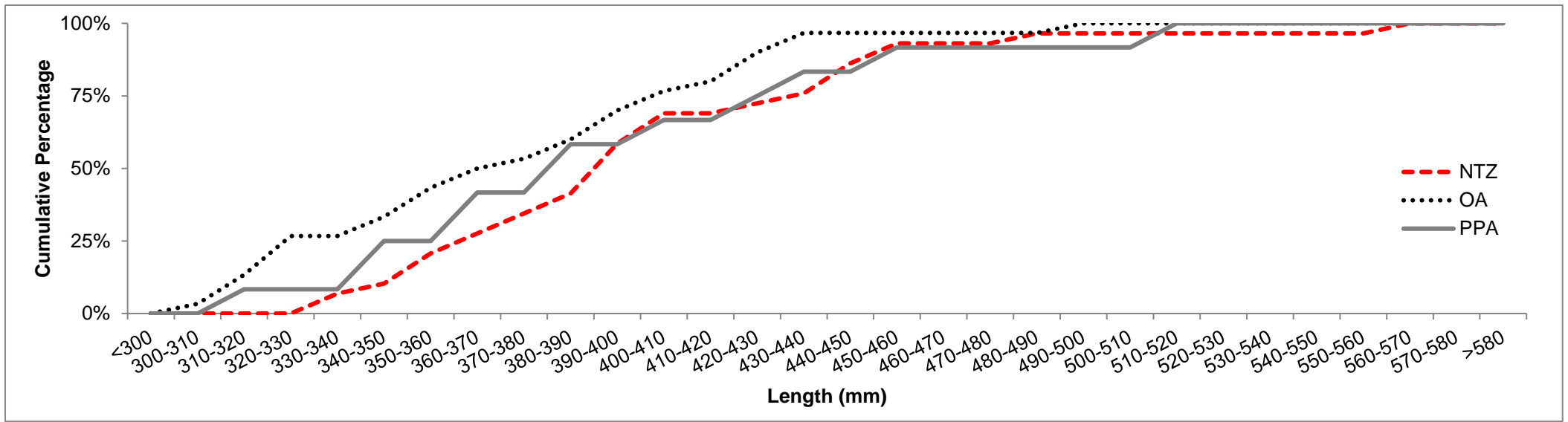
Figure 3.7: *Platycephalus caeruleopunctatus* cumulative length frequency distributions by management zone (Top) and length frequency distributions, in 20 mm increments, by management zone (Bottom); NTZ (n=40), PPA (n=17), OA (n=35)

3.4 **Discussion**

My results revealed an effect of management zone on demersal fish assemblages associated with coastal soft sediments. As predicted, multivariate comparisons revealed that there was a clear distinction in composition of assemblages among no-take zones (NTZ), partially protected areas (PPA) and fished open access areas (OA). However, OA had considerably more variation between replicates than either of the other two management types. Comparisons of Shannon diversity and Pielou's evenness suggested that, although multivariate composition of each assemblage was different, the level of diversity was equivalent in each. Contrary to my predictions, there were no differences in total abundance or abundance of the most common individual species; *Platycephalus grandispinis*, *Platycephalus caeruleopunctatus* and *Trygonorrhina fasciata.* The single exception at the species level was the ocean jacket (*Nelusetta ayraudi)* which showed a difference in abundance by zone. The number of *N. ayraudi* observed in no-take zone (NTZ) locations was much higher than in partially protected and open access areas, however, the difference was only significant between the NTZs and partially protected areas (PPAs). There was also no effect of zone on abundances of species that were not as common, though still observed relatively frequently (on 25–38% of samples); *Meuschenia freycineti, Platycephalus richardsoni, Meuschenia scaber*, *Sillago flindersi* and *Pseudocaranx georgianus*. Although abundances of *P. grandispinis* were the same among management zones, they were larger in NTZs than in both PPZ and OA. This was not the case for *P. caeruleopunctatus*, where no impact of zone on length was observed.

The abundances of *Nelusetta ayraudi* was substantially higher in no-take zones compared to PPAs and OAs. This difference was spatially consistent with all three no-take locations recording a higher abundance than at any location in PPAs and OAs. An elevated abundance in no-take zones is not surprising as this species does appear to come under considerable fishing pressure. For example in New South Wales this species is fully fished (Stewart et al. 2015) and although this species is not considered to be over fished, it is caught in large quantities by both the commercial trap and line fishery (Stewart et al. 2015) and recreational fishers (West et al. 2015).

Effects of zoning on the most common species in the assemblage, *Platycephalus grandispinis* were also observed. The response, however, was noted in the sizes of individuals rather than their relative abundance. Supporting my prediction, on average larger individuals of *P. grandispinis* were observed in NTZs than in both other zones. However, there were no differences in lengths between PPAs and OA areas, suggesting that larger fish were being removed in both zones. The pattern of the largest fish within NTZs and the smallest in OAs aligned with my predictions, however the magnitude of difference between zones was relatively small. There is no stock assessment for this species in NSW (Stewart et al. 2015) and it is difficult to know the level of commercial fishing pressure *P. grandispinis* is experiencing, however my results suggest that it is sufficient for management zones to have a small effect. In addition, *P. grandispinis* is also caught unintentionally by fishers targeting *Platycephalus caeruleopunctatus* who often assume they are *P. caeruleopunctatus* (Authors pers. obs.) and as a result, may experience some degree of recreational fishing mortality, especially as they are prone to being gut hooked (Authors pers. obs.). This species is relatively long lived and has been recorded living to 16 years of age (Barnes et al. 2011). It has been reported previously that longer living temperate reef species in the region have had the older age classes removed through fishing (Stewart 2011). Similarly, my data provide evidence of age class truncation for *P. grandispinis* in OA areas (there were no fish over 25 cm), an effect consistent with continuous removal of larger fish (Beamish et al. 2006, Longhurst 2006). Although lower sample size in OAs may have played a part in my results, it does suggest that length differences across zones for this species should be investigated further.

There was no response by management zone for *Platycephalus caeruleopunctatus,* in either length or abundance. There are a number of lines of evidence that this species experiences a high level of fishing pressure in the study region. First, *P. caeruleopunctatus* is a primary species in the ocean trawl fishery and currently listed as fully fished in NSW (Stewart et al. 2015). Second, *P. caeruleopunctatus* is the third most caught recreational species in NSW (West et al. 2015) and the south coast region, which includes the area sampled in this study, accounts for the largest proportion of the annual recreational catch (West et al. 2015). Third, it appears that juveniles are rarely present at this depth and likely prefer shallower depths (e.g. Chapter 2); I recorded very few undersized *P. caeruleopunctatus* (<9%) and none under 30 cm (and therefore most *P.*

caeruleopunctatus at these depths can legally be retained by recreational fishers). It has been reported that *P. caeruleopunctatus* are generally only discarded after capture by recreational fishers if undersized (West et al. 2015). So, it then follows that when this species is caught in the study depths, mortality is close to 100%. Given this expected level of fishing pressure, this species appears to be an obvious candidate to show a response to protection. Why I failed to detect a response here is unclear, however it suggests that (A) the fishing effort on this species may be at an ecologically sustainable level or potentially that (B) the area under protection is not large enough (i.e. the fish are regularly moving over larger areas than the zones cover; see Chapter 5 for further discussion) or (C) illegal fishing may be occurring within the MPAs (see compliance discussion below).

The lack of difference in abundances by management zone for any of the remaining individual species was another unexpected finding, given the clear differences at the assemblage level. The patchy nature of the abundance of several taxa likely explains this pattern. For example, although there were almost no *Trygonorrhina fasciata* recorded in open access areas (one in a total of seventeen BRUV deployments), they were present at all no-take and partially protected locations. I observed up to 3 individuals in a single NTZ BRUV deployment and up to 6 in PPA BRUV deployments. However, they were present in highly variable numbers within each management type among locations and among samples. Consequently, comparisons across zones were not statistically significant due to high variability surrounding mean estimates within each management zone resulting in low statistical power (despite the current study being logistically large and costly and difficult to carryout). Due to the lack of statistical power, a precautionary approach would be appropriate here and I would suggest that more sampling (i.e. primarily more sites) is required to determine whether the observed pattern of difference is real. The lack of detections in OAs combined with the undefined stock status of *T. fasciata* in NSW (Stewart et al. 2015) suggests that further investigation of abundances for this species is warranted.

Three other species were recorded in highly variable numbers between samples and locations. *Sillago flindersi,* and *Pseudocaranx georgianus* were not observed on the majority of video samples (42/56 and 41/56 samples respectively) but on a small subset of samples were observed in very high numbers. Often these species arrived at the BRUV

96

in large schools of up to 100 fish (mean±SD of 26 ±28 & 14±20 fish respectively). *Meuschenia scaber* were also observed to be highly variable in abundance although they did not form such large schools; generally, less than 10 individuals in a school. The schooling behaviour of these three-species resulted in bimodal, zero-inflated datasets that made analyses and comparisons across zones difficult, and therefore caution should be taken in interpretation of results for these species. It is apparent that owing to the schooling nature of many species in the assemblage very large effect sizes are required to detect effects. Zero inflated GLMS may provide a better means to analyse these data and this method will be explored when preparing the manuscript for publication.

There were relatively few *Platycephalus richardsoni* observed, and those that were, were in the southern half of sample locations. This is the only species in the assemblage that has detailed biological information dating from when the population was first fished to the present (Jacobsen 2010). The first commercial trawl fisheries in NSW were found to target this species and despite their population collapsing in the late 1950s (Gowers 2008). It has since recovered and is currently estimated at ~50 percent of its virgin biomass (Stewart et al. 2015), based on when trawling commenced in 1915. This species is thought to move into deeper water as they mature (AFMA 2017) and is generally, but not always, caught by recreational fishers in deeper waters than those I sampled (Authors pers. obs).

Univariate measures of diversity indicated that alpha diversity between zones was equivalent and that communities were similarly even. That is, there were no differences among zones detected in the number of species present (species richness), the effective number of species (Shannon diversity) or how even the communities were (Pielou's evenness measure). These results may appear to contradict the multivariate results in this chapter (that show dissimilarity between all three assemblages). However, unlike the multivariate tests, the diversity measures do not consider community composition and therefore tell us nothing about the similarity or dissimilarity of assemblages, only how diverse they are relative to each other (see supporting information in Chapter 2 for a worked example). Therefore, when considering these diversity measures together with multivariate comparisons, it suggests that while each zone has similar number of species

present and maintains equivalent levels of diversity, this diversity is represented by different species in each zone.

The deeper coastal waters (>50 m) sampled in this chapter are particularly challenging to sample due to the distance from ports, depth that equipment needs to be retrieved from and the variable weather events. The current sampling effort was all that was logistically feasible for this study and although extensive, the results suggest that increasing the number of locations sampled would be beneficial and provide better univariate estimates of abundance and size. Greater sampling at the location level would increase the statistical power of univariate assessments and provide better estimates of patchily distributed taxa.

In this study, I investigated the patterns in fish assemblages over three management levels, but made no assessment of the drivers of these patterns. However, the results suggest a number of avenues of future research that would aid in understanding my results here. Very few of the soft sediment associated species in the assemblage have movement data (Chapter 1). As excessive movementor "spill over" between zones will likely negate the effect of protection, movement information could explain some of the patterns revealed here (Grüss et al. 2011). The lack of differences between management zones for some species in my study, may be a result of a high level of mobility. Alternatively, even where fish movement is shown to be restricted, if populations within NTZs are still being fished then no differences (or a smaller magnitude difference) by management zone would be expected (Advani et al. 2015). In the same manner, if fishing pressure outside the NTZs is within the bounds of natural mortality of these species then no differences or very little difference among zones would be expected. I had limited estimates of fishing pressure and had no estimates of compliance with zone regulations. However, there is evidence that non-compliance may be an issue, for example there have been confirmed instances of illegal trawling occurring in BMP (see ABC News 2013) and at various times there have been reports of spikes in the number of fishers caught in no-take zones (e.g. 32 people found fishing in BMP no-takes zones on one 2013 long weekend; Fishing World 2013). Research into the movement of fish in the assemblage and into the level of fishing effort (including compliance) would greatly aid in determining drivers of abundances and patterns in the composition of the fish assemblage.

In conclusion, coastal marine soft sediments are heavily exploited and are the most common habitat in New South Wales MPAs and many other locations, however, the impacts of protection on demersal fish found on this habitat are rarely studied. This study provides some of the first estimates on abundances and lengths of fish on soft sediments in relation to MPAs and does so across a gradient of fishing pressure. The data in this study provide baseline fishery independent data that will be particularly important to assessments of protection impacts in these MPAs in the future. Overall, I showed that after ~8.5 in BMP and ~13 years of protection in JBMP, there was a clear difference at the assemblage level among all three management zones. At the species level, there were more Ocean jackets (*Nelusetta ayraudi)* in NTZs and for the most common species in the assemblage, *Platycephalus grandispinis*, greater size of fish in NTZ was also observed. There were, however, no effects of zoning on any of the other common species in the assemblage. Future sampling should focus on providing more precise estimates of their abundances within zones (i.e. by sampling more locations within more replicate zones along the NSW) to better assess these patterns at both the assemblage and individual species level and assess whether they are consistent over time.

Chapter 4 Designing and Testing an Acoustic Telemetry Array on Unvegetated Soft Sediments

Plate 4.1: Top: Preparing acoustic tracking equipment before deployment. Bottom: A V9 acoustic tag used in detection range testing (Photo: Paul Jones).

4.1 Introduction

Acoustic telemetry has become a widely used method for studying the movement patterns of marine animals (Hussey et al. 2015). The technology revolutionised the study of fish movement by allowing tagged individuals to be tracked for long periods with relatively high spatio-temporal resolution (DeCelles and Zemeckis 2014). Unlike traditional mark-recapture techniques where only release and re-capture locations can be obtained, acoustic tags provide data across multiple timepoints and fish do not need to be re-caught to collect the data (Dudgeon et al. 2015). Acoustic tags also have advantages over archival (dataloggers) and satellite tags, being much smaller in size and not needing to be above the surface to send a signal or relay data. Acoustic telemetry also generally provides much better resolution than archival tags and the development of acoustic positioning systems provides the ability to collect fine <5 m scale movement data (Espinoza et al. 2011), rivalling that obtained by satellite tags.

Passive acoustic telemetry involves the use of static underwater receivers that detect and then record transmissions from nearby acoustically tagged animals, with each transmitter or 'tag' sending a unique ID code at programmed time intervals. The use of multiple passive receivers in a tracking array has the advantage of allowing many animals with acoustic tags attached to be tracked continuously for extended periods (Clements et al. 2005, Hussey et al. 2015). Tagged animals can also be tracked over a large spatial area, limited only by the number of receivers that can be deployed (DeCelles and Zemeckis 2014). More recent developments can also allow these tagged animals to be detected and then positioned with a high level of accuracy in acoustic monitoring systems (Espinoza et al. 2011). Prior knowledge of the study species likely movements is important so that receivers can be placed to maximise the likelihood of detecting tagged animals or key stages of their movement using the available resources (Heupel et al. 2006). Designing an effective passive tracking array is also greatly aided by information on the ability of receivers at the study site to detect tagged animals (Kessel et al. 2014, Stocks et al. 2014).

Acoustic telemetry, like all methods, has several constraints and it is important to understand how these influence array design and the subsequent results (Heupel et al. 2006, Kessel et al. 2014). One of the main limitations is the detection range of receivers. Detection range defined by defined by Kessel et al. (2014) as "the relationship between

detection probability and the distance between the receiver and tag", is influenced by numerous factors including tag type and signal strength, sound attenuation over distance, environmental noise (wind, waves, currents etc), biological noise (noise from animals such as snapping shrimp) and topography either directly blocking or interfering with transmissions (Heupel et al. 2006, Huveneers et al. 2016). There are many other complex factors that will reduce detection rates or interfere with tag detections, including false detections caused by collision of tag transmissions (Simpfendorfer et al. 2015) and in some cases close proximity transmission echoes (Kessel et al. 2015). Combined, these environmental, biological and technical factors result in variable detection rates of transmissions both over space and time.

Only a small proportion of published passive acoustic telemetry studies effectively account for detection range, particularly spatial and temporal variation in detection range. Kessel et al. (2014) reviewed 378 passive acoustic tracking studies in detail and found that the vast majority failed to assess and monitor detection range adequately. As a result of inadequate monitoring of detection range, the behavioural inferences made in many passive acoustic studies may not be reliable (Kessel et al. 2014). Kessel et al. (2014) provide a series of recommendations on detailed range testing to assess detection ranges and give minimum requirements that should be met to ensure an understanding of detection range variation (and therefore subsequent tracking results). The recommendations cover both "in-situ" range testing using permanent sentinel range takes within an array that assess range variation at the same time as tracking of tagged fish and "prior" range testing used in the design of tracking arrays.

Detection range testing at a study site prior to array deployment provides an estimate of detection range and detection probability on which to base receiver spacing and array design (Heupel et al. 2006, Kessel et al. 2014). Acoustic detection range tests can be done in a number of different ways, including passive tests with receivers deployed in a variety of configurations and numerous types of mobile range tests (e.g. Clements et al. 2005). An effective and common method of passive detection range testing involves placement of various combinations of receivers and tags at set distance intervals (e.g. Stocks et al. 2014, Dance et al. 2016, Huveneers et al. 2016, Kessel et al. 2016, Selby et

al. 2016). The data can then be used to create distance detection profiles that allow an estimation of detection probability at a given distance from a receiver.

The final spacing and configuration of a passive array will depend on the study aims, resources available and limits imposed by detection range (Heupel et al. 2006, Stocks et al. 2014). If data are only required on how often tagged animals visit a small restricted location or series of locations then single or a small number of receivers placed at specific locations of interest may be sufficient. In such a case, detection range testing can provide guidance on the number of receivers required to provide effective coverage of each specific point. For example, if a reef outcrop was of interest then range testing can provide an estimate of whether the entire outcrop will be within the detection range of a single receiver or if additional receivers are required to provide effective coverage. Where data are required on whether tagged animals leave an area, such as a bay or estuary, then a line or a series of lines of receivers may suffice (Heupel et al. 2006). Detection range testing can minimise detection gaps in the line and estimate likelihood of tags passing through the receiver line without being detected (Stocks et al. 2014). If the objective is to maintain a relatively continuous monitoring of a tagged animal's presence (or absence) in an area then an overlapping grid array using multiple receivers may be most effective (Heupel et al. 2006). If overlapping detection range is required then range testing allows an estimate of maximum distance receivers can be spaced apart while still maintaining overlapping detection range (Heupel et al. 2006).

In Jervis Bay, Australia a passive tracking array on marine soft sediments was proposed in 2011. The main objective of the planned array was to measure movements and residency of demersal fishes on soft sediments over multiple years. Before setting up the passive tracking array, information specific to the study site was required. Specifically, estimates of 1) likely movement by soft sediment associated fish, and 2) receiver detection range and spatiotemporal detection range variability on soft sediments. This information would allow the most effective balancing of limited resources (in this case 16 receivers initially) and the two conflicting requirements of the array; maximising the area covered whilst ensuring sufficient detection probability over that area in order to be able to confirm residency. A preliminary study using active acoustic tracking on tagged bluespotted flathead, a common demersal fish species found on soft sediments in the

study area, provided the initial movement data (Fetterplace et al. 2016). These data provided guidance on strategic positioning of receivers to best align with likely fish movement and the ideal area of coverage needed to obtain useful information on movement. However, information on receiver detection range and detection variability was still needed to inform decisions on receiver spacing and array configuration.

The specific aims of this chapter were to a) estimate the probability of detecting tag transmissions by distance from receiver on soft sediments in Jervis Bay (Chapter 4.1), b) use this information on detection probability in conjunction with likely fish movement (Fetterplace et al. 2016) to design at large passive tracking array on soft sediments (Chapter 4.2), and c) test the arrays performance once it was in place (Chapter 4.2). To ensure that an adequate understanding of detection range was achieved, I used the recommendations in the comprehensive review by Kessel et al (2014) as a guide in all of the detection range testing undertaken in this thesis.

4.2 Static acoustic range testing on unvegetated soft sediments in Jervis Bay: Exploring variation in detection probability.

To determine the detection range of Vemco VR2W acoustic receivers on soft sediments in Jervis Bay, Australia, I carried out a long-term static range test over seventy days. The study area in the north east of Jervis Bay lacks any obvious impediments to acoustic transmissions and is protected from weather and swell. Prior to this study there had been no passive acoustic range testing carried out on soft sediment habitats in Jervis Bay. However, tracking on soft sediments adjacent to reef with portable active tracking units (VR100s) suggested a potential maximum detection range of ~300 m (Ferguson et al. 2013). I also sought to test whether there were differences in detection success between day and night, and between high and low power transmissions.

Range Test Methods

Hare Bay lies in the north east of Jervis Bay (Fig. 4.1) and is dominated by unvegetated soft sediments with *Posidonia australis* and *Zostera* spp. seagrasses in the shallow waters of the Bay. The substratum from 6–15 m depth within Hare Bay is relatively homogenous, predominately comprised of unvegetated bioturbated sand. The seafloor topography is flat and slopes gradually towards the south west. There is very

little hard structure beside the macrobenthos (e.g. polychaete tube worms; 30–40cm small branching structures) to directly obstruct acoustic signals (see Fig. 4.3 for example photographs of the substratum). In addition, Hare Bay is relatively protected from swell, and most wind directions. As a no-take sanctuary zone on the opposite side of the Bay to the major populated centres it receives little boat traffic.
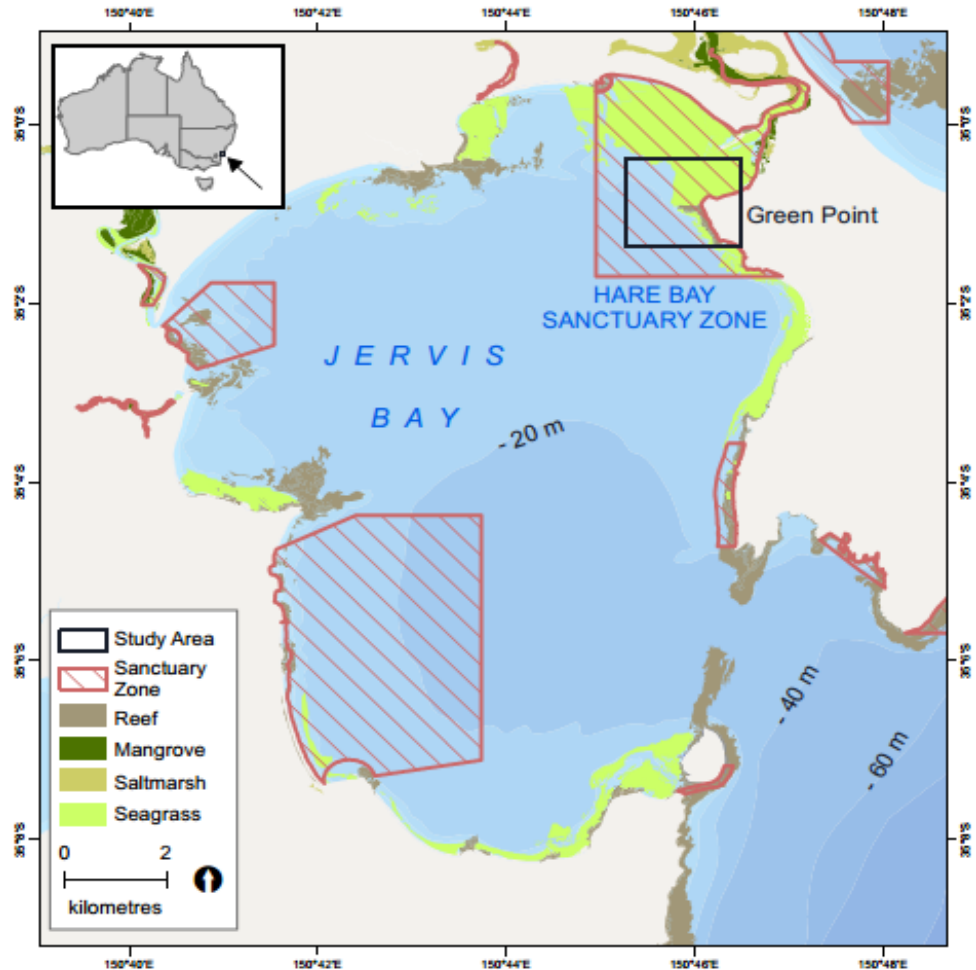
Figure 4.1: Study location in Jervis Bay, NSW, Australia. Area where the range test was carried out in Hare Bay no-take sanctuary zone is shown within the black square. The areas shaded blue are marine sand. This map is from Fetterplace (2016) used under a CC BY 4.0 licence (http://creativecommons.org/licenses/by/4.0/).
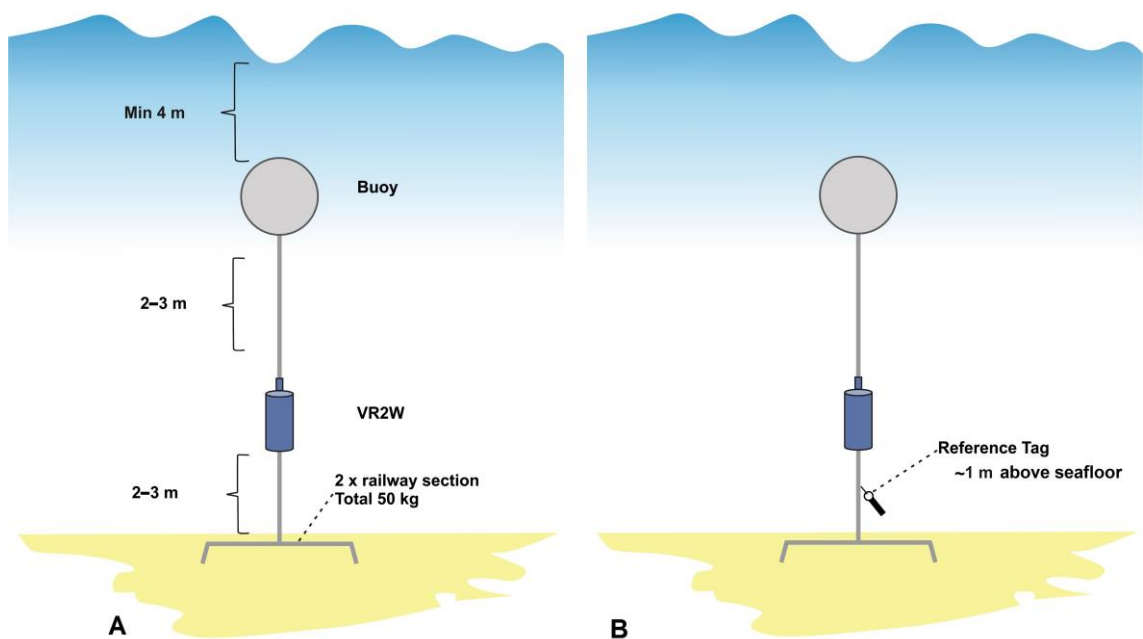
Figure 4.2: Mooring configuration used in acoustic monitoring in Hare Bay. a) Standard mooring setup with VR2W receiver attached, b) mooring with "demersal" reference tag attached (Figure from Fetterplace 2018, CC-BY).

Moorings and Receivers

All moorings used in range testing and fish tracking in Hare Bay were based on the same design and consisted of a single rope attached to a 50-kg section of railway line (Fig. 4.2a). Both the buoy and anchor were attached using an eye splice. Receivers were then attached to the rope 2–3 m above the substrate facing upwards (Fig. 4.2), using four cable ties (Fig. 4.3). Permanent array moorings have a subsurface buoy to reduce the risk of gear loss through theft or boat entanglement and receivers were deployed and collected using SCUBA divers. For this range test where moorings were temporary, surface buoys were used to allow retrieval of receivers by boat. Before deployment the head and base of each receiver was painted with anti-foul and the receiver body covered with thin plastic and silver tape which allowed easy removal of fouling (Fig. 4.3).

106
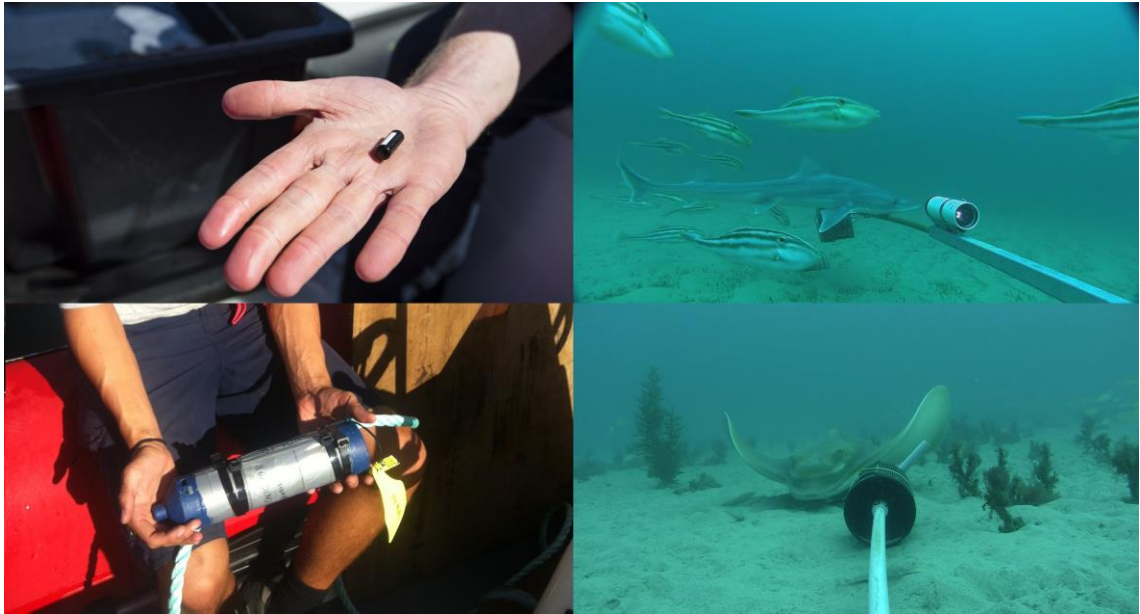
Figure 4.3: Top left: V9 acoustic tag. Bottom left: VR2W acoustic receiver attached to mooring rope. Right panels: two examples of soft sediments habitats within Hare Bay, captured using remote underwater video.

Range test

The static range test was carried out between the 11/3/14 and 19/5/14. The relatively long 70-day period was used to ensure a wide range of weather and other environmental conditions were included in final detection probability estimates. Five VR2W-69KHz receivers were placed on moorings at set distance intervals, with one receiver at each of the following distances; 0 m, 190 m, 200 m, 320 m, 420 m (Fig. 4.4). Water depth varied from ~12.6 m at the mooring at 0 m to ~11.0 m at the mooring at 420 m. A single V9–2x range testing tag (Tag 1) was attached to the first mooring at 0 m (Fig. 4.4). To aid in removal of fouling the tag was firstly placed inside a section of stocking. The tag came with an external cap attached, and two cable ties were passed through the stocking and the cap to attach the tag to the mooring. Tag 1 was programmed to switch between low (145dB at 1 m) and high power (151dB at 1 m) every eight days (Table 4.1) to allow the assessment of the effects of high and low transmission on detection rates. Another V9–2L tag (Tag 2) was attached to the mooring at 420 m (Fig. 4.4). This tag was placed inside a section of stocking and the stocking was sealed at each end using a single cable tie. Cable ties were then passed through the sealing ties to attach the tag to the mooring.
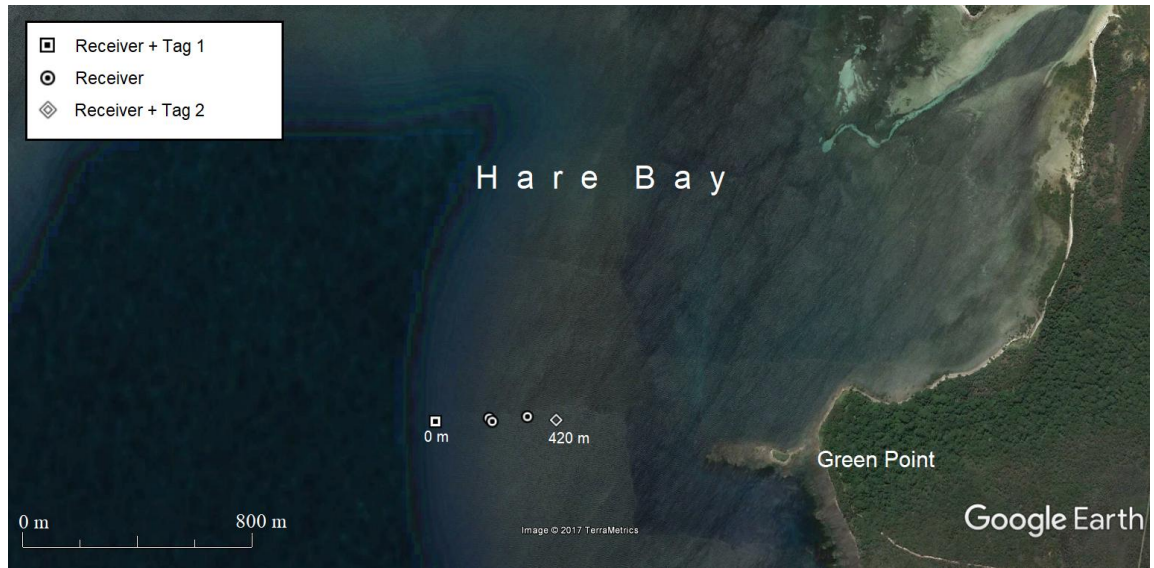
Figure 4.4: Location of static range testing showing acoustic receivers, range testing tag and standard tag in Hare Bay, NSW, Australia. An acoustic receiver was moored at each of five set distance intervals; 0 m, 190 m, 200 m, 320 m and 420 m.

Table 4.1: Range test tag specifications.

Tag Type	Tag ID	Tag Family	Battery Life	Program
Range Testing Tag	A69-1601-25369	V9-2x	82 Days	1) On 4 days; Power H; Fixed Delay: 15 seconds
				2) On 2 Minutes; Power H; Fixed Delay: 5 seconds
				3) On 4 days; Power L; Fixed Delay: 15 seconds
				4) Loop back to step 1
Standard Tag	A69-1303-37315	V9-1L	286 Days	Power L; Fixed Delay: 120 seconds

As the main objective of the planned tracking array in Hare Bay was to track demersal species, both reference tags were attached 1 m above the seafloor to better represent where tagged demersal fish are likely to be located most often (Fig. 4.2b). The staggered distance intervals of moorings combined with attaching a tag at each end of the receiver line, allowed me to test a greater number (i.e. eight distance intervals) of detection distances using five receivers (i.e. distances intervals for Tag 1; 0m, 190m, 200m, 320m, 420m, and for tag 2 in reverse; 0m, 100m, 220m, 230m, 420m. Tag 1 and the receiver on the same mooring (Fig. 4.4; 0m) were retrieved from the water on 1/5/14, however tag 2 and the four other receivers remained in the water for 18 days longer as bad weather cut short the first retrieval attempt. In addition to testing the maximum

detection range, placing a receiver on the same mooring as the tag allowed a test of minimum effective detection range; that is whether there was an area underneath or close to a receiver that would have a lower than expected probability of detecting a tag due to signal shadowing or interference.

Data were downloaded into the Vemco User Environment (VUE), time corrected and false detection analysis carried out (See VUE user manual and Pincock 2012). Detections from the first and last day when each tag was in the water were excluded from results as each tag was in the water only a portion of a full day and handling of receivers, boat noise and tag transmissions throughout the water column on descent all likely influenced detections on those days.

The mean detections per day (±SE) for each tag at each distance interval was calculated and plotted. For each tag and receiver, the detection proportion was calculated for each day of the study by dividing the number of detections by the known number of transmissions (daily transmissions; Tag 1: 4768 and Tag 2: 701). These proportions were plotted by distance interval for each tag and to allow estimates of detection probability the data for each was fitted with a LOESS curve (locally weighted polynomial regression). A complete range test detection probability plot across all eight distance intervals was then created using daily detection proportions from both tags fitted with a LOESS curve. LOESS is a non-parametric method to obtain a smoothed curve that combines multiple weighted least squares regressions to estimate overall trends in the data (Cleveland 1993). LOESS allowed the proportion of transmissions detected to be predicted at unsampled distances with a better fit than linear models. The proportion of values included in each local regression is determined by a user specified 'span' value (Zuur et al. 2009), in this case a span of 0.8 was used for all LOESS curves as this value gave the best fit in all cases. LOESS curves were produced using the 'ggplot2' package (Wickham 2009) in R (R Development Core Team 2014).

For tag 1 differences in the total proportion of transmission detected when using high power and using low power were compared using a chi square goodness of fit test and binomial confidence intervals were calculated (McDonald 2014). The proportion of high and low power transmissions detected were also compared at each distance interval

to examine whether there was a difference in detection probability. So that complete 8-day periods of each power setting were included, only data from the power switch on the 15 March to power switch on the 16 April (36 days) were included in this comparison. A Bonferroni correction was applied to account for the multiple (6) tests and a significance level of 0.0083 assigned.

To further explore variation in detection efficiency in detail across days and between night and day, the mean proportion of detected tag transmissions by 12-hour intervals were plotted (Day 0700 to 19:00 and night 19:00 to 0:700 UTC +10) for each tag across the entire range test. Night time was defined to account for sunrise and sunset times (Sunrise UTC +10, 06:50 to 07:42 hrs and sunset 19:20 to 18:00 hrs respectively) over the study. For both tags differences in total proportion of transmission detected between day and night periods across the study were compared using a chi square goodness of fit test and binomial confidence intervals calculated (McDonald 2014). A Bonferroni correction was applied to account for the multiple (2) tests and a significance level of 0.025 assigned. Lastly, changes in detection rate within days by hour was examined visually by plotting total proportion of hourly transmissions detected at each distance interval.

Static Range Test Results and Discussion

Understanding the range at which a receiver can detect an acoustic tags transmission and how likely it is to do so are important components of designing an acoustic array (Singh et al. 2009). In the present study, I undertook range testing prior to deployment of a passive acoustic tracking array to track demersal fishes on soft sediments in Jervis Bay, Australia. This was important as few estimates exist over these kinds of habitats and furthermore it has been suggested that in situ site specific range testing is essential to understand detection probability (Huveneers et al. 2016). The primary aim of the range test was to provide an overall estimate of detection probability by distance from an acoustic tag. These data would then be used to inform decision making on receiver spacing in the passive array.

A total of 642339 detections were recorded on the five receivers over the range test. False detections made up only 0.004% of total detections and none of these were range test tag IDs (see S1 for full details). The main findings are that, overall, detection probability was between ~50% and ~75% up to 250 m from the acoustic tag (Fig. 4.5). The probability was relatively stable for the first 100 m from the tag with >75% of transmissions successfully detected, before a gradual decrease in detection probability occurred over the remaining 320 m in the range test (Fig. 4.5). The distance at which 50% of transmissions were detected was ~250 m (Fig. 4.5) and detection probability by 420 m from the tag was <5% (Fig. 4.5). The results suggest that the maximum detection distance for V9s on soft sediments in Hare Bay is not far beyond 420 m in the day and ~420 at night. It is likely that a small proportion of detections would be made beyond this distance, particularly during daytime, however as we did not have receivers further than 420 m from the tag I do not attempt to interpolate beyond this distance. I was more concerned about the range of effective detection distance in this test, especially given that the maximum detection distance could be confirmed once the larger array was in place.
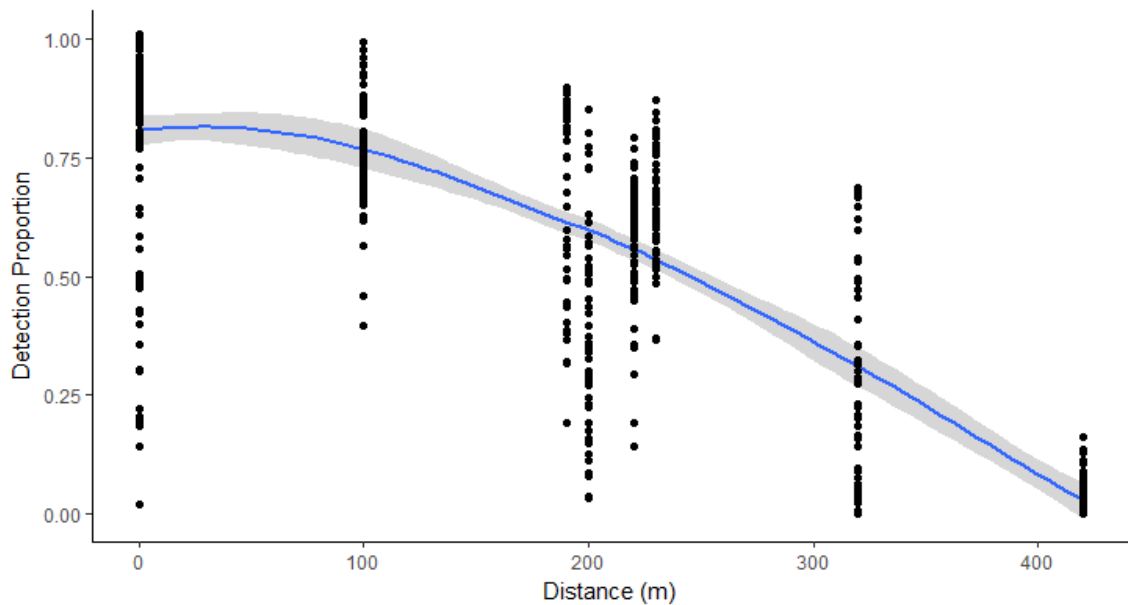
Figure 4. 5: Detection probability by distance based on data from both tags in the range test. Data are daily proportion of tag transmissions successfully detected by fixed acoustic receivers at eight distance intervals (Tag 1; 0, 190, 200, 320, 420, and tag 2; 0, 100, 220, 230, 420). A LOESS curve (Local Polynomial Regression, ± 95 CI) of detection probability by distance is fitted to the data.

The range test also provided additional information on likely variation in detection probability at each distance interval between days and within days. Although there was variability in detection rates between days, from 0 m to 230 m both tags in the range test were detected on every day of the range test at the relevant distance intervals (Fig. 4.6), and always more than 100 times in a day. This outcome suggests that a tagged fish remaining within 230 m of a receiver in the study area would be detected even on days where detection probability was relatively low. While at 320 m from the tag there was a much lower proportion of transmissions detected there were still only two days on which the tag was detected fewer than 100 times (6 and 33 times respectively).

The proportion of detections successfully detected was lower for range tag 1 then for range tag 2 (Fig. 4.6). At 0 m from the tag there was a ~20% difference in detection probability between tags at ~70% for tag 1 (Fig. 4.6a) and ~90% for tag 2 (Fig. 4.6b). Detection probability of 50% of transmissions for tag 1 was at ~ 220 m (Fig. 4.6a), compared to at ~280 m for tag 2 (Fig. 4.6b), however by 420 m both were detecting very few transmissions on all days so detection probability was similar (Fig. 4.6). The proportion of daily detections was also much more variable at each distance for tag 1 (Fig. 4.6a) compared to tag 2, which generally had a smaller range of daily proportions with few outliers (Fig. 4.6b). As a result, most of the variability in detection probability in the combined dataset (Fig. 4.5) was caused by fluctuations in detection rate of tag 1. Many studies use multiple range tags and combine the data sets before analysis to account for differences among tags (e.g. Stocks et al. 2014, Huveneers et al. 2016) with the expectation that there may be some variation between tags, caused by, for example, difference in signal strength of tags with the same model specifications (Heupel et al. 2006)
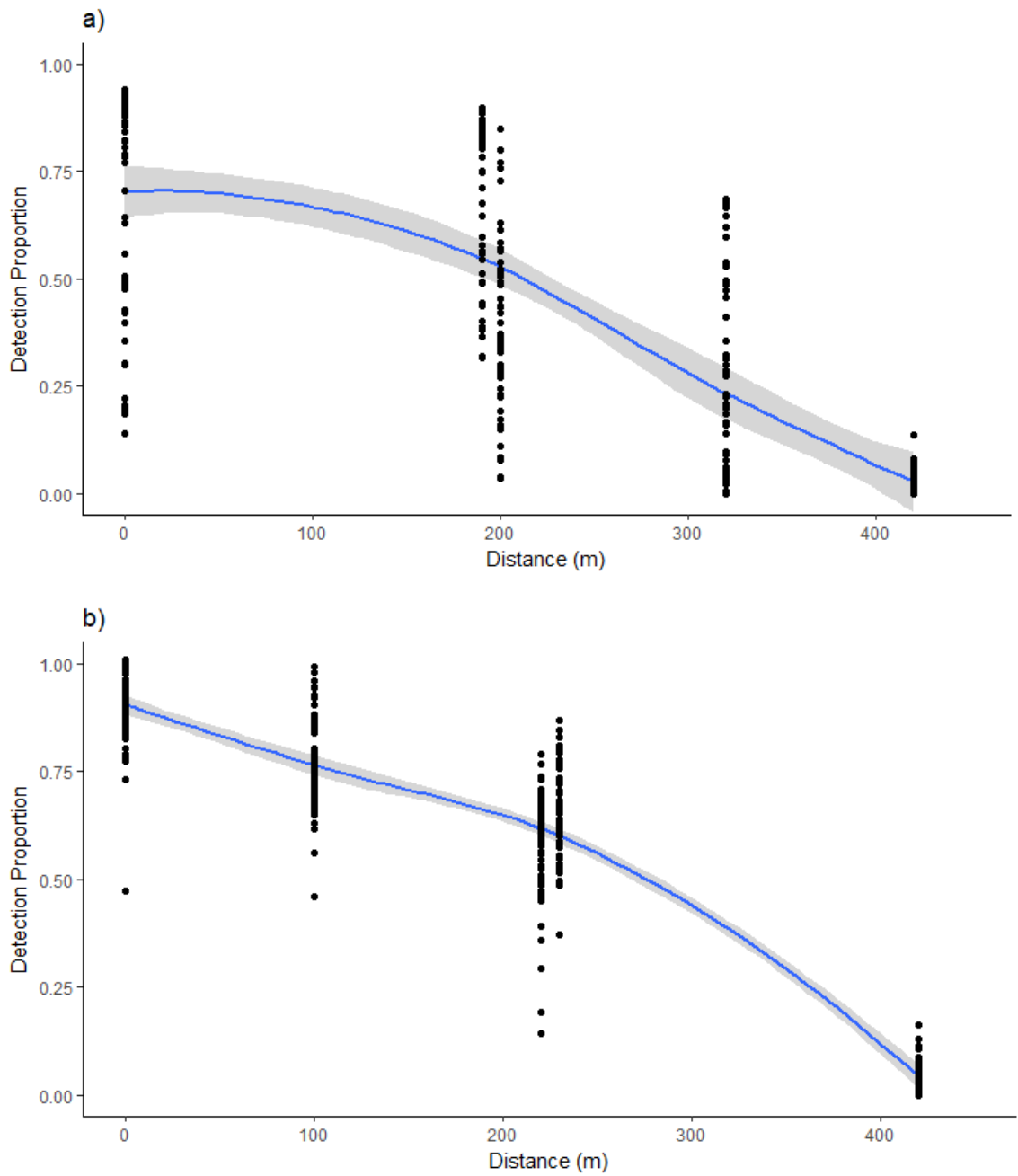
Figure 4.6: Detection range profiles for acoustic receivers and tags over five distance intervals in range tests in Hare Bay. Data are daily proportion of tag transmissions successfully detected by fixed acoustic receivers at five distance intervals for a) tag 1 (V9-2x, 12 March–30 April, 2014) and b) tag 2(V9-1x, 12 March–18 May, 2014). A LOESS curve (Local Polynomial Regression, ± 95 CI) of detection probability by distance is fitted to both data sets.

Lower detection success and greater variability for tag 1 was caused in part by a noticeable and prolonged reduction in detectability on the 16th of April at all distance intervals. Lower detection proportions then continued over the last 16 days that tag 1 was in the water (Fig. 4.7). This did not occur for tag 2 across those dates (Fig. 4.8), suggesting that tag 1's drop in detections was not related to any environmental changes or fouling as such an effect would be expected to be observed for both tags. The reduction in detections over the last part of the study for tag 1 seems consistent with a technical issue such as battery power loss or tag malfunction, although environmental or biological reasons cannot be categorically ruled out. When the data for the last 17 days are removed for tag 1 (Fig. 4.9) the detection probability aligned more closely with that for tag 2 (Fig. 4.6b). Nevertheless, variation in detection success between days at each distance for tag 1 remained high, even with these data removed. For example, at each distance interval, tag 1 commonly had a >50% difference in the proportion on transmissions detected between adjacent days (Fig. 4.7). The cause of the high variability in detection proportion between days for tag 1 over the whole range test may again be related to tag function. As I was not able to confirm the reason for the reduction in detections I included the full data set in the results (other than Fig. 4.9). This likely resulted in a conservative estimate of detection probability and has the advantage of including the potential for unexpected variability in detection success in decision making on future array design and receiver spacing.

The proportion of tag 2 transmissions detected were much more stable and generally followed the expectation of lower detections the further from the tag (Fig. 4.8). The only notable exception for tag 2, was a sharp drop in the number of detections by the receiver at 220 m from the 4th of May down to ~5% detections at night time on the 5th of May before gradually increasing again to the end of the study. The drop appeared to be independent of the other receivers and although the proportion of transmissions climbed again, the receiver at 220 m always detected a lower proportion of transmissions than the receiver at 230 m after this date (Fig. 4.8) and as a result had lower average detections per day (Fig. 4.10a). The same receiver also on average recorded much fewer detections of tag 1 transmissions compared to the receiver 10 m closer to the tag (Fig. 4.10b). Together these anomalies suggest that the receiver was performing sub-optimally (either

the signal was blocked by fouling, mooring line etc or a technical issue) or there was another reason, such as fine scale habitat differences I am unaware of, influencing detection success.

Minimum Detection Range

Assessing minimum effective detection range is an often over looked component of acoustic telemetry and is arguably as important as assessing maximum effective detection range (Kessel et al. 2015). Reduced detections underneath or close to a receiver can be caused by transmissions being shaded by the receiver body or moorings or by close proximity detection interference (See Kessel et al. 2015 for detailed description). If a minimum detection range is present and not accounted for, the implications can impact substantially on results, for example if fish are aggregating around moorings they may not be detected and considered to be absent from the study site. In this study, I placed a receiver and tag on the same mooring at 0 m to allow an estimate of how reliably a tag will be detected if it is very close to or under a receiver. In this range test, both tags had the highest proportion of detections achieved at 0 (Fig. 4.6) and therefore a minimum detection range was not apparent for either tag. Kessel et al. (2015) reported that close proximity detection interference was greater for higher power tags such as V16s.

Therefore, if higher power tags are deployed in Hare Bay in the future the minimum detection range should be re-evaluated for those tags. Conversely, in some cases transmission echoes (a form of close proximity detection interference) can result in detections being higher than transmissions (Kessel et al. 2015). There was no apparent 'positive' echo effect in the range test for tag 1 (Fig. 4.7). However, for a small number of days tag 2 detections were greater than 100% of transmissions at 0 and 100m (Fig. 4.8). Given that the number of days with incidence of echoes was very few and the magnitude tiny (Fig. 4.8), I considered the impact of the echoes in this range test to be negligible.
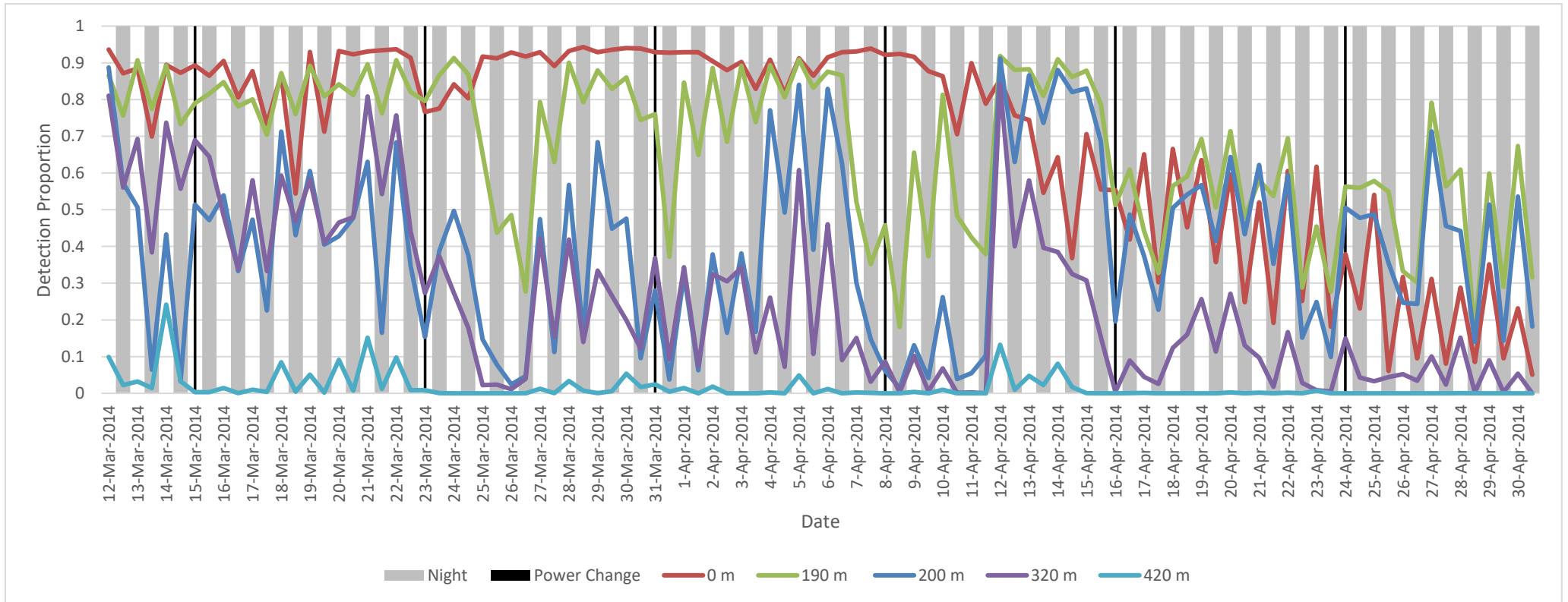
Figure 4.7: Detection proportion profiles at five distance intervals for acoustic range test tag 1 (V9-2x), by day (0700–1900) and night (1900–0700) across 50 days (12 March–30 April, 2014). Range tag 1 sent 2384 transmissions/12 hrs and was programmed to switch between high power (151 decibels at 1 m) and low power (146 decibels at 1 m). Black line indicates power change over point, with the first 4 days being low power and power changes every 8 days after that.
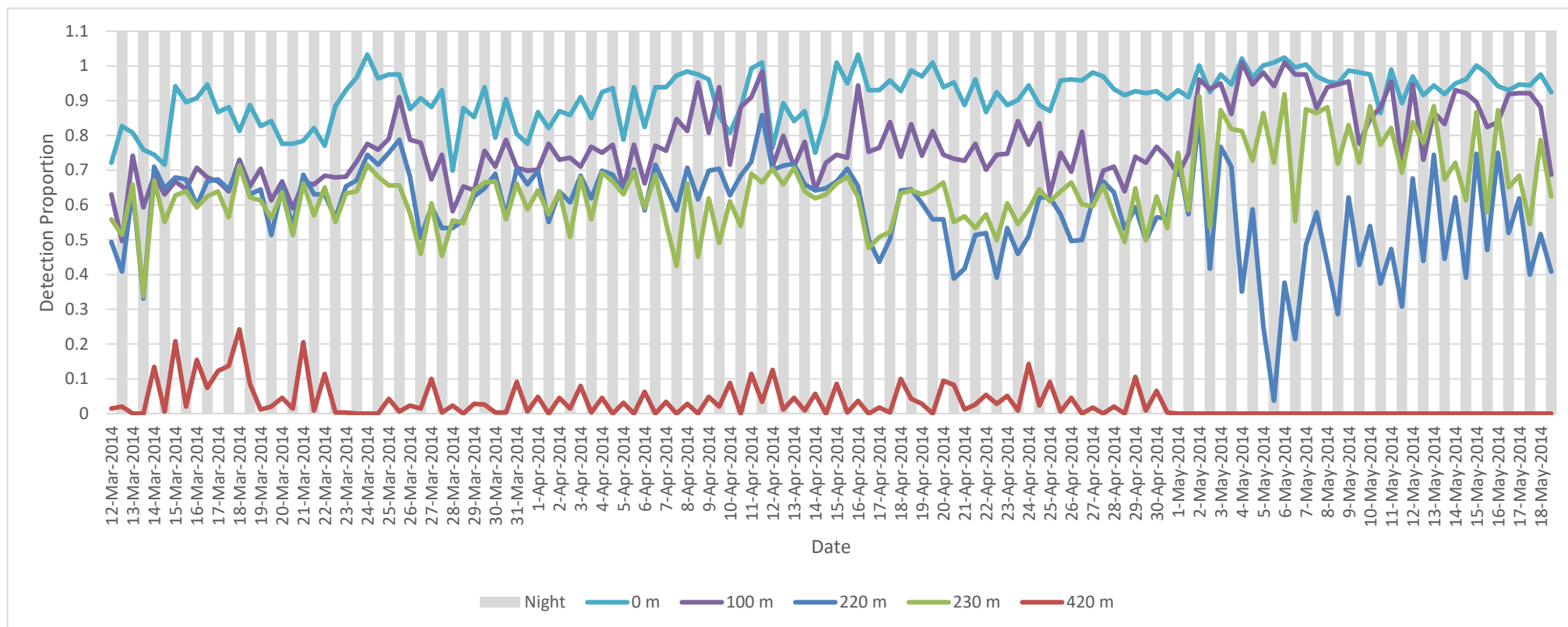
Figure 4.8: Detection proportion profiles at five distance intervals for acoustic range test tag 1 (V9-2x), by day (0700–1900) and night (1900–0700) across 50 days (12 March–30 April, 2014). Range tag 1 sent 2384 transmissions/12 hrs and was programmed to switch between high power (151 decibels at 1 m) and low power (146 decibels at 1 m). Black line indicates power change over point, with the first 4 days being low power and power changes every 8 days after that.

117

Other detection considerations

When transmitting in high power, range tag 1 had a significantly greater total proportion of transmissions detected compared to when in low power (Table 4.2). The difference was also apparent at all distance intervals (Table 4.2). However, the effect size of 5–7 % difference in high and low detection proportions at 0 m, 190 m, 200 m, 420 m distance intervals and for total detections was relatively small (Table. 4.2). At 320 m the difference of 16% was higher (Table 4.2). Given the substantial reduction in battery life associated with power differences (e.g. a V9 tag with 110–250s delay; high power = 487 days battery life or low power = 666 days battery life) the small magnitude of difference was somewhat surprising.

There appeared to be a relatively consistent decrease in proportion of detections made each night compared to day for both tag 1 (Fig. 4.7) and tag 2 (Fig. 4.8) and overall, there was 3–12% difference in detection rate between night and day (Table 4.3). Although the difference in proportion of total transmissions detected between day and night was significant both for tag 1 ($\chi^2 = 10938$, df = 1, p < 0.0001) and tag 2 ($\chi^2 = 88.166$, df =1, p < 0.0001), the difference was greater for tag 1 (Table 4.3). Visual inspection of patterns in mean proportion of hourly transmissions revealed that differences in detection rate between day and night were relatively consistent by distance interval for tag 1 (Fig. 4.11). However, for tag 2 there was a greater change in proportion of transmissions detected between night and day the further from the tag (Fig. 4.12). Detections increased after 05:00 and began decreasing again after 17:00. The lowest detection rates appear to be between 18:00 and 20:00 for both tags (Fig. 4.11, 4.12), the exception was at 0 and 100 m for tag 2 where the mean proportion of transmissions from tag 2 detected was stable across day and night (Fig. 4.12).

When comparing tagged animal movements between night time and day time, differences in detection probability should be accounted for and data standardised using reference tags (Payne et al. 2010). All things being equal the number of day time and night time detections would be expected to be the same. However, variability in detection success between day and night has been reported in numerous studies previously (e.g. How and de Lestang 2012, Villegas-Ríos et al. 2013) and increased biological noise at night time has been suggested as a possible explanation where reduced night time

detections are found (Payne et al. 2010). Although determining the reason for the reduction in detections at night was beyond the scope of this range test it would an interesting area of further investigation.
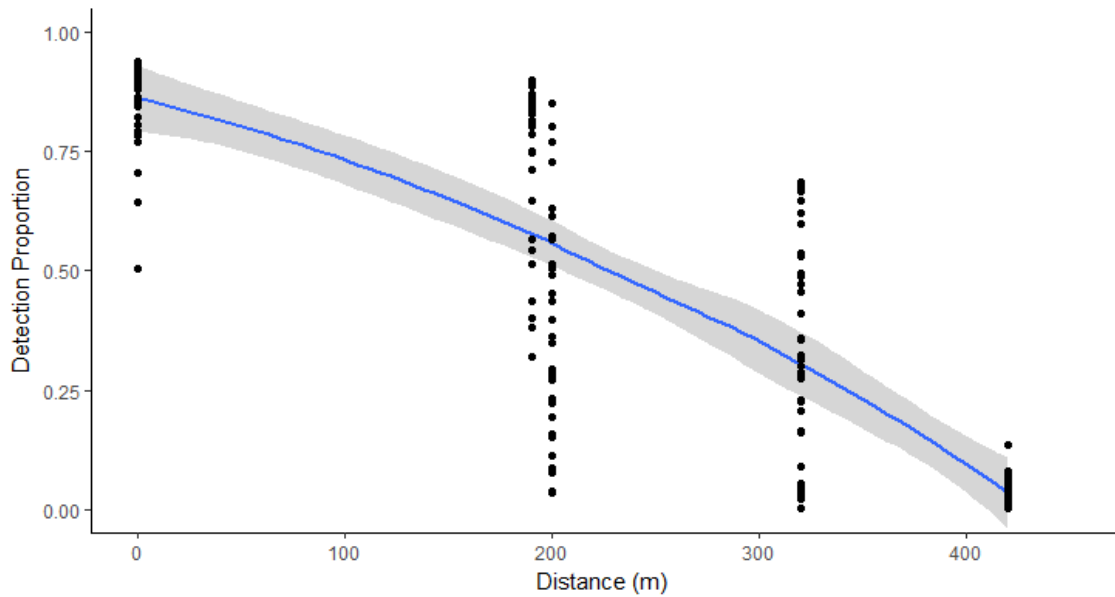
Figure 4.9: Detection range profile for range tag 1 over five distance intervals in range tests in Hare Bay with the last 17 days of data removed. Data are daily proportion of tag transmissions successfully detected by fixed acoustic receivers at five distance intervals for a) Tag 1 (V9-2x, 12 March–14 April, 2014). A LOESS curve (Local Polynomial Regression, ± 95 CI) of detection probability by distance is fitted to the data.
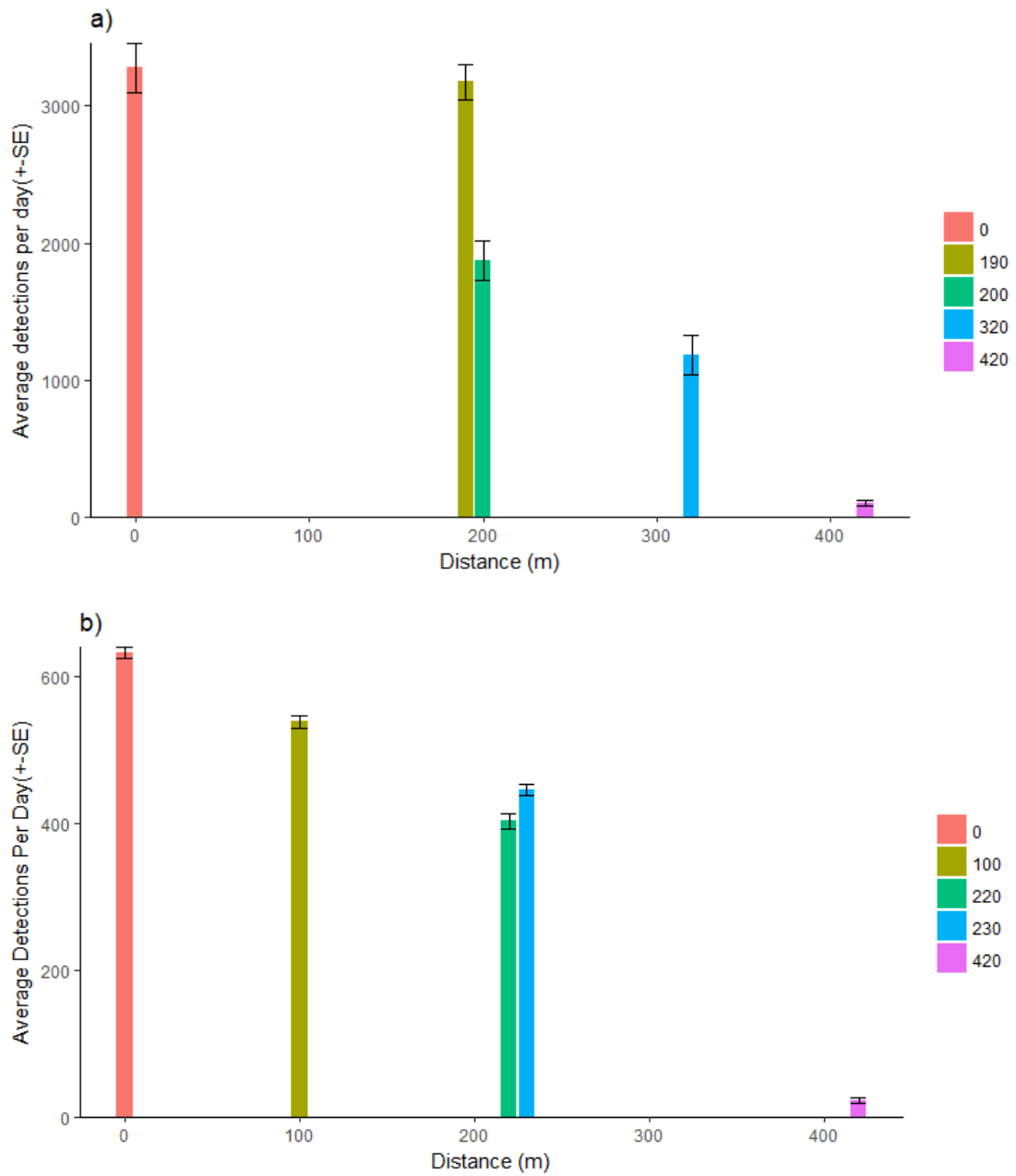
Figure 4.10: Average (±SE) daily range tag transmissions successfully detected by fixed acoustic receivers at five distance intervals for a) tag 1 (V9-2x, 12 March–30 April, 2014) and b) tag 2; (V9-1x, 12 March–18 May, 2014). Range tag 1 transmissions per day = 4768, and range Tag 2 transmissions per day = 701.

Table 4.2: Proportion of range tag 1 high and of low power transmissions detected by distance and in total, with 95% binomial confidence intervals (CI). Chi square goodness of fit significance level is 0.0083 and all tests had 1 degree of freedom.

Distance		0 m	190 m	200 m	320 m	420 m	Total
Power	Low	0.768	0.660	0.347	0.196	0.014	0.397
	High	0.827	0.730	0.399	0.357	0.019	0.467
95% CI	L Lower	0.765	0.656	0.344	0.193	0.013	0.395
	L Upper	0.77	0.663	0.35	0.199	0.015	0.398
	H Lower	0.825	0.727	0.396	0.354	0.019	0.465
	H Upper	0.83	0.733	0.403	0.36	0.02	0.468
	χ^2	181.67	288.05	299.71	3790	73.315	2285.6
	p-value	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001	< 0.0001

Table 4.3: Overall proportion of transmissions detected at night and day for each range tag, with 95% binomial confidence intervals (CI). Chi square goodness of fit significance level is 0.025 and all tests had 1 degree of freedom.

	Day	Night	CI Day	CI Night	χ^2	p-value
Tag 1	0.470	0.347	0.469–0.471	0.346–0.349	10938	< 0.0001
Tag 2	0.599	0.570	0.597–0.602	0.567–0.573	88.166	< 0.0001
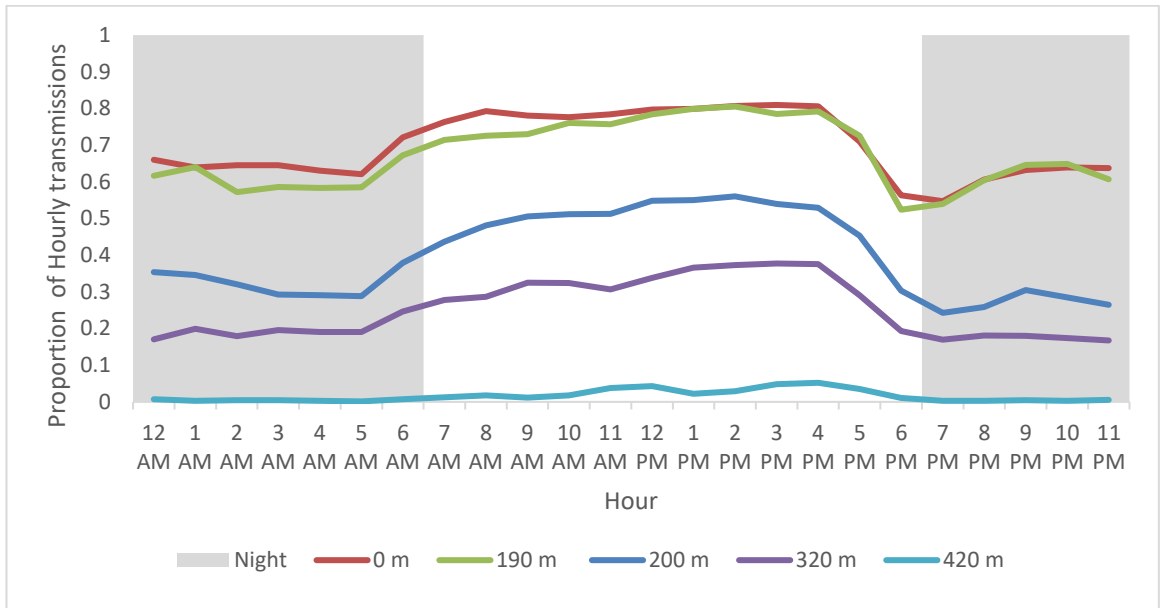
Figure 4.11: Tag 1 V9-x; Proportion of total hourly transmissions detected at each distance interval. Shaded areas represent night time from 19:00 to 07:00 hrs.
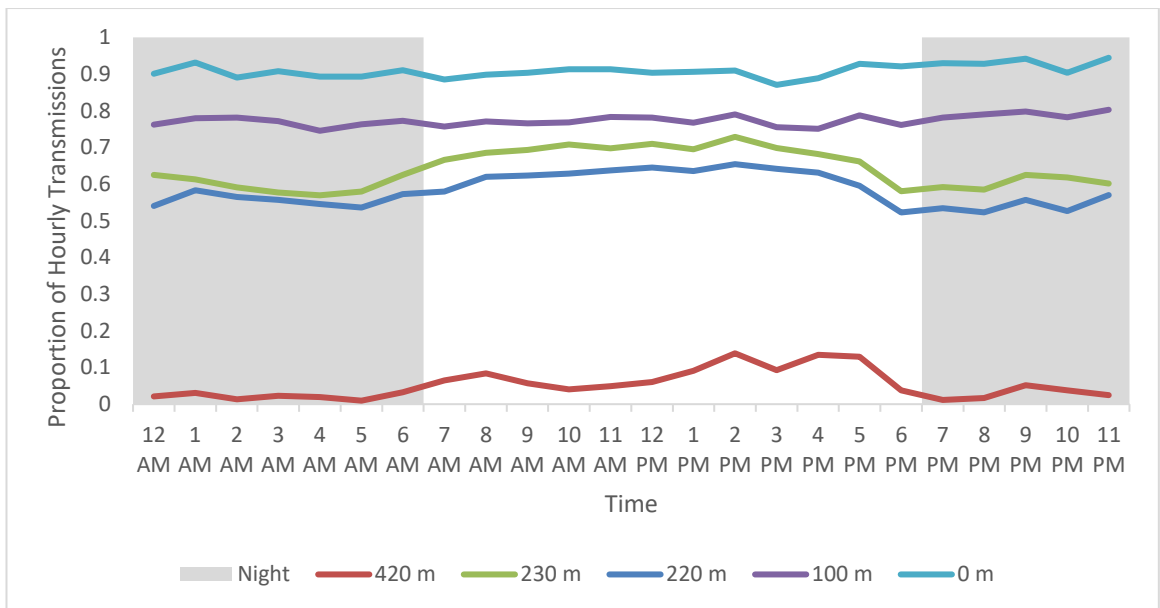
Figure 4.12: Tag 2 V9-L; Proportion of total hourly transmissions detected at each distance interval. Shaded areas represent night time from 19:00 to 07:00 hrs.

4.3 Hare Bay Array Design and Testing

The main requirement of a passive tracking array in Hare Bay was that it maintain a high probability of detecting tagged demersal fish over a large area, whilst still allowing sufficient detections to provide high confidence in estimates of presence/absence of tags. Taking into account this requirement, preliminary active tracking results (Fetterplace et al. 2016) and results from static range testing (Chapter 4.1), I deployed 16 omni-directional (Vemco VR2Ws) acoustic receivers in an isometric grid with spacing of ~ 300 m between receivers (Fig. 4.13). Receivers were attached 2–3 m above the substratum to fixed moorings with a single float, and weighted with 50 kg railway line (Fig. 4.14). Two additional moorings, with no receivers, were placed in the array so that reference sentinel tags could be attached during future fish tagging studies (Fig. 4.13).

Chapter 4.1 static range testing results suggested that the receiver spacing could have been greater than 300 m and still have achieved good presence-absence detections over a larger area than the array that was put in place. For example, a 500 m receiver spacing, based on range test detection probability of ~50% at 250 m, would have likely provided acceptable detection rates. However, pulling in the receiver spacing to 300 m had three advantages 1) greatly increased detection probability over the whole array, 2) as the range was based on a conservative estimate, provide a buffer should unexpected fluctuations in receiver detection range occur and 3) it created multiple overlapping detection ranges (Fig. 4.13) which meant that the array could be VPS enabled (Vemco positioning system referred to hereafter as a "VPS" and the array as Hare Bay VPS), a useful addition that would give fine scale positions of tagged fish over a large area (Fig. 4.13).

Upgrading the array to a VPS provides the ability to collect fine <5 m scale position data. When a tag signal is detected by three or more receivers in the VPS, a position can be calculated using a time difference of arrival algorithm (Espinoza et al. 2011; Wolfe and Lowe 2015). A horizontal position error is estimated using synchronising tag data and environmental conditions (Bergé et al. 2012, Roy et al. 2014). Fixed synchronising acoustic tags are deployed within the VPS so that receiver

clocks can be synchronised (See Espinoza et al. 2011 for in-depth VPS analysis description). The Hare Bay VPS allowed fine-scale positioning of tagged fish over the core area of the array covering ~78 ha, in which the maximum distance to three receivers was 300 m (Fig. 4.13). (Here: water temperature: 13.7–23.8∘C; salinity 35.4–35.8 ppt). In the Hare Bay VPS, a Vemco V16 synchronising tag (V16-6x L, 69 kHz, 540–720 delay) was attached 0.5–1 m above each receiver (Fig. 4.14).

Once the Hare Bay VPS was in place, I carried out additional range tests within the VPS to test the arrays performance and assess whether my decisions on receiver spacing were suitable. To test the likelihood of detecting and then positioning a tagged fish moving through the array, mobile range testing was undertaken. The mobile range test was complimented by two short term stationary range tests. These results would allow me to decide if adjustments to the array, to fine tune its performance, were required before tagging of fish could begin.
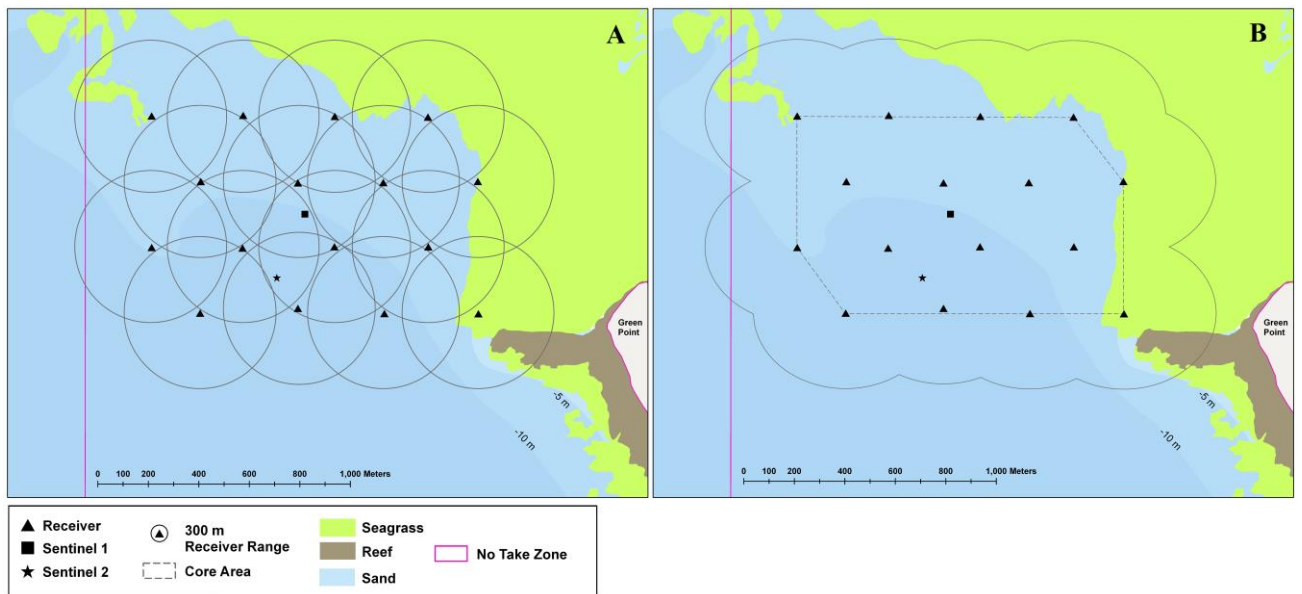
Figure 4.13: Hare Bay VPS acoustic receiver array. Receivers (triangles) were deployed with ~300 m spacing on the 1/9/14 and each receivers range and range overlap is indicated here by the grey circles in A. The area within 300 m range of at least on receiver was ~202 ha and the core area of the array covered ~78 ha.

124
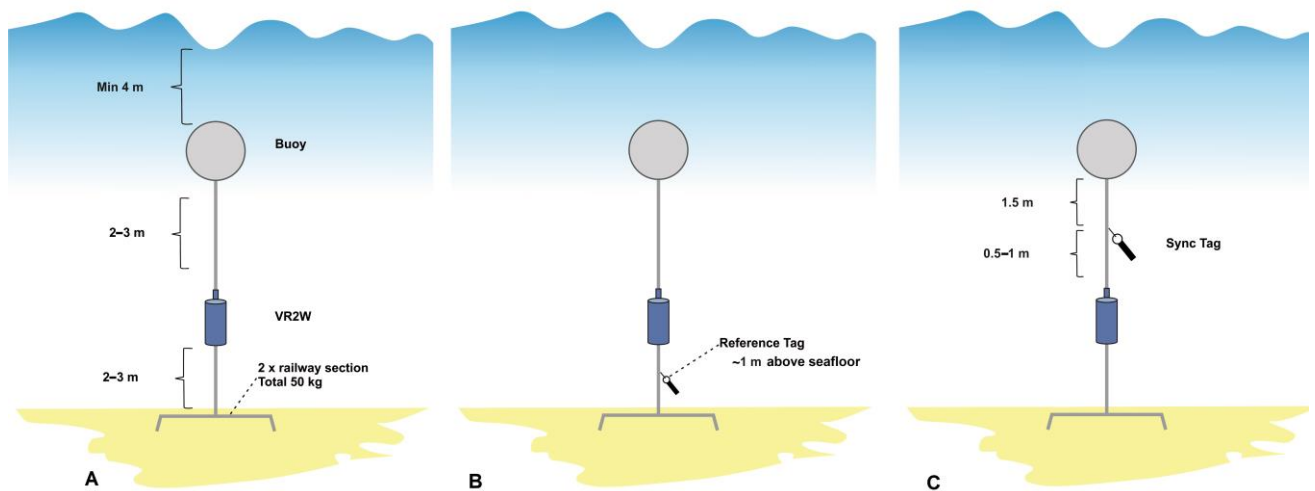
Figure 4.14: Mooring configuration used in acoustic monitoring in Hare Bay. A) standard mooring setup with VR2W receiver attached, B) mooring with "demersal" reference sentinel tag attached, and C) mooring with synchronisation tag attached. Initial deployment of moorings with attached receiver was carried out by boat and any subsequent change-over of receivers (both retrieval and deployment) was carried out by SCUBA divers.

Methods and Results: VPS Performance Testing

Mobile Range Testing

      Following the placement of the Hare Bay VPS, mobile detection range testing to estimate detection and positioning success of a moving tag by multiple receivers simultaneously was carried out. A coded V9-2H (110–250s delay) tag, the same model used to tag fish in the main study, was used for all tests in this section. For mobile tests, the tag was taped to a weighted line attached to a boat. The boat was then allowed to drift through the array with the prevailing wind dragging the tag to simulate the movement of a fish through the array. The tag was then positioned using the VPS and the positions compared to the path of the boat drift recorded using GPS. There was probably a slight difference in the tag location (or position) relative to the GPS on the boat at the surface of the water. This was not corrected and assumed to be relatively small (e.g. <3m). Eight drifts in total were completed on two days. A detailed overview of drifts, number of detections recorded and positions calculated follows;

Day 1: Thursday 11/9/2014 **-** Westerly wind blowing drifting boat to the east. A tag on the line was deployed to 3 m depth.

Drift 1: 12:16pm (Australian EST) the tag was deployed on the southern border of the VPS receivers. The boat and the tag began to drift to the east. Four VPS positions successfully calculated for this drift until 1:01pm (Fig. 4.15) and in this time 33 detections were recorded. The tag was lifted from the water and the boat was driven to the south-western side of the VPS within the array and the tag redeployed into the water.

Drift 2: At 1:14pm the first position of this drift was calculated. In total, 47 detections were recorded and 8 VPS positions calculated across this drift, the last one at 1:38pm (Fig. 4.15). The tag was then lifted from the water and the boat was driven back to the western side of the array and slightly to the north of Drift 2.

Drift 3: At 1:54pm the first position of this drift was calculated. Twenty-two detections were recorded and 3 positions calculated in total across this drift, the last one at 2:14pm (Fig. 4.15). The tag was then lifted from the water and the boat was driven back to the western side of the array and slightly to the north of Drift 3.

Drift 4: At 2:22pm the first position of this drift was calculated. Thirty-six detections were recorded and 8 positions were calculated in total across this drift, the last one at 2:48pm (Fig. 4.15). The tag was then lifted from the water and the boat was driven to the central area of the array.

Day 2: Tuesday 16/9/2014

Drift 5: The tag was deployed within the central north section of the array. The wind was blowing from the north, so the boat was drifting to the south. The first position was calculated at 10:18am and there were 5 positions calculated across this drift (Fig. 4.16). The last position was calculated at 10:35am. There were 24 detections in total in this time. The tag was lifted from the water and the boat was driven to the northern side of the VPS again just outside of the array and the tag redeployed into the water.

Drift 6: At 10:48am, the first position of this drift was calculated. Three positions were calculated in total across this drift, the last one at 11:24pm (Fig. 4.16). Twenty-six detections were recorded in this time. The tag was then lifted from the water and the boat was driven back to the north-western side of the array.

Drift 7: At 11:37am, the first position of this drift was calculated. Six positions were calculated in total across this drift, the last one at 12:07pm (Fig. 4.16). In this time 27 detections were recorded. The tag was then lifted from the water and the boat was driven to the north-east corner of the array.

Drift 8: At 12:35am, the first position of this drift was calculated. Six positions were calculated in total across this drift, the last one at 12:52pm (Fig. 4.16). In this time 26 detections were recorded. The tag was then lifted from the water and removed from the array.

All VPS positions from mobile testing were within 10 m of the GPS track (and most <5 m).

Short-term Stationary Range Tests.

Two short term stationary tests with the tag were also carried out, one overnight for ~17hrs and the other during the day for ~3hrs. A coded V9-2H (110–250s delay) tag was used in these tests. GPS positions were taken straight above the mooring line. A detailed overview of drifts, stationary deployments and positions calculated is follows;

Stationary deployment 1: At this central position in the array (-35.011025, 150.757112) the tag was deployed close to the seafloor (within 0.5 m from the bottom) on a weighted line with a float of the surface. On 11/9/2014 at 4:44pm the first position of the stationary deployment was calculated. The tag was left in place until 12/9/2014 at 10:23 am. Over 200 positions were calculated during this stationary deployment (4:44pm 11/9/2016 to 10:23 am 12/9/2014). The tag was then lifted from the water after the overnight stationary deployment 1 and the boat was driven back to the south-western corner of the array.

Stationary deployment 2: At this south-western corner of the array (-35.014989° 150.754984°) the tag was deployed as in stationary deployment 1. At 10:42am the first position of the stationary deployment was calculated. The tag was left in place until 12/9/2014 2:37 pm. >40 positions were calculated in during this stationary deployment (10:42am to 2:37 pm 12/9/2014). The tag was then lifted from the water and removed from the array.

All VPS positions from stationary testing were within 5 m of the GPS position (and most <1 m).
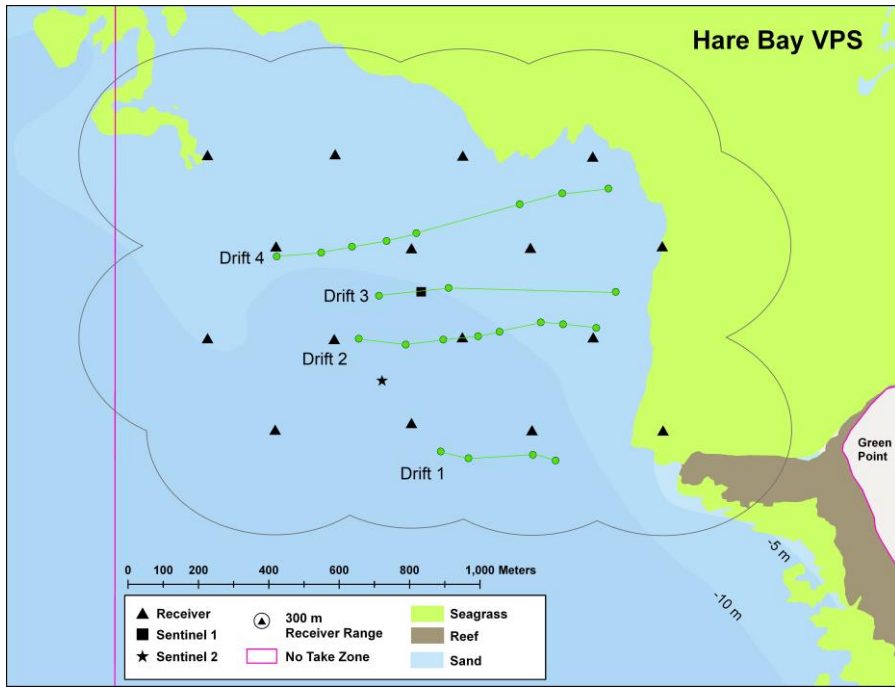
Figure 4.15: Day one mobile range testing (11/9/2014), drift one to four. Green circles indicate successful positioning by Hare Bay VPS.
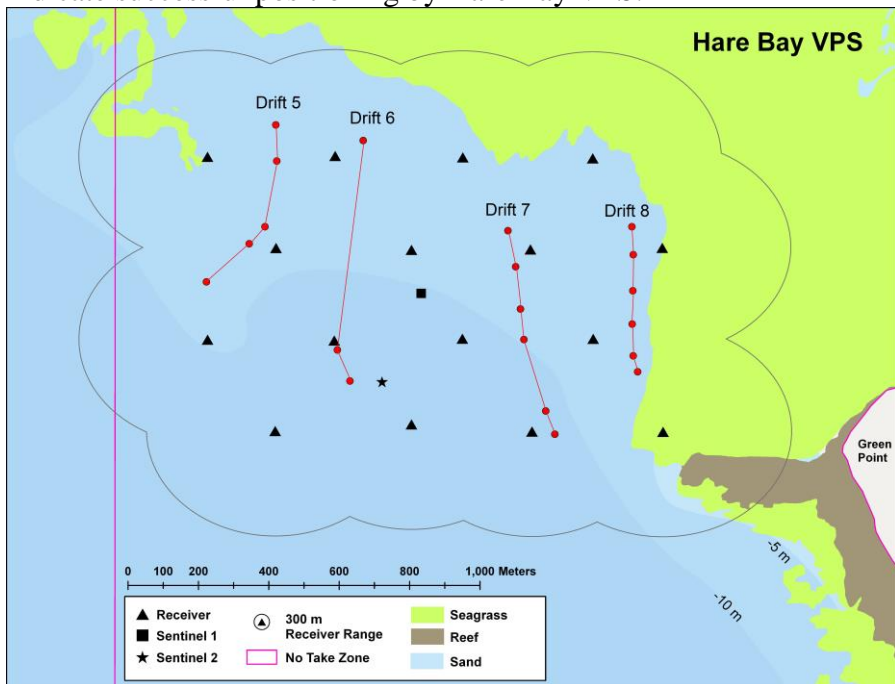
Figure 4.16: Day two mobile range testing (16/9/2014), drift five to eight. Red circles indicate successful positioning by Hare Bay VPS.

Discussion: VPS Performance Testing

Mobile range tests, to simulate a fish moving through the Hare Bay VPS suggest that a tagged fish would almost certainly be detected in the array if present. In addition, the fish would also likely be positioned (requiring a signal to be detected by

at least three receivers) relatively frequently. Detections by 1–2 receivers on drifts through the array were frequent and consistent and 3–8 positions were successfully calculated on each drift through the VPS. Stationary tags were also detected frequently and a large number of positions calculated during the time the tag was in the water. The stationary test results suggest that a fish within the VPS, that remains motionless, or moves over very small scales for extended periods, will also be detected and positioned frequently. VPS positions from stationary testing were within 5 m of the GPS track (and most <1 m) compared to <10 m for mobile test, which is likely to be in part because I could ensure the GPS unit was right above the tag mooring in static tests (rather than trailing a few metres behind the boat on a rope).

There were some gaps in positions along drifts during mobile range tests. This does not impact on presence/absence results as detection rates were so high, however, if finer details are required in future tracking studies within the Hare Bay VPS, then these occasional holes in positioning ability need to be considered. Inserting additional receivers into the VPS to reduce distance between three receivers is an option to increase positioning success. Alternatively, another option is to narrow down the area a tag transmission originated from by calculating centre of activities (COA) using a detection on two receivers (Simpfendorfer et al. 2002), and using these two COAs to fill holes in positions along a track.

Unlike in the long-term static range tests in Chapter 4.1, I did not use fixed delay tags in these mobile and short-term stationary tests. The rationale behind this was that using tags with the same delay as tags to be used in tagged fish would give a better estimate of the likelihood of a fish being detected moving through the delay. The drawback with this approach was that the exact number of transmitted signals could not be known (Only a range of values, with a known min and max number of transmissions) and compared against number of detections. Using fixed delay tags and random delay tags in unison could potentially improve future detection range tests of this nature, although the benefits need to be weighed against the increased costs.

4.4 Discussion: Designing and Testing an Acoustic Telemetry Array

Range testing prior to passive acoustic tracking array deployment provides information on which to base array design, and subsequent array testing provides validation of array performance (Kessel et al. 2014). In this study, long-term stationary detection range testing in Chapter 4.1 provided estimates on which to make decisions on receiver spacing, and range tests in Chapter 4.2 demonstrated that the VPS array put in place functions as required. These combined 'pre' tagging range tests were important in designing the array and suggest that, under the conditions encountered, a tagged fish would almost certainly be detected in the array if it was present. The results also provide data with which to make broad estimates of future detection range. However, range testing prior to tagging fish should not replace in-situ range testing undertaken alongside tracking of tagged fish (Kessel et al. 2014, Huveneers et al. 2016).

As detection range is dynamic and fluctuates through time, detection range under actual study conditions needs to be accounted for in order to accurately interpret tracking results (Payne et al. 2013, Kessel et al. 2014). The most effective way to monitor and account for detection range fluctuations at the time tracking is underway, is to have fixed sentinel tags in place at the same time (Kessel et al. 2014). Without sentinel tags in place, it is not possible to know whether the detections obtained reflect the presence-absence patterns of tagged fish or are an artefact of receiver performance at that time (Payne et al. 2010, Kessel et al. 2014). Thus, sentinel range tags should ideally accompany all future tagging studies in the VPS. I have added permanent moorings within the VPS for this purpose, and carried out in-situ range testing using sentinel tags in Chapter 5 in unison with the first tagging study under taken within the array.

An understanding of detection range, probability and variability at a study site is required to fully interpret acoustic tracking data (Cagua et al. 2013, Huveneers et al. 2016). Nevertheless, the comprehensive range testing undertaken in this thesis has rarely been carried out in passive acoustic tagging studies. Based on the in-depth criteria in Kessel et al. (2014), the range testing here (including use of sentinel tags in

Chapter 5) scores ~40/45, a score higher then all 378 passive acoustic telemetry studies assessed ($\bar{x}$ score = 11.1 ± 0.4SE). According to the review's criteria, modelling the impact of biotic and abiotic variables on detection range and proportion of transmissions detected would have increased the score to 45/45.

Numerous variables may drive fluctuations in detection range. In previous studies, windspeed, wind direction, precipitation, atmospheric pressure, water temperature, thermocline, salinity, background noise, turbidity, ground swell size, tides, current speed, moon phase and water column stratification are some examples of variables that have been shown to, or shown not to reduce detection range in various cases (Gjelland and Hedger 2013, Kessel et al. 2014, Huveneers et al. 2016). Monitoring environmental variables can be important for two reasons; a) it can be useful if you need to later model their impact on detection ranges e.g. if in a subsequent tracking study, you do not know the actual detection range and want to attempt to correct for fluctuations in detection ranges because of wind (if you found earlier that wind has an impact). However, that is the fall-back method and has limitations (Huveneers et al. 2016). Ideally, you will know the actual detection range and probability at the time, based on a continuous assessment using sentinel tags "because this technique inherently monitors variability in the detection range as a function of all anthropogenic and natural parameters, this provides the most comprehensive technique for assessing acoustic range and should be adopted whenever possible" - Kessel et al. (2014), b) you want to understand how these environmental variables influence fish behaviour. If so, monitoring these parameters before tagging is underway is not ideal. In this case, it is better to monitor these variables during the course of the tracking study, correct tracking data for range fluctuations based on sentinel tag data and then compare fish movement to environmental fluctuations to understand how tagged fish respond to them.

Understanding the causes of detection range fluctuation would not have improved the VPS array design. There are a number of reasons why I did not assess how environmental factors impact on detection range in range testing prior to tagging of fish. Firstly, and most importantly, given the aims of my study, having the information does not add anything to the decisions on receiver spacing and array

131

design. For example, if I measured wind speed and modelling had shown that wind was a major reason behind some decreases in detection range in Hare Bay (e.g. using methods such as those outlined in Gjelland and Hedger 2013), how would that knowledge further inform the array design? It would not change the design, nor spacing of receivers in any way, because I already know and it is more relevant to know, what the minimum, maximum and average detection range is. It does not matter in this case why the minimum or maximum detection range is what it is, only that I know what it is and account for it. It is of course interesting to know why receiver range fluctuates at various times. However, given the numerous potential factors that could cause a reduction in receiver performance and the cost associated with assessing them properly (not to mention the complexity of deciding which factors to measure and why), assessing them here was not warranted. I instead focused on meeting the key aims of the chapter, by directing limited resources to extra range test tags and extending the range test study length.

4.5 **Supporting Information**

False IDs

False detections are a result of transmissions from multiple tags colliding and causing receivers to detect a new incorrect tag ID (Simpfendorfer et al. 2015). These false detections come in two forms. The first is detection of a tag that wasn't part of the study and cannot have been in the study area. Although it should be noted that there is always the chance that these 'false' detections could be from other researchers tagged fish that have moved into the area unexpectedly. However, for the purposes of the original study it makes little difference whether they are false detections or other researchers tags as these data are generally excluded from analyses. The second more problematic type of false detections results in a tag ID code that is the same as a tag ID code in use in the study and is therefore more difficult to identify and more problematic if included in analyses. False detections in the static range testing section of this study (Chapter 4.1) that were later identified as coming from real tags included; one tag identified on IMOS animal tracking database as a V16 in the NSW DPI coastal sharks project (details embargoed) and 14 unknown tags. All 14 unknown tags identified in false detection analysis accounted for 27 detections in total. The only false detection identified that aligned with a known tag ID was one detection of the V16 shark tag.

Chapter 5 Movement Patterns of Soft Sediment Associated Bluespotted Flathead Reveal Long-term Site Attachment in a Marine Protected Area

Plate 5.1: Top: Preparing to launch UoW research vessel Maara. Bottom: Surgery to insert an acoustic tracking tag into a bluespotted flathead under general anesthesia (Photos: Paul Jones).

5.1 Introduction

Globally, marine protected areas (MPAs) are rapidly increasing in both spatial coverage and numbers being implemented (Worm 2017). A large proportion of the total area under protection globally comprises marine soft sediments, and many individual MPAs are dominated by soft sediment habitat (Caveen et al. 2012). However, the benefits of protection for demersal fishes on these habits remain poorly understood and largely unassessed (Lester et al. 2009, Caveen et al. 2012, Sciberras et al. 2013). A key component often missing from our understanding of protection for demersal soft sediment fishes, is information on their patterns of movement, particularly how that movement relates to MPA size and configuration.

Scale of movement has a large influence on the effectiveness of MPAs (Gerber et al. 2003, Grüss et al. 2011). Generally, species that show strong site attachment are considered to benefit the most from protection (Kramer and Chapman 1999, Barnett et al. 2012), as all or a considerable portion of their life cycle will be encompassed within reserve boundaries (Kramer and Chapman 1999, Gaines et al. 2010). Those species with movements over large areas or frequent movements out of a reserve are expected to be less likely to gain from protection as their movements make them more susceptible to capture (Kramer and Chapman 1999). Although, it has been shown that in some cases MPAs can be effective for highly mobile species (see Game et al. 2009, Breen et al. 2015 for disscussion), particularly if reserves are protecting key life history stages such as spawning aggregations (Grüss et al. 2011) or important areas of habitat like foraging grounds (Barnett et al. 2012). If movement outside reserve boundaries is high, because MPAs are too small or do not cover important lifecycle stages, then protection can be diminished or ineffective. Consequently, movement data to understand reserve function and to inform reserve size and placement is of critical importance.

The home range patterns of fish are generally expected to be influenced by food and shelter from predation. It is expected that fish in areas with little available shelter or widely dispersed food sources are expected to have larger home ranges than those with shelter and concentrated food availability (Grüss et al. 2011). On some habitats,

such as on rocky reef and in estuaries, we have a good understanding of fish movement, and site attachment has been demonstrated numerous times for reef associated demersal species (See Kramer and Chapman 1999 and references therein). Recently site attachment has been demonstrated within MPAs directly for an increasing number of demersal reef associated species (e.g. Lee et al. 2015, Matley et al. 2015, Ferguson et al. 2016). In contrast, demersal fish movement on marine soft sediments is poorly understood in general and very little research has been collected *in-situ* within MPAs (Chapter 1, case study 1). This lack of information for demersal fishes on marine soft sediments has meant that MPAs have been designed with no informed estimate of what size and locations are likely to be adequate for these species. As marine soft sediments are relatively homogeneous with little obvious habitat differentiation, current theories (e.g. Grüss et al. 2011) would suggest that there would be no obvious reason why soft sediment fishes might show site attachment for extended periods (Caveen et al. 2012). Hence, these fish would be unlikely to remain within reserve boundaries for long and therefore be unlikely to be affected MPAs (unless the MPA was enormous).

The little available data on the movement of fishes over marine soft sediments generally supports the idea of wide-ranging movement by fishes on marine soft sediments. For example, white croakers show nomadic movement (Wolfe and Lowe 2015), summer flounder move to and from estuaries and inner shelf areas (Sackett et al. 2007) and plaice make long seasonal migrations between spawning and feeding grounds (Hunter et al. 2003). Consequently, the data suggests that soft sediment areas in MPAs will be of little value. However, the number of studies assessing movement of fishes over soft sediments in total is small (Chapter 1) and more research is needed to assess the apparent general patterns. Particularly as; (1) for most fish species found on soft sediments (beaches and deeper unconsolidated sand habitats) there is no data on movement patterns and their residency behaviour; and (2) there is a subset of the current studies that suggest that movement patterns may not be as consistent as predicted. For example, species such as lemon sole appear to show strong site attachment (Jennings et al. 1993) and the Senegalese sole has even been shown to show substantial intraspecific variation in movement patterns with most fish showing the predicted transient pattern, while a substantial proportion appeared to demonstrate site attachment (Abecasis et al. 2014a).

On the East Coast of Australia, the Government of the state of New South Wales (NSW) has implemented a system of 5 Marine Parks along the state's 1700 km of its mainland coastline (extending 3 nautical miles offshore). These Marine Park cover 345,000 hectares of the state's coastal waters with the major habitat type being soft sediments. In NSW, only very limited data exist on the movement of the major commercially and recreationally fished species that inhabit soft sediments on the open coast. No movement data exist for any of the nine most common species on soft sediments on the open coast (Chapter 3).

The bluespotted flathead (*Platycephalus caeruleopunctatus*) is the most common species of the commercially and recreationally targeted demersal species fish in the assemblage from 0 to ~60 m depth off the South-Eastern coast of Australia (Chapter 2 and 3). Despite its commercial and recreational importance, no data on the movement patterns of this species existed prior to my research. In my initial research assessing the short-term movement patterns of bluespotted flathead using active acoustic tracking, I was surprised to find that these fish exhibited residency to small areas of soft sediments over a period of 60 days (Fetterplace et al. 2016). Furthermore, their movement patterns within this area, which was a no-take sanctuary zone (within the Jervis Bay Marine Park), generally only covered a small proportion of the no-take area. The majority of fish remained in a compact area close to their tagging site and the remaining fish made larger movements and left the study area. The study, although relatively short term (due to the active tracking approach and the battery life of the acoustic tags) and involving only 5 individual fish, provided important preliminary data on an unstudied species. These preliminary data appeared to contradict the general theory that soft sediment fishes were highly mobile and unlikely to spend much time in any one place. Hence, they appeared to be a valuable species to assess the consistency of patterns I observed with a small number of individuals and also to determine whether any residency in this species extends beyond a sixty-day period.

In the current study, I developed (along with Dr Nathan Knott, DPI Fisheries NSW) a passive acoustic tracking system to comprehensively assess and quantify the short- and long-term movement patterns and residency of bluespotted flathead within the Jervis Bay Marine Park (New South Wales, Australia).

5.2 Methods

Study Location

Jervis Bay is located on the South-East coast of Australia (Fig. 5.1) and covers approximately 50 km^2. Much of the area of the marine park is subtidal soft sediments (predominately coarse sand). Jervis Bay Marine Park was considered a suitable location to carry out this research as it appeared representative of the NSW Marine Parks generally and logistically feasible due to its relatively wave-sheltered environment meaning general ease for carrying out acoustic tracking. Nonetheless, it is an oceanic dominated embayment (Marine Parks Marine Parks Authority 2008) with the same species found on soft sediments on the open coast (Chapter 2 and 3).

A mosaic of rocky intertidal, subtidal reefs and seagrass beds are scattered around the edge of the Bay (Fig. 5.1). The majority of Jervis Bay lies within the waters of Jervis Bay Marine Park and a small section in the south of Jervis Bay is covered by the Commonwealth Waters of Booderee National Park. The current zoning within the Bay was implemented in October 2002 (Lynch 2006). There are five designated no-take sanctuary zones within Jervis Bay where all extractive harvesting activities, including all forms of fishing, are prohibited. The remaining area of the Bay covered by Jervis Bay Marine Park has zoning that allows for recreational fishing and very limited forms of commercial fishing (e.g. no trawling but limited bait collection and beach netting). In Booderee Commonwealth waters, a small section covering the south of the Bay, recreational fishing is also permitted, however spearfishing and all commercial fishing is prohibited.

Figure 5.1: Left panel: IMOS receiver lines at Bondi and Narooma in relation to Jervis Bay. Right panel: Map of study site and locations of receivers within Jervis Bay Marine Park

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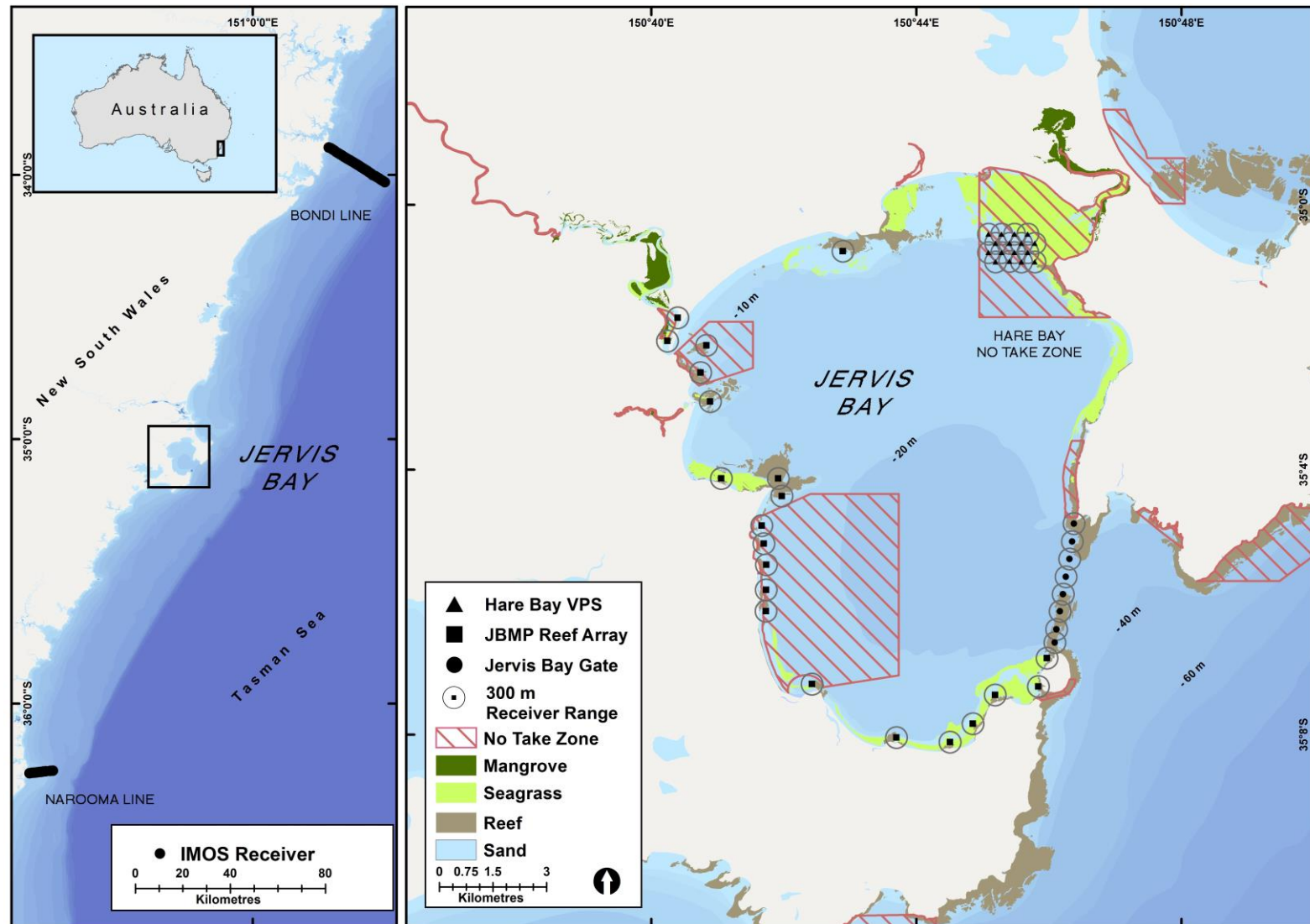
Study Species

The bluespotted flathead (*Platycephalus caeruleopunctatus*, Fig. 5.2) is a demersal species found on coastal marine sands and is recorded as occurring in waters from 5–100 m in south-eastern Australia (Imamura 2015). It is also regularly caught in coastal waters from 0–5 m (Author pers.obs.). Bluespotted flathead are commercially and recreationally exploited (Hall 2015) and are the most common targeted soft sediment associated demersal species in Jervis Bay and surrounding waters (Chapter 2 and 3). They can be distinguished from other flatheads found on soft sediments in the region by markings on their caudal fin (three to six dark bars), the presence of an interopercular flap, and length of the preopercular spine. The preopercular spine being about equal in length or slightly longer than the upper spine (Fig. 5.2). The other common flathead species in the study area, longspine flathead (*Platycephalus grandispinis*) has a lower preopercular spine that is much longer than the upper spine. The full diagnostic features of both species are reported in Imamura (2015).

Movement data on bluespotted flathead is limited to short-term data published from this project (Fetterplace et al. 2016) and a recently published assessment of their residency around a large artificial reef deployed for recreational fishing (Keller et al. 2017). Some life history information is available for bluespotted flathead from the northern half of its range; Based on fisheries research trawl data, adults prefer deeper waters and juveniles under 25 cm in length prefer depths shallower than 30 m (Hall 2015), although in Jervis Bay and South Coast, New South Wales waters, adult flathead are regularly caught in waters under 30 m and I have recorded them consistently on BRUVs in these depths (Chapter 2). Spawning occurs in late winter, spring and summer and bluespotted flathead exhibit sexual dimorphism, with females growing to larger sizes than male; males mature at 1 year of age and 21–23 cm, females mature 2–3 years and 28–35 cm (Barnes et al. 2011, Hall 2015). Barnes et al. (2011) recorded a maximum age for bluespotted flathead of 5 years for females and 9 years for males. Only a very small proportion of the bluespotted flathead population is over 60 cm (Hall 2015). There is no evidence of protandrous sex change in this species (Barnes et al. 2011). For recreational fishers in New South Wales, there is a minimum legal length of 33 cm for bluespotted flathead, under which they cannot be retained.

Figure 5.2: A) Adult bluespotted flathead (*Platycephalus caeruleopunctatus*), B) caudal fin banding patterns on six tagged bluespotted flathead showing slight variation between fish (banding tends to be faded when examining more than 24hrs post mortem), C) Upper preopecular spine (arrowed) equal in length or slightly shorter than lower spine

Passive acoustic receiver array

To record tagged fish within Jervis Bay passive, omni-directional (Vemco VR2Ws) acoustic receivers were used. Receivers were attached 2–3 m above the substratum to fixed moorings with a single float and weighted with 50 kg railway line (see Fig, 4.14 for mooring configurations). Receivers were retrieved and deployed by SCUBA divers. A total of forty-nine receivers were deployed in three arrays within Jervis Bay;

To determine residency of bluespotted flathead to Hare Bay, sixteen receivers were deployed in an isometric grid in Hare Bay no-take zone (Fig. 5.1) on the 1/9/14, in depths of 7–13 m. Receivers were placed with a ~300 m spacing (Fig. 5.3), allowing presence/absence detection of tags to within 300 m of a receiver over an area of ~202 ha. Based on range testing prior to setting up the array (Chapter 4), detection probability over the majority of this area was predicted to be greater than 65%. The exception being the outer perimeter 200–300 m outside the grid, where detection probability would be ~40–65%. To determine actual receiver range under study conditions and monitor receiver range fluctuations over time during the study, reference sentinel tags (Vemco V9-2x L and V9- 2x H, 69 kHz, 500–700 delay) were attached to two additional fixed moorings (one tag per mooring) within the Hare Bay array (see Fig. 5.3 for location within array).

To detect tagged fish moving out of Jervis Bay, eight moorings with receivers were placed across the mouth of the Bay on the 15/9/14 (Fig. 5.1), referred to hereafter as Jervis Bay acoustic gate. Receivers were deployed in a single line between Dart Point on the north side and Bowen Island on the south (mouth width 3670 metres), on sand with a distance between receivers of ~460 m so that detection ranges would overlap. If a tagged fish were to be detected on the gate and not detected later by any other receiver inside the Bay, we considered this fish to have left the Bay at last detection on the gate receivers. If fish were detected on the gate and subsequently detected within the Bay, we would have considered them to have remained in the bay (i.e. turning back from the Bay entrance and continuing to inhabit the Bay). A further twenty-one receivers were placed around the perimeter of Jervis Bay next to most of the fringing reef patches and these receivers provided coverage over reef, seagrass and soft

sediments (Fig. 5.1). These receivers were part of a separate tracking study, but nonetheless provided an indication of whether the bluespotted flathead were using these areas of the Bay. This array is referred to as the JBMP Reef Array.

Finally, as a member of the integrated marine observing system (IMOS) animal tracking facility network, access was granted to data (under a CC BY 4.0 licence) from ~1800 receivers scattered around the coast of Australia (For detailed description of IMOS network see Taylor et al. 2017). If fish were to leave the Bay there was the potential that they could then be detected on these receivers along the East Coast of Australia. These data would be available via the IMOS Animal Tracking Facility database. The arrays closest to Jervis Bay are the IMOS "Bondi Line" ~140 km the north and the IMOS "Narooma" Line ~138 km to the south, both as straight-line distance from the middle of the Jervis Bay acoustic gate (Fig. 5.1).

Fish Tagging

Bluespotted flathead were caught within the core area of the Hare Bay VPS (Fig. 5.3), using rod and line with baited circle hooks or soft plastic lures with barbless hooks. Fish were then placed in a covered holding tank. The fish were anaesthetised in seawater containing 60 mg L^{-1} of Aqui-S before a Vemco acoustic transmitter (for tag details see Table 5.1) was inserted through a ~1 cm mid-ventral incision in the abdomen (Fig. 5.4). The incision was closed with one or two dissolving stitches tied with a double surgeon's knot and surgery lasted ~2 mins. Fish were then transferred to a holding tank and monitored for a minimum of 20 min, before releasing them at the site of capture. A boat-based mobile receiver and hydrophone (Vemco VR100 and VH110 hydrophone) were used to check tag function before fish were released.

Fish Tracking

Bluespotted flathead were passively tracked between 16/09/2014 and 1/06/2016. The fish were caught and tagged in three batches; in spring 2014 (n = 25), autumn 2015 (n = 15) and spring 2015 (n = 6). Although the study ran for 625 days, tagged fish were monitored for a maximum of 618 days, depending on tagging date and tag battery expiry (Table 5.2). Tags types fell into one of two categories based on battery life; accelerometer tags (with 108–155 days

battery life) and coded tags (376–738 days battery life). The shorter battery life accelerometer tags were used to enable activity data to be collected at the same time as movement and residency data, for use in another study.
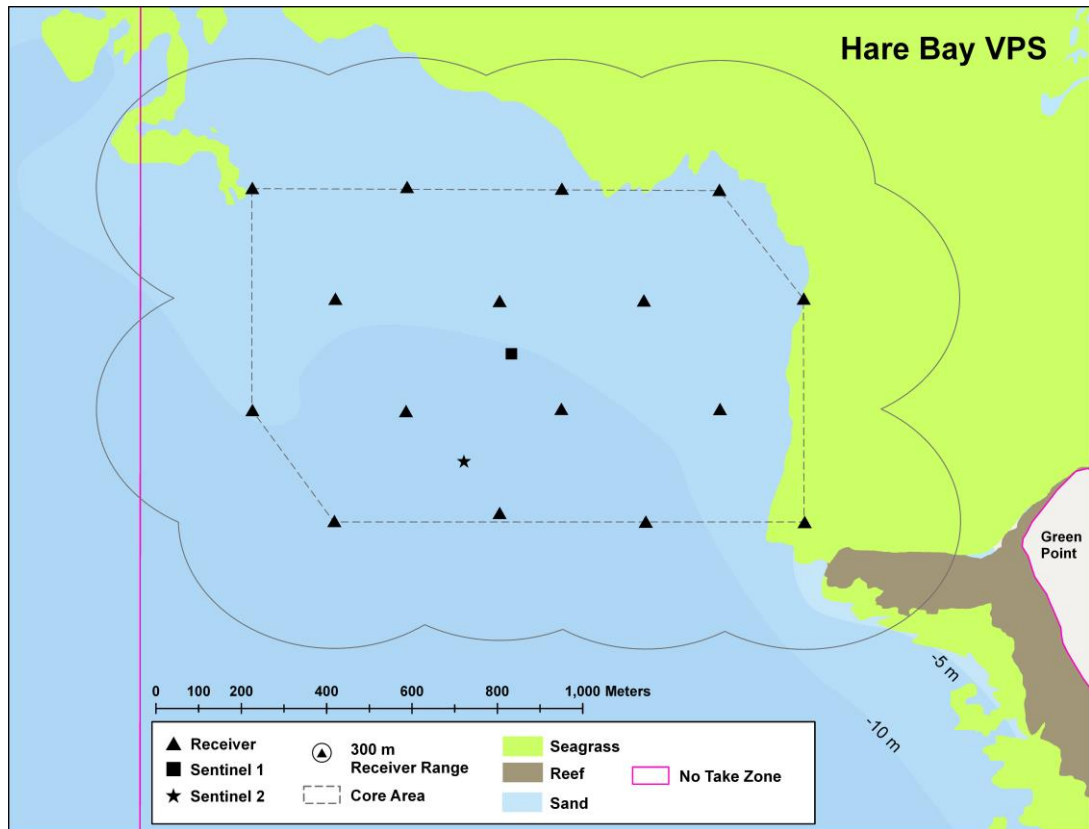
Figure 5.3: Hare Bay VPS acoustic receiver array. Receivers (triangles) were deployed with ~300 m spacing on the 1/9/14. The area within 300 m range of at least on receiver was ~202 ha.

Table 5.1: Number of fish tagged with each tag model and tag specifications.

Model	n	Length (mm)	Diameter (mm)	Weight in water	Battery life (Days)	Ping Interval (Seconds)	Power output (dB @1m)
V9-2H	8	29	9	2.9	487	110-250	151
V9-2L	16	29	9	2.9	665-738	110-250	146
V9A-2H	16	43	9	3.3	108	110-250	151
V7-4L	5	22.5	7	1	381	110-250	136
V13AP-1H	1	48	13	5.9	155	23-37	153

Figure 5.4: A bluespotted flathead post tag implantation, with ~1 cm mid-ventral incision in the abdomen. The incision was closed with one dissolving stitch tied with a double surgeon's knot.

Table 5.2: Summary of tagging data for 46 Bluespotted Flathead. Days detected on Hare Bay VPS = DD. Days post tagging of last detection on VPS = LD. Last detection residency index = RI_{LD}. Residency index = RI. Residency Index by days post tagging = RI_x (where x is the days monitored). LG largest gap in detections on the Hare Bay VPS. Gate and Bondi values are number of days post tagging with days on array 1 unless indicated in brackets. * Indicates the study ended rather than battery expiry.

Tagging Batch	Fish ID	Tail Length	Tagging Date	Tag Type	Battery Life	Days Monitored	DD	LD	RI_{LD}	RI	RI_{60}	RI_{108}	RI_{183}	RI_{449}	LG	Gate	Bondi Line
Batch 1	Fish 1	441	16/09/2014	V9-2H	487	487	68	158	0.43	0.14	0.75	0.61	0.37	0.15	48		
	Fish 2	365	22/09/2014	V9-2H	487	487	188	189	0.99	0.39	1.00	1.00	0.99	0.42	1	194 (2)	
	Fish 3	410	23/09/2014	V9-2L	666	618*	241	600	0.40	0.39	0.90	0.82	0.57	0.54	135	312	
	Fish 4	410	23/09/2014	V9A-2H	108	108	65	82	0.79	0.60	0.90	0.60			6		
	Fish 5	405	23/09/2014	V9-2H	487	487	123	241	0.51	0.25	0.88	0.72	0.52	0.27	18		257
	Fish 6	330	23/09/2014	V9-2L	666	618*	293	469	0.62	0.47	1.00	0.75	0.44	0.61	135		
	Fish 7	380	2/10/2014	V9-2H	487	487	139	253	0.55	0.29	0.93	0.76	0.51	0.31	36	257 (2)	273
	Fish 8	445	2/10/2014	V9A-2H	108	108	93	108	0.86	0.86	0.90	0.86			6		
	Fish 9	338	2/10/2014	V9-2L	738	609*	107	227	0.47	0.18	0.85	0.54	0.43	0.24	45		274
	Fish 10	345	3/10/2014	V9-2L	738	608*	45	45	1.00	0.07	0.75	0.42	0.25	0.10	0		
	Fish 11	326	3/10/2014	V9-2L	666	608*	389	445	0.87	0.64	1.00	1.00	0.98	0.87	10		
	Fish 12	440	3/10/2014	V9A-2H	108	108	75	108	0.69	0.69	0.83	0.69			18		
	Fish 13	370	3/10/2014	V9-2L	666	608*	35	40	0.88	0.06	0.58	0.32	0.19	0.08	5	42 (20)	
	Fish 14	417	3/10/2014	V9A-2H	108	108	96	108	0.89	0.89	0.93	0.89			2		
	Fish 15	431	3/10/2014	V9A-2H	108	108	106	108	0.98	0.98	1.00	0.98			1		
	Fish 16	375	3/10/2014	V9-2L	738	608*	358	381	0.94	0.59	0.98	0.99	0.96	0.80	3	394	424
	Fish 17	357	3/10/2014	V9-2L	666	608*	395	418	0.94	0.65	1.00	0.95	0.94	0.88	4		
	Fish 18	425	9/10/2014	V9-2L	666	602*	86	194	0.44	0.14	0.40	0.44	0.44	0.19	18		
	Fish 19	390	9/10/2014	V9-2H	487	487	103	108	0.95	0.21	1.00	0.95	0.56	0.23	3		
	Fish 20	350	9/10/2014	V9-2H	487	487	134	419	0.32	0.28	1.00	0.59	0.35	0.30	208		
	Fish 21	325	9/10/2014	V9-2H	487	487	352	385	0.91	0.72	0.87	0.93	0.95	0.78	14	394 (3)	444
	Fish 22	410	9/10/2014	V9A-2H	108	108	69	93	0.74	0.64	0.73	0.64			6		
	Fish 23	340	9/10/2014	V9-2L	666	602*	317	322	0.98	0.53	1.00	0.97	0.98	0.71	2	324	
	Fish 24	385	9/10/2014	V9-2H	487	487	180	242	0.74	0.37	0.93	0.76	0.82	0.40	7	247	
	Fish 25	330	9/10/2014	V9-2L	666	602*	429	458	0.94	0.71	0.98	0.99	0.97	0.94	8		
Batch 2	Fish 26	355	4/03/2015	V9A-2H	108	108	3	3	1.00	0.03	0.05	0.03			0	3	
	Fish 27	410	4/03/2015	V9A-2H	108	108	3	3	1.00	0.03	0.05	0.03			0		
	Fish 28	345	4/03/2015	V9A-2H	108	108	102	105	0.97	0.94	0.97	0.94			1		
	Fish 29	370	4/03/2015	V9-2L	666	456*	411	455	0.90	0.90	0.98	0.99	0.99	0.90	27		462
	Fish 30	335	4/03/2015	V9-2L	666	456*	158	271	0.58	0.35	1.00	1.00	0.76	0.35	37		452
	Fish 31	320	4/03/2015	V9-2L	738	456*	137	206	0.67	0.30	0.60	0.62	0.30	0.31	8		
	Fish 32	365	6/03/2015	V9A-2H	108	108	108	108	1.00	1.00	1.00	1.00			0		
	Fish 33	390	6/03/2015	V9A-2H	108	108	106	108	0.98	0.98	0.97	0.98			1		
	Fish 34	360	11/03/2015	V9A-2H	108	108	86	108	0.80	0.80	0.80	0.80			5		
	Fish 35	370	11/03/2015	V9-2L	666	449*	385	415	0.93	0.86	0.97	0.98	0.97	0.86	12		
	Fish 36	410	11/03/2015	V9-2L	666	449*	60	205	0.29	0.13	0.42	0.26	0.30	0.13	37	209	232
	Fish 37	390	11/03/2015	V9A-2H	108	108	95	108	0.88	0.88	0.90	0.88			3		
	Fish 38	490	15/04/2015	V13AP-1H	155	155	56	56	1.00	0.36	0.93	0.52			0	60	81
	Fish 39	415	15/04/2015	V9A-2H	108	108	9	10	0.90	0.08	0.15	0.08			1	24	40 (2)
	Fish 40	295	15/04/2015	V9A-2H	108	108	95	101	0.94	0.88	0.92	0.88			3		
Batch 3	Fish 41	420	7/12/2015	V9A-2H	108	108	58	108	0.54	0.54	0.85	0.54			34		
	Fish 42	250	9/12/2015	V7-4L	376	176*	68	70	0.97	0.39	1.00	0.63			2		
	Fish 43	260	9/12/2015	V7-4L	376	176*	66	67	0.99	0.38	0.98	0.61			1		
	Fish 44	415	9/12/2015	V7-4L	376	176*	65	68	0.96	0.37	0.97	0.60			1		234
	Fish 45	420	10/12/2015	V7-4L	376	175*	85	171	0.50	0.49	0.52	0.40			18		
	Fish 46	235	10/12/2015	V7-4L	376	175*	155	175	0.89	0.89	1.00	0.93			4		
Average		373					148	198	0.79	0.51	0.83	0.71	0.65	0.47	20	205	288
SE		8					18	22	0.03	0.04	0.04	0.04	0.06	0.06	6	41	44
Median		373					103	133	0.89	0.48	0.93	0.76	0.57	0.38	6	228	274

Data Analyses

Data were downloaded into the Vemco User Environment (VUE), time corrected and false detection analysis carried out (See VUE user manual and Pincock 2012). Presence-absence of each tagged fish by day within each array was plotted for each tagging batch. Within each batch, tags were placed into a further category based on battery life; accelerometer tags and coded tags and each category sorted by length of detection period within Hare Bay VPS for visual clarity.

An overall residency index (RI) for Hare Bay VPS was calculated, to examine residency over the entire monitoring period for each fish. RI was defined as the number of days a tagged fish was detected, divided by the number of days monitored (Garcia et al. 2015, Fontes and Afonso 2017). Days monitored ended either when the study ended or when the tag battery expired, whichever occurred first. An RI value of 0 indicates no residency and increases to complete residency at 1. As fish were tagged across a wide time period, some transmitters were still active at the end of the monitoring period and some had battery expiry dates during the study. Therefore, days monitored varied between 108 and 618 days (Table 5.2).

A one-way ANOVA was used to test whether residency patterns were temporally variable across the batches (3 levels, B1, B2, B3). RI data were non-normally distributed and a $\ln(X+1)$ transform was applied before analysis. To examine residency only during the time a tagged fish was detected in Hare Bay, a second measure of residency was also calculated. This index excluded times after fish had left the Hare Bay array by adjusting RI to last detection day. Last day RI (RI_{LD}) was defined by the number of days a tagged fish was detected in Hare Bay VPS divided by the number of days from tagging until the last detection (Abecasis and Erzini 2008, Fontes et al. 2014). A one-way ANOVA using the same design as above was then carried out to compare the effect of batch on RI_{LD}.

Change in residency over time was also calculated on data from all batches using RI standardised to tagging day for 60 (RI_{60}), 108 (RI_{108}), 183 (RI_{183}), 449 days (RI_{449}) and 608 days (RI_{608}) post tagging for fish with battery life remaining at each

time point (i.e. RI and number of tags at each point: $RI_{60} = 46$, $RI_{108} = 46$, $RI_{183} = 24$, and $RI_{449} = 24$ tagged fish). These cut off points were aligned with battery expiry dates for a number of tags and chosen to maximise number of tags available for each measure e.g. Increasing RI_{108} to RI_{109} reduces available tags for the index by 16. The rate of tagged fish loss or "decomposition" from the Hare Bay VPS over the whole study period was used to estimate the probability of fish loss over time. This was achieved by plotting each coded tags' (n= 25) last day of detection post tagging against the cumulative percentage of tags remaining in the array and fitting a local polynomial regression (LOESS curve with 95% confidence intervals, span = 0.7).

Large scale and migration movements

Relocation outside Jervis Bay was defined as tagged fish detected on the Jervis Bay gate array and not subsequently detected by a receiver inside the Bay. Migration was defined as any fish detected ≥2 times on IMOS receivers. Site fidelity was defined as a fish being detected on any receiver outside the Hare Bay VPS array and then subsequently returning and being detected inside the Hare Bay VPS again.

Length frequencies for flathead that left Jervis Bay and those that stayed were plotted by 20mm length intervals for visual comparison. The cumulative length distribution of the two groups were compared and tested using a two sample non-parametric Kolmogorov–Smirnov (KS) test conducted in R (R Development Core Team 2014) using the 'ks.test' function in the package 'dgof' (Arnold and Emerson 2011). The data contained no ties which enabled exact *p*-values to be calculated without the need for bootstrapping (Ogle 2016).

5.3 Results
Detection summary

Forty-six bluespotted flathead (*Platycephalus caeruleopunctatus*) were tagged with acoustic tags and detected successfully (Table 5.2). Tagged fish ranged in size from 23.5 cm to 49 cm (mean 37.3 ± 7 SE) with a skew towards larger fish (Fig. 5.5). Over 9.5 million detections from 709 transmitters were logged during the study (See supporting information) of which 1,215,075 were detections of bluespotted flathead (Fig. S5.1). Detections per bluespotted flathead ranged from 233 to 84534, with a mean

of 24056 ± 3206. False detections of bluespotted flathead tag IDs accounted for 0.008% of these detections (See false detections in supporting information 5.5).
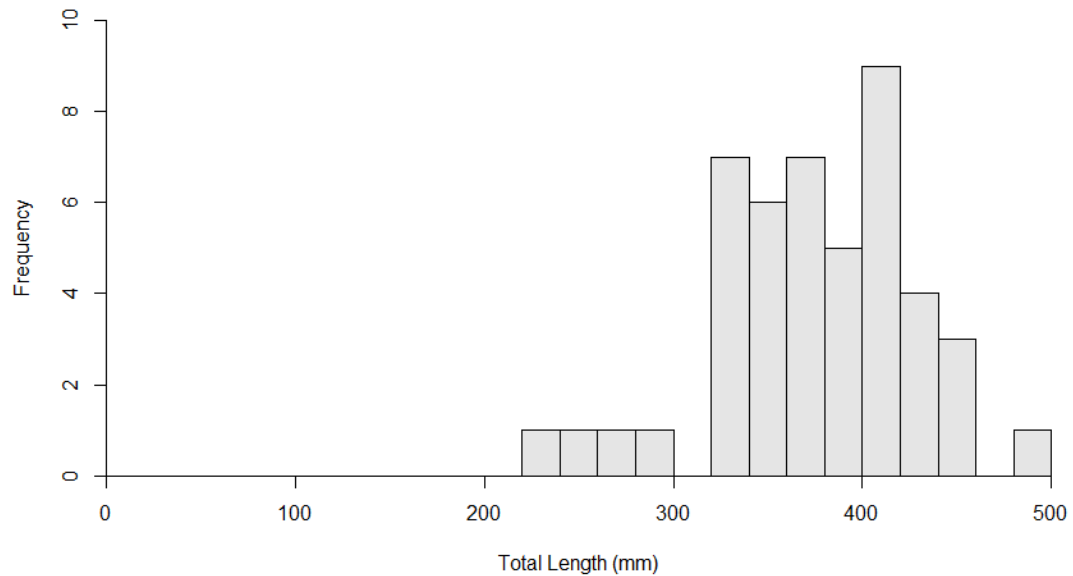
Figure 5.5: Length frequency distribution of 46 tagged bluespotted flathead based on 20 mm length intervals.

The results of range testing using static sentinel tags in the Hare Bay VPS during the study suggested that average detection rate at 150 m from a receiver was ~65% of high power and ~30% of low power transmissions (Fig. S5.4). This was likely an underestimate due to issues with the sentinel tag range test (See supporting information 5.5). Sentinel tag detections declined gradually over time with bio-fouling a possible cause (see Fig S5.2, Fig. S5.3 and supporting information 5.5). Receiver time synchronisation was consistently achieved over the study (see Fig S5.5 and supporting information). During the period when most tags were in the array (when all the fish in tagging batch one had been released), it might have been expected that collision rates would be highest and therefore sentinel tags detected less often, however sentinel tag detection rates were highest over the first 60 days of the range test (~80% high and ~50% low power transmissions detected at 150 m). Importantly, over the 164 days both sentinel tags were detected every day, suggesting that although detection range fluctuated, sufficient detections were achieved on individual receivers to confirm the presence of a stationary tagged fish on a day it was present in the array.

Due to the comprehensive grid design used in the array, the much higher transmission rate of tags used to track fish and the likelihood that fish regularly moved to new positions rather than were stationary for days and therefore mobile range tests are more representative of tagged fish (see Chapter 4.2), it was unlikely a tagged fish could be present in the array for very long before being detected.

Movement Patterns and Residency

The mean residency time of bluespotted flathead in the Hare Bay VPS (days from tagging to last detection) was 195± 22 SE days (Table 5.2). Residency time in each tagging batch ranged from 40–600 days in batch 1, 3–455 days in batch 2 and 67–175 days in batch 3 (Table 5.2, Fig. 5.6). The lower longest detection value in each subsequent batch after batch 1 is an artefact of the shorter monitoring period. Fish from all three batches could be broadly divided into five movement patterns in the Hare Bay array;

(1) fish that left the array ≤ 10 days from tagging and were not detected on the array again (n = 3, 6.5%; Table 5.2, Fig. 5.6);

(2) fish that showed short term site attachment with their last detection on the array 40–93 days after tagging (n = 8, 17.4%; Table 5.2, Fig. 5.6);

(3) fish (n = 15, 32.6 %) last detected >100 days after tagging and with a high RI_{LD} (Table 5.2), as they were detected in the array over a long period and had consistently confirmed daily presence over that time (Table 5.2, Fig. 5.6);

(4) fish (n = 12, 26.1 %) last detected >100 days after tagging and over that time had intermediate to long term periods of site attachment where they were consistently present in the array. However, these periods in the array were split by lengthy gaps in detections (>27 days) where they were absent before returning to the array again (Table 5.2, Fig. 5.6).

(5) The final group (n = 8, 17.4 %) were last detected >100 days after tagging and were regularly present in the array, however, there were numerous short gaps where over a few days, they were not detected (Table 5.2, Fig. 5.6).

There was no difference in RL_{LD} (days detected/number of days from tagging until last detection) among batches (Table 5.3). Half of the tagged fish were detected on 89% or more days between tagging and their last detection (Fig. 5.7) and most fish were consistently detected while they were in the Hare Bay array. This is reflected in a mean RI_{LD} of 0.79 ± 0.03 SE (Table 5.2). The average overall residency index score (RI, days detected/days monitored) was 0.51 ±0.04 SE (Table 5.2) and there was no difference in RI among tagging batches (Table 5.3, Fig. 5.8). RI over time showed a gradual decrease from 0.83 at RI_{60} to 0.47 RI_{449}, as fish were gradually lost from the array (see Fig. 5.9 for coded tag loss from the VPS array) a parallel decrease in RI was to be expected (Table 5.2). However, there was considerable variation between fish and a large proportion of tagged fish still being monitored at 449 days post tagging (8 of 25 fish) had an RI_{449} of between 0.70 and 0.94 (Table 5.2, Fig 5.6).

Large Scale Movements

A total of 16 fish left Jervis Bay and were not detected inside the Bay again. There was no difference in mean length or cumulative length distribution between fish that were detected leaving and those that remained inside Jervis Bay (Fig. 5.10, KS test D = 0.217, p = 0.694). Six of these fish were detected on both the Jervis Bay gate and the IMOS Bondi Line, five were detected only on the Jervis Bay gate and another five were detected only on the Bondi line (Table 5.2). Although five of the fish detected at Bondi were not detected crossing the gate, the timing of the last gate download meant that three of these fish may still have been detected on the gate array. However, these data will not be available until the receivers are next collected and downloaded in late 2017. Further, a number of tags in fish had varying lengths of battery life remaining at the end of this study (Table 5.2) and could potentially be detected on future array downloads. Nevertheless, in this study I found no evidence that any of the fish that left the Bay returned

The majority of fish that left the Bay appeared to move quickly from the VPS to the gate and then travel north (crossing the Bondi Line). All the fish that were detected on the gate had left the Hare Bay VPS array within a relatively short time period; usually within hours to 14 days (Table 5.2, Fig. 5.6). The distance between the VPS and the gate was ~8.5 km (Fig. 5.1). Of those travelling north and passing the Bondi Line, nine did so rapidly. For these fish, the average time from Hare Bay VPS to Bondi, a shortest swim distance of ~155 km, was 30 days (±5 SE) which equates to roughly 5.2 km per day. The fish detected on the Bondi line were only detected briefly on the array with eight detected for less than 25 minutes each and the remaining three being detected for between 3:26 and 15:06 hours. Nine of the fish were detected on two receivers, Bondi Line 3 and 4. These two receivers were in depths of ~62 m to ~66 m. The remaining two fish were detected in deeper water on Bondi Line 10 at ~82 m. Seven of the fish detected at Bondi were detected at the same time of year: five in a 38-day period between late May and early July in 2015 and another two in 2016 over a 10-day period from late to early July (Fig. 5.6). The remaining four fish were detected between late July and late December with no obvious patterns in temporal detection between them (Fig 5.6).

One other fish, Fish 3, was also detected on the gate array before moving back into the Hare Bay VPS array (Fig. 5.6) and only one tagged fish, Fish 25, was detected on any of the JBMP reef array receivers (Fig. 5.6), on a receiver on the southern side of Jervis Bay.

Figure 5.6: Daily presence–absence of bluespotted flathead between the 16th September 2014 and 1st June 2016. Listed by order of tagging. Fish were tagged in three batches with the following IDs and tagging periods;

Batch 1: ID 1–25, 16/9/–9/10/2014,

Batch 2: ID 25–40, 4/3–15/4/2015,

Batch 3: ID 41–45, 7/12/–10/12/15

* Indicates accelerometer tags.
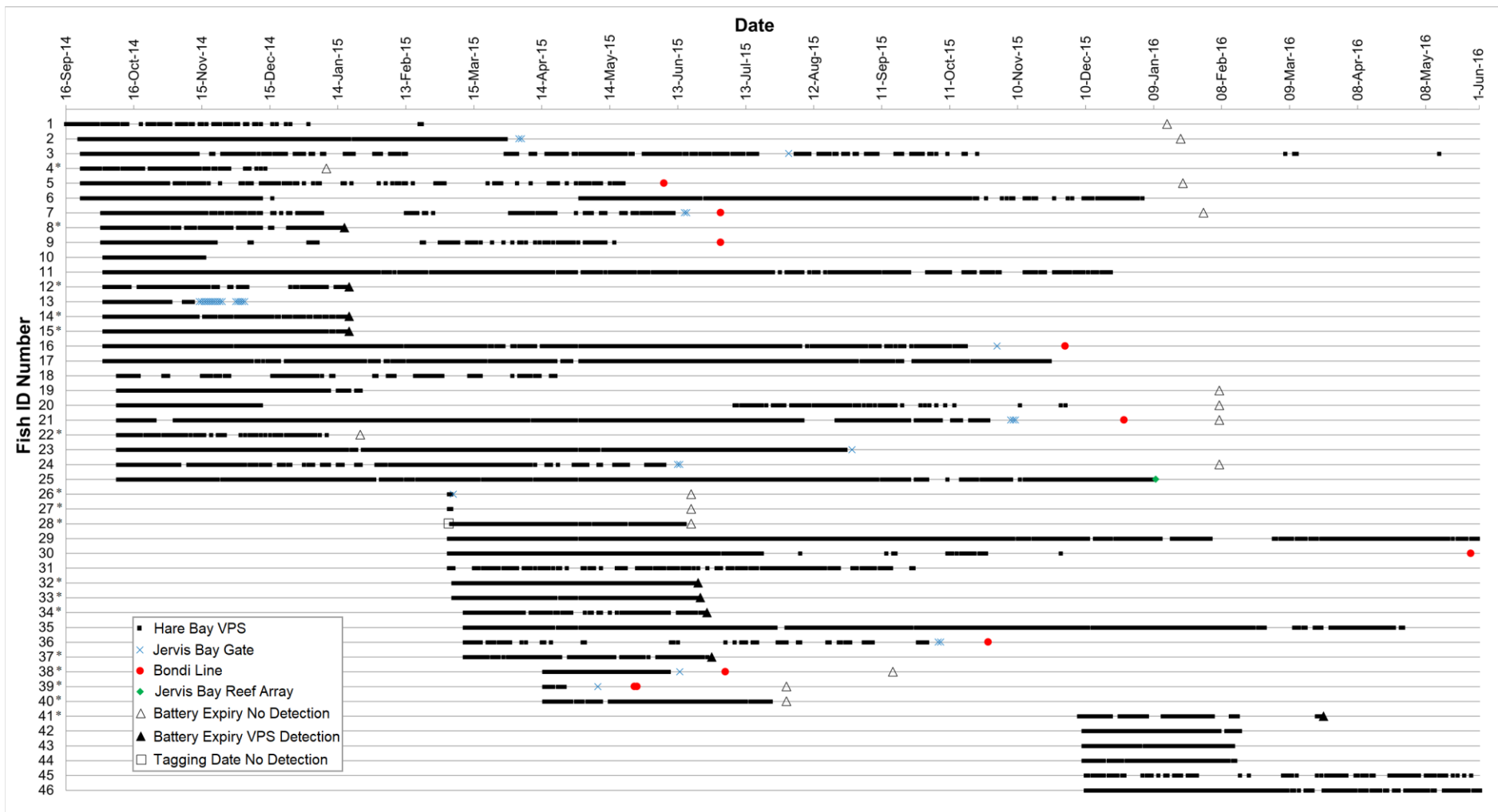
Note: Longest possible monitoring time for any tag in batch 1 was 618 days and longest length of detection was 600 days after tagging (Fish 3); in batch 2 was 456 days and longest length of detection was 455 days after tagging (Fish 29); in batch 3 was 176 days and longest length of detection was 175 days after tagging (Fish 46).
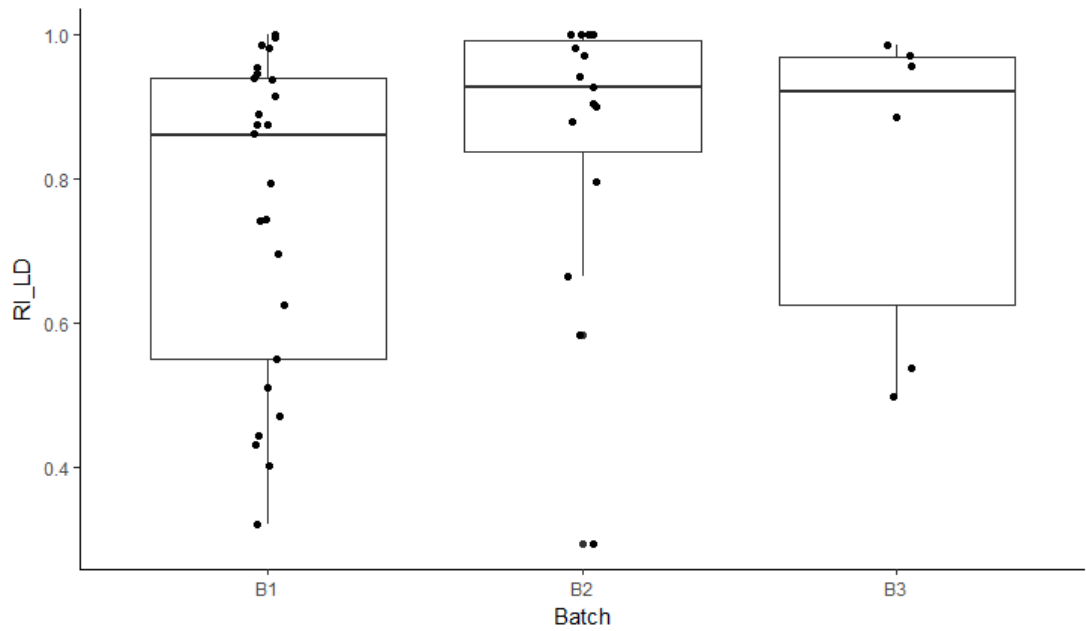
Figure 5.7: Residency index adjusted to last day detected for each fish (RI_LD) by tagging batch; batch 1(B1: n = 25), batch 2 (B2: n = 15), and batch 3 (B3: n = 6). Data plotted as box of 50% of values, median as black horizontal line and whiskers are 1.5x interquartile range. Black circles are each fish's RI_LD plotted with jitter applied so over lapping values are distinguishable.
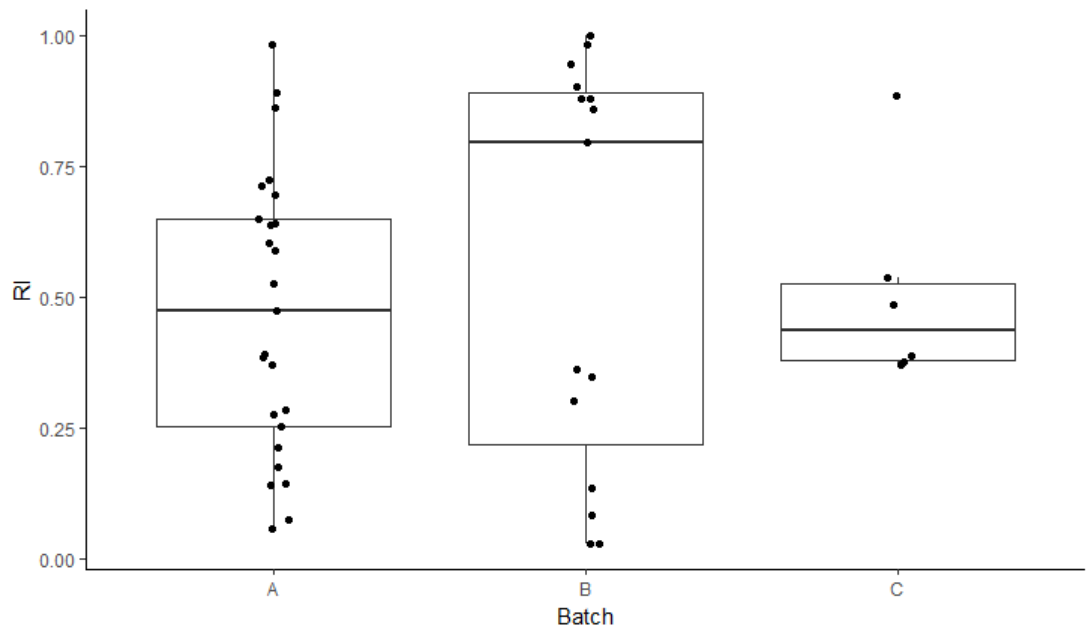
Figure 5.8: Residency index for each fish (RI) by tagging; batch 1(B1: n = 25), batch 2 (B2: n = 15), and batch 3 (B3: n = 6). Data plotted as box plot of 50% of values, median as black horizontal line and whiskers are 1.5x interquartile range. Black circles are each fish's RI plotted with jitter applied so over lapping values are distinguishable.

153
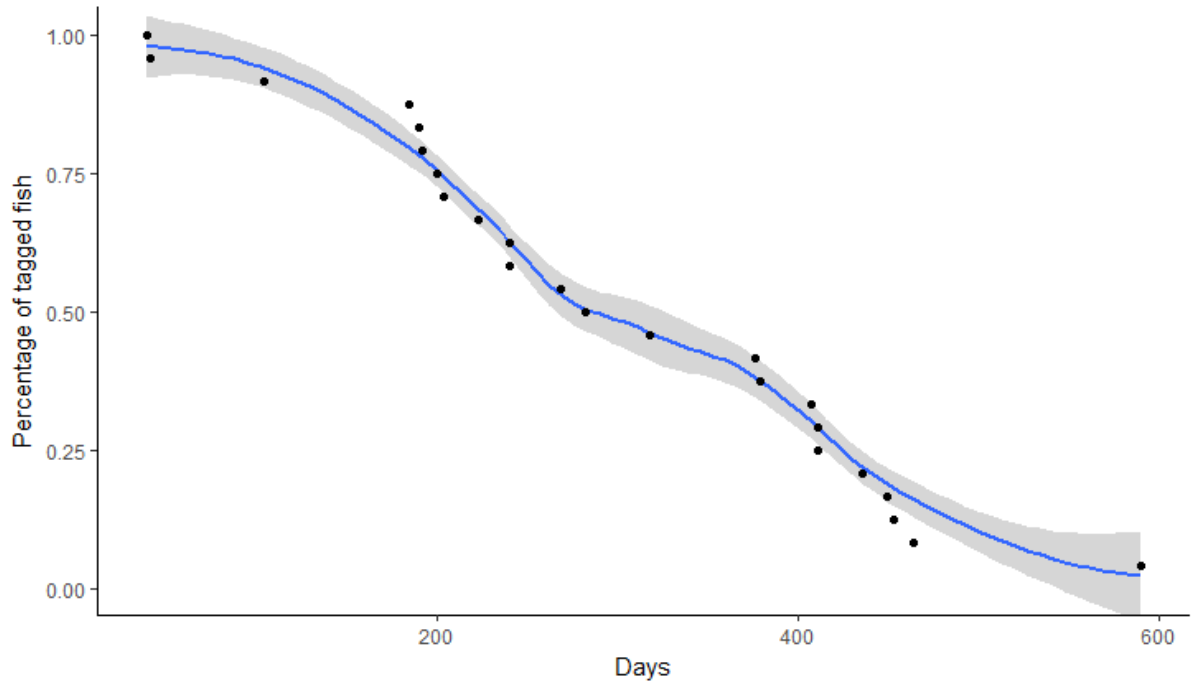
Figure 5.9: Last detection of each fish with a coded tagged plotted (back circles) as cumulative % loss from the Hare Bay array and loess curve fitted to estimate tag loss probablity over time.

Table 5.3: a) One-way analysis of variance comparing residency index (RI) by tagging batch. **b)** One-way analysis of variance comparing residency index (RI_{LD}) by tagging batch.

	df	*SS*	*MS*	*F*	*P value*
a) RI					
Batch	2	0.083	0.042	0.249	0.780
Residuals	43	7.163	0.166		
b) RI_{LD}					
Batch	2	0.159	0.080	1.068	0.353
Residuals	43	3.204	0.075		

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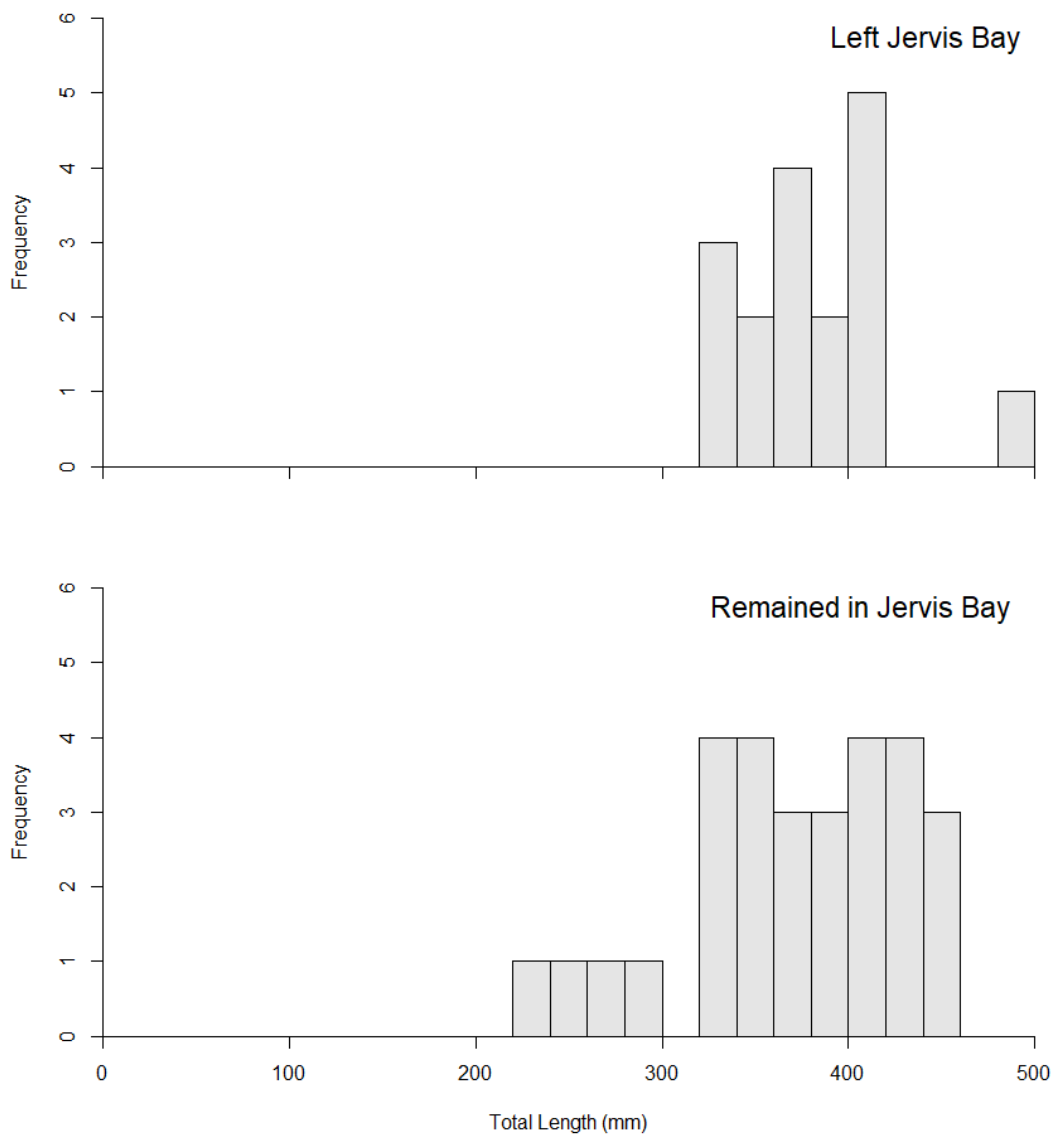
Figure 5.10: Length frequency comparison between tagged bluespotted flathead that were detected leaving Jervis Bay and those that were not detected leaving the Bay. The cumulative length distribution was tested using a two-sample Kolmogorov-Smirnov test (KS test D = 0.217, p = 0.694).

5.4 Discussion

This study provides a rare data set on the long-term movement of demersal marine soft sediment associated fish in relation to a marine protected area (MPA), and is the first examining the long-term movements of bluespotted flathead (*Platycephalus caeruleopunctatus*). These data demonstrate that a substantial proportion of tagged

bluespotted flathead showed long-term site attachment to a relatively small section of a no-take zone in Jervis Bay Marine Park. This overall pattern of long-term residency within a relatively compact area by a large portion of the population has rarely been demonstrated on marine soft sediments. Although the majority of the tagged fish showed site attachment to Hare Bay (up to 600 days), there was some variability in residency and movement patterns observed among individuals over the long-term. Close to two thirds of the tagged fish were only detected in Jervis Bay, while just over a third of the tagged fish also moved outside of the Bay and were detected up to 155 km from where they were tagged. Generally, these fish had a prolonged period of site residency before making these large-scale movements. The degree of site attachment shown and these larger movements have implications for the management of this species generally, and particularly for MPA management within this species range. In a broader context, this study is one of the first to show long-term site attachment by marine demersal fish associated with soft sediments and the results contradict current general theory, suggesting that no-take MPAs have the potential to affect soft sediment fishes.

Over the first 108 days of this study, most fish remained within Hare Bay VPS and were detected frequently and consequently short-term site attachment in this study was very high. Of note is that these short term residency patterns (e.g. $RI_{60} = 0.83$), and the number of fish that left the Hare Bay array were very similar to those found previously using active tracking ($RI_{60} = 0.75$) in 2011 (Fetterplace et al. 2016). As residency results in 2011 are so similar to those estimates in this study (2014, 2015 and 2016), they provide further support for these results being representative of general movement patterns over a wide time frame.

It seems likely that I underestimated some fish's residency for two main reasons; 1) It is likely that at least some were still in Hare Bay no-take zone (NTZ) given that the VPS covered less than half of the soft sediment habitat in that zone; 2) It seems reasonable to assume that fish with activity spaces centred on the edge of the VPS or just outside were behaving in the same way as those with activity spaces in the core area of the VPS. Therefore, those inside the centre of the VPS would be detected more consistently then those on the edge and as a result have a higher RI

and be expected to have few gaps in detections i.e. higher RI$_{LD}$. Those with numerous short gaps in detections, seemingly most consistent with the fish's activity space being on the edge or just outside of the array, will be underestimated. This is because the fish may be using areas that only partly over lapped with the array and they would only be detected when they moved into the part of their activity space that was within the VPS or when receiver performance was sufficient to detect them when outside of the array. However, without more receivers outside the array there was no way of confidently confirming this.

For fish that left and were not detected on the Hare Bay array again or were not detected on any of the other arrays between periods of consistent detections, it is not clear whether they had moved just outside of detection range of the Hare Bay array or further afield within Jervis Bay. Either way, 75% of tagged fish remained inside Jervis Bay for their entire monitoring period and therefore were under no fishing pressure (if in NTZs) or recreational fishing pressure only during this time (If outside NTZs). In addition, approximately half of Jervis Bay Marine Park lies outside the Jervis Bay gate and is made up of mostly marine sand. Of the 16 fish that left the Bay five were only detected on the gate and although it cannot be confirmed by this study, presumably these fish may have moved past the gate and remained within the MPA.

Large Scale Movements

The results of this study suggest that there is intraspecific variation in movement patterns shown by bluespotted flathead. Migration appears to be a consistent strategy among a reasonably large portion of the population, as 35% of tagged fish moved out of Jervis Bay. Migration in fish has been observed in demersal species previously (e.g. Wilhelm et al. 2015) and some species make regular movements between inshore and offshore locations that may be related to spawning (e.g. Willis et al. 2003), other species exhibit divergent migration patterns by parts of the population (e.g. DeCelles and Cadrin 2010). No bluespotted flathead were detected moving back into Jervis Bay which may be because they do not return after leaving or that they return outside the battery life of the tags in use. Double tagging of fish, with the second tag programmed to start transmitting when the first dies, would likely be required to detect fish moving back at ~600–700 days post tagging. Where fish moved

to after crossing the Bondi Line is of great interest, however, will be difficult to ever determine. This is because apart from the IMOS lines, which are widely spaced, there are rarely (if ever) acoustic arrays on soft sediments at the depths the fish appeared to be moving at (i.e. deeper than 50 m). Narrowing the search may require the use of other tracking methods such as tagging large numbers of fish with cheap external T-bar anchor tags and hoping some are captured once they have moved past the Bondi Line.

I found no evidence of size differences among fish that left and those that stayed in this study and the driver behind the large-scale movements observed by some fish in this study is unclear. As bluespotted flathead males mature at 1 year of age (21–23 cm) and females mature at 2–3 years (28–35 cm) (Barnes et al. 2011, Hall 2015), all the fish I tagged were 1 year or older, and likely considerably older for most fish, given the average length of 37 cm. It would be extremely useful in future research to evaluate juvenile movement; the implications of that information coupled with the present study are potentially large. We now know that flathead can be site attached for up to 600 days. If these fish were using the area as juveniles (and juveniles are caught in Hare Bay) then the fish tagged in this study may have been site attached to Hare Bay for an extensive period before tagging. If that is the case then the long residency periods shown here could be extended by a considerable amount.

Just under half of fish with long battery life coded tags were still in Hare Bay 300 days after tagging and based on the results all fish will be gone from an area the size of the VPS after ~600 days. Although it should be noted that some of these fish may have been detected after 625 days post tagging when the study ended and the data could be on receivers that are currently in the water.

Reef Array

The results of this study lend further support to the idea that the bluespotted flathead is a predominately soft sediment associated species. Only one fish over the entire study was briefly recorded on the JBMP reef array (3 detections). Further, that receiver range covers a large area of soft sediments, so it's impossible to say whether that fish was on sand or reef. Either way the vast majority of fish were not detected on reef. This result accords with findings from other tracking studies. Fetterplace et al.

(2016) noted that tagged fish were not detected on rocky reefs, and Keller et al. (2017) found little evidence of site attachment of these fish around an artificial reef or nearby rocky reefs. Other studies using baited underwater video have recorded bluespotted flathead occasionally on low profile sand inundated reef (Author pers. obs.), adjacent to reef (Wraith et al. 2013) and rarely if ever on complex rocky reef . For example in Jervis Bay Marine Park they were recorded on 0/96 samples over two years (Wraith 2007) and in nearby Batemans Marine Park in a study they were recorded on 5/384 samples over 5 years, some of which maybe have been on patch reef or sand between reefs (Kelaher et al. 2014).

Future Research

The drivers of the residency patterns observed here were not investigated directly, although there were no differences in the degree of residency across the seasons. There may be abiotic and biotic factors that are not obvious influencing residency patterns, even at the broad presence-absence scale investigated here, and this possibility merits further investigation in the future. I can also only speculate on whether there were environmental or biological cues for fish to make the large-scale movements observed in this study and this could also be a useful area of further research.

Bluespotted flathead appear to be very robust to internal tagging and mortality associated with tagging appeared to be zero in this study. However, a caveat in the overall results is that I assumed that overall mortality rates were negligible, though I had no data on either natural or anthropogenic mortality rates. Estimating fishing induced mortality inside Hare Bay no-take zone as zero seems reasonable, although there are occasionally instances of non-compliance by recreational fishers (Author pers. obs.). Outside the no-take zones, the risk of fishing induced mortality is higher, though still likely relatively low. If mortality was higher than the negligible level I assumed, it may mean that residency has been underestimated for some fish and as such future research should attempt to quantify mortality rates.
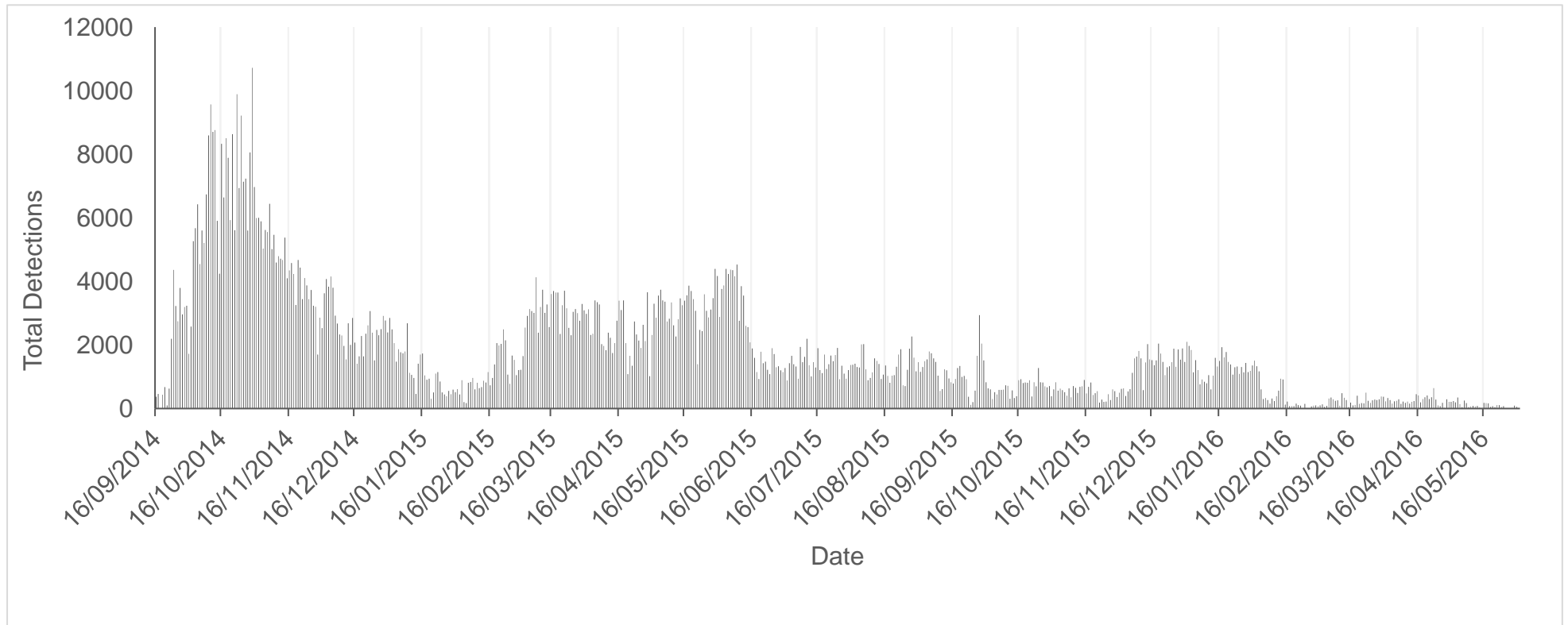
Figure S5.1: Total detections of bluespotted flathead between the 16th September 2014 and 1st June 2016. Fish were tagged in three batches; batch 1: n=25 16/9/–9/10/2014; batch 2: n=15, 4/3–15/4/2015; batch 3: n=5, 7/12/–10/12/2015.

Detection Range & Receiver Performance

Detection range testing prior to array deployment provides information on which to base array design (see Chapter 4). However, further in-situ detection range testing and monitoring of detection range variability once the array is in place is required to understand detection range under actual study conditions and determine whether the detections obtained are a reflection of the presence-absence patterns of tagged fish rather than an artefact of receiver performance (Payne et al. 2010, Kessel et al. 2014). In addition, this range testing data will aid in understanding the data that are collected e.g. how confident we can be that a fish was not within X distance of a receiver and conversely how close a detected fish was likely to be to a receiver at a given time and location in the study. The results of detection range monitoring using sentinel tags carried out in the Hare Bay VPS alongside tracking of fish are discussed in more detail below.

Fixed Long Term Range Monitoring: Sentinel Tags

Detection range monitoring using static sentinel tags was undertaken within Hare Bay VPS. Detection range monitoring was first undertaken over a 164 day period from 20/9/14 to 2/3/14. With two range tags deployed to two separate moorings (See Fig. 5.3). There was a steady decline in detections from day 1 in the test to when the tags were removed (Fig. S5.2). Although there was variability between days, the overall trend was a steady almost linear decline over time of detections of both sentinel tags. This suggests biofouling was a probable cause. Although the steady build-up of fouling can reduce receiver function, it has previously been shown to be less of an issue with new receivers (Heupel et al. 2008), such as used in the present study. Although the mooring lines were heavily affected by fouling, the receiver hydrophones, which had been painted with anti-foul, were mostly free of fouling. Further, in this study both fish positioning and synchronisation tags didn't show a similar linear decrease in detections through time, which would be expected if receiver function was the cause. It therefore is more likely it was fouling of the sentinel tags, particularly as the tags had no anti-foul and were heavily bio-fouled when retrieved, so much so they couldn't be distinguished from the mooring line they were attached too. Biofouling in Hare Bay appears to occur relatively quickly on surfaces without

antifoul (e.g. Fig. S5.3), and I recommended that in future studies sentinel tags are cleared of fouling more regularly where possible.

Successful detection of sentinel tag transmissions (Fig. S5.4) was considerably lower than detection success of range testing tags in range testing carried out prior to when the VPS array was put in place (Chapter 4), particularly for the low power tag. Lower detection rates of sentinel tags was not unexpected as in contrast to testing in Chapter 4, the test was not set up in a linear layout but rather used the isometric layout of the VPS. Consequently the moorings blocked direct line of signal to at least half the receivers and likely considerably reduced detections. An issue that would not occur in tagged fish.

Issues with detections of our two sentinel tags during this testing period mean that I more than likely have underestimated detection range of tagged fish in this study. However, even based on this conservative estimate, a presence of sentinel tags was achieved on all study days suggesting it is highly likely a tagged fish in the array would be detected also if it were present (as discussed in main study results) - particularly when mobile range testing results in Chapter 4.2 are considered.
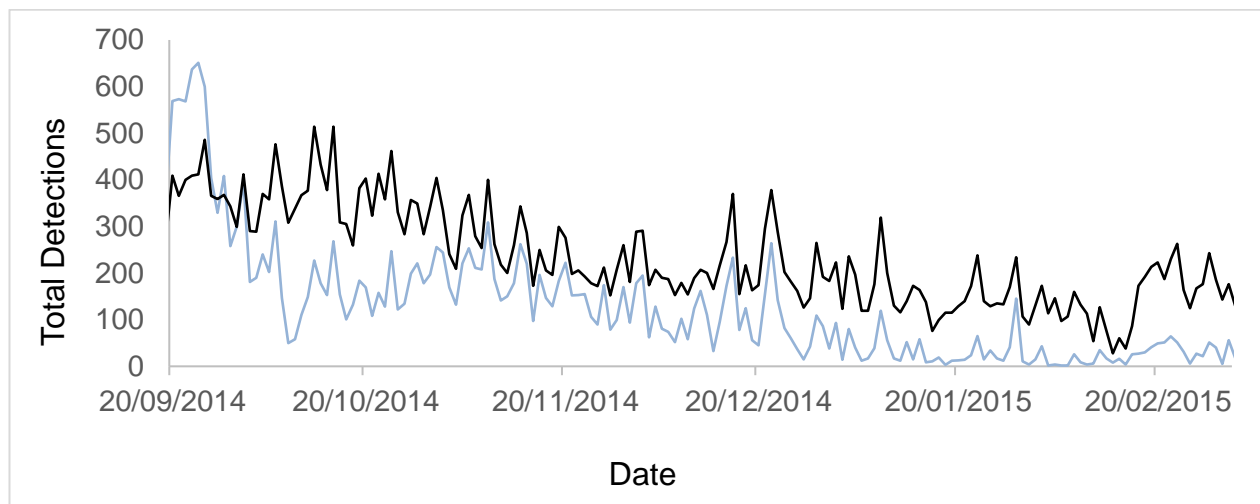
Figure S5.2: Total detections by day (165 days) for each of the two sentinel tags. Tag 1 (low power) blue line and sentinel tag 2 (high power) black line.

Range test using sentinel tags was planned for the whole study, however, I moved the sentinel tags after 165 days, to a new location with different distance

intervals. In hindsight, this was a mistake and detection comparisons couldn't be directly compared to the first 165 days. In the second location, both tags had relatively constant but much lower detection rates than the first location for the first 6 months (they were cleared of biofouling in the middle of this period). Detection rates then dropped gradually to very few detections by the end of the study. The cause of the generally lower detection rate at the 2nd location seems likely to be because the new location was closer to the perimeter of the VPS. The marked decrease in sentinel detections in the 2nd location that began after the first 6 months was likely because they weren't cleared of fouling in that time (> 360 days). As a result of these issues I did not include the 2nd period range estimates.

Figure S5.3: Biofouling of a mooring line and buoy 189 days after deployment. The tracking receiver itself was relatively fouling free, particularly the receiver head containing the hydrophone which had been painted with anti-foul.
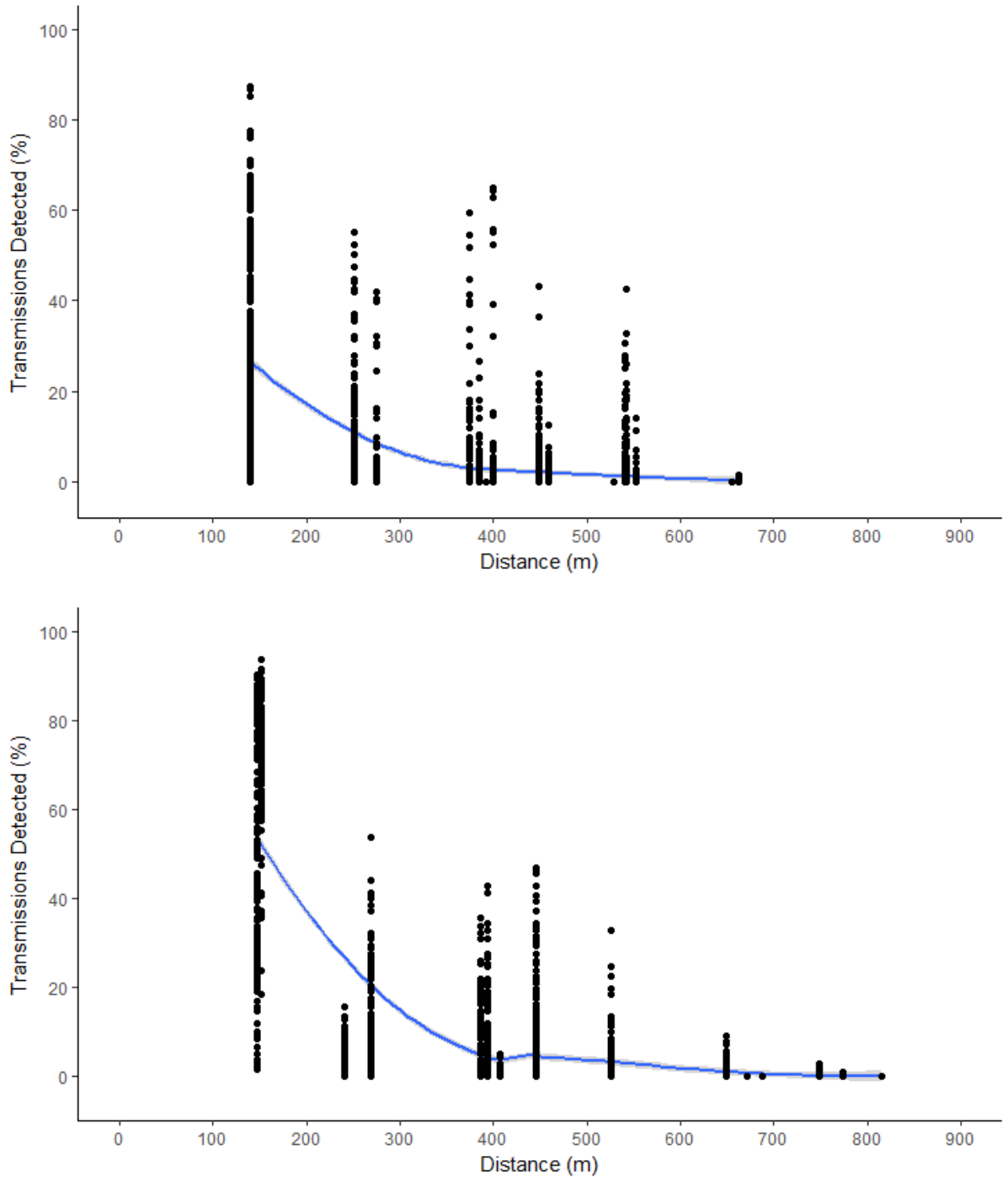
Figure S5.4: Detection range profiles for acoustic receivers and sentinel tags over 16 distance intervals using all the receivers in the Hare Bay VPS. Data are daily percentage of tag transmissions successfully detected by fixed acoustic receivers. Top panel: tag 1 (V9-2L), and bottom panel: tag 2(V9-1H). A LOESS curve (Local Polynomial Regression, ± 95 CI) of detection probability by distance is fitted to both data sets.

Synchronisation Tags

All 20 receivers in the Hare Bay array had a V16 synchronisation tag attached to the mooring so that receivers clocks could be time synchronised. Time drift on individual

receivers occurs following a predictable linear pattern and the longer that receivers are in the water, the greater the time drift (receiver clocks are reset at each download). Time drift results in some receivers with overlapping detection range recording the same tag transmission as being detected at different times. For example a transmission sent at 10:00, detected at 10.00:18 on one receiver and 10:01:00 on another receiver will appear to be two unique detections unless time corrected. In some cases during this study, time differences of transmissions due to time drift was greater than 10 minutes. To achieve accurate time synchronisation within a VPS ~ 3 detections of synchronisation tag transmissions on mulitple receivers is required per hour. In this study, detection of synchronisation tags was very high (Fig. S5.5). This also means that detection probability of V16 tags generally would be high. While this provides support for receiver detection range being good throughout the study (i.e. tags were being detected by more than 3 receivers consistently), nearly all the tags used in fish were V9 tags with lower power and thus the results of mobile, stationary and sentinel range testing with V9s are more informative.

The results of these various tests combined (here and those in Chapter 4) suggest that that the presence-absence patterns observed in this study are a good estimate of bluespotted flathead presence-absence over the study period. Positioning success was not as high and should be taken into account in future tracking studies looking at fine scale behaviour patterns within the VPS.

False Detections

Over 9.5 million detections from 709 transmitters were logged during the study. Of these tags, 605 transmitters were identified as highly likely to be false tag IDs in false detection analysis in VUE software and one further tag in further manual inspection. These 606 tags only accounted for 992 of total detections. Of the remaining 103 legitimate tags, 93 were identified as belonging to Jervis Bay DPI linked projects (including the 2 reference tags, 20 sync tags and all tags in bluespotted flathead in this study), 7 tags were in sharks from various researchers in other locations (identified by word of mouth, Vemco assistance or through the IMOS animal tracking database), 1 was an embargoed tag listed on the IMOS animal tracking database and two tags were unknown.
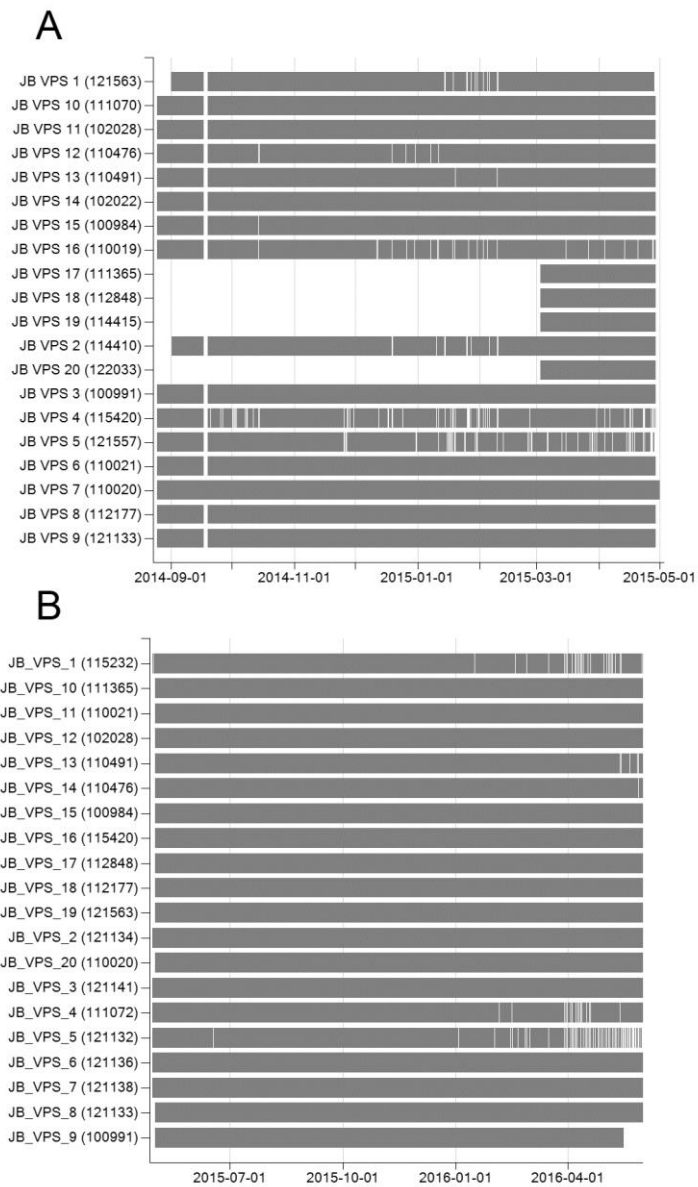
Figure S5.5: Time sync availability over the study for the 20 receivers making up the Hare Bay VPS. Grey line represents each day of successful day time synchronisation between each receiver location. A) test download and first download (empty line through data is receiver changeover and download when only one receiver was in the water). Note that receivers 17–20 were added to the VPS at a later date in March 2015). B) Second download of receivers

Chapter 6 General Discussion

This thesis is a rare example of an assessment of the impacts of protection on marine soft sediments over a wide spatial and temporal scale and represents a significant step in addressing the lack of research on this habitat.   Prior to this study, there had been very little research attempting to gauge whether demersal soft sediment fishes respond to protection in marine protected areas (MPAs). This is despite the extensive inclusion of this habitat within MPAs. The use of stereo BRUVs provided a level of detail on soft sediment fish assemblages that was non-existent prior to this study. The 245 successful BRUV deployments provide a permanent baseline record over a wide spatial area, a range of depths and fisheries management levels. These data can be used in future studies to make long term assessments and compare changes in patterns in assemblages over various management levels.

In this study, I did detect effects of protection, however they were not those that were necessarily predicted or of a magnitude that might be expected if demersal fish assemblages were experiencing a high level of fishing pressure on the soft sediment habitats assessed. The strongest effects were for species that are not considered highly targeted species by recreational fishers; eastern fiddler rays and longspine flathead. Fiddler rays appear to be affected within Jervis Bay and on the open coast, where there was only one observed in fished areas outside the MPAs. Fiddler rays have been reported to form a considerable component of bycatch in commercial trawling operations (Marshall et al. 2007), which could explain the patterns I observed. This species may be a particularly useful indicator species to assess the effects of trawling. Incorporating long term monitoring into assessments of the effect of MPAs may have additional fisheries benefits for species in the assemblage that have no stock assessments, those assessed sporadically and for bycatch species. For species like longspine flathead (no stock assessment in NSW) and eastern fiddler rays (Undefined stock status) long-term fisheries independent studies could be used to flag population crashes or changes in population characteristics that may otherwise go unnoticed.

There were clear assemblage wide multivariate effects shown in offshore comparisons. These effects were probably the result of many taxa contributing, but with few of these taxa having differences among zones detected in univariate analyses on their own. Clearly, only having 3 replicate locations within each zone for the open coastal study meant that the power of the analysis was low. Adding more locations in repeat testing in Batemans marine park (BMP) and fished open access areas would increase the power of the assessment and provide a better indication of the patterns observed in the current study. There are another three no-take zones on the open coast within BMP that definitely include suitable soft sediment habitats. Sampling these extra sites was beyond the scope of the current study but including these in future surveys would more than double the number of locations in the assessment, substantially increasing the power of future tests. Within Jervis Bay, no extra locations were possible as the zones within the Bay were extensively sampled both spatially and also over time.

Intriguingly, I found no indication of any effect on the main targeted fish species, blue spotted flathead. This species showed no striking difference among zone in terms of abundance or size. This was surprising, considering how important this fish species is to commercial and recreational fishing sectors (Stewart et al. 2015, West et al. 2015). This may indicate that recreational and commercial fishing pressure is at an ecologically sustainable level. The use of no-take MPAs as references for fisheries assessments has been suggested previously (e.g. Breen 2007), however, the utility of this approach has not been tested in the study region as far as I am aware.

The age of an MPA can influence the response of species to protection, with targeted species in older MPAs more likely to show a positive and stronger response to protection than those in young MPAs (Claudet et al. 2008, Edgar et al. 2014). Bluespotted flathead mature quickly (males mature at 1 year of age and females at 2–3 years) and are highly fecund broadcast spawners (Barnes et al. 2011), so have therefore had multiple generations since MPA implementation (~8 years 6 months in BMP and ~13 years 8 months of protection in JBMP at last sampling). Given this, it seems unlikely that MPA age is the reason behind lack of response detected here.

However, duration of protection is not always a clear predictor of response (Malcolm et al. 2016), particularly for slower maturing and less fecund species.

In contrast to bluespotted flathead, Eastern fiddler ray (*T. fasciata)* and shovelnose ray (*A. rostrata)* are slower to mature and produce few young; 2–3 pups (Bray 2018) and 4–18 pups respectively (Kyne and Bennett 2002). Although age at maturity data isn't available for Eastern fiddler ray, it is likely similar to the closely related Southern fiddler ray (*Trygonorrhina dumerilii*) in which age at maturity is 4+ years for males and potentially 10+ years for females (Izzo and Gillanders 2008). Based on these life history data one might predict that these species would have a considerable lag in response to protection. The Eastern fiddler ray and shovelnose ray in fact only showed a response in abundances inside Jervis Bay NTZs in the last year of sampling. Although this prediction, life history and results are in concordance, further sampling is needed to determine if the trend continues consistently post 2015. Additionally, some responses may take decades to manifest (Malcolm et al. 2016), so establishing some impacts of protection in MPA's that have been in existence even for the timescales here may not yet be possible. A further complication is that for many of the other species in the assemblage there is little or no life history available so it is difficult to make predictions on response times or to say whether it is likely that reserve age is the reason for a lack of effect in these individual species. Repeating the sampling undertaken in this thesis in follow up years will help shed light on the longer-term impacts of protection and should be a priority.

An alternative explanation for the patterns I have observed is that the zones and MPAs were not large enough or positioned in the right places for more substantial effects to occur. Fish moving between the various management zone would explain many of the observed results. However, the tracking data suggested that soft sediment fishes such as bluespotted flathead do show a high level of residency (some individuals up to 600 days within the one area), which suggests the size of the zones is not the issue. However, there are very few species in the soft sediment assemblage that have published movement data on which to base predictions on effective zone size. Currently, there are simultaneous studies on other soft sediments species underway in Jervis Bay, that together with movement data gathered in my thesis should go some

way to filling this gap in the literature. These studies, although still ongoing, suggest that fiddler rays (preliminary data; Adams 2016) and longspine flathead (*in prep.* Fetterplace, Knott, Adams, Taylor, and Davis) also show high levels of residency comparable to those presented here. Considering that both these species showed effects of zoning it may be that they have greater levels of residency over a longer period and, hence, would be more likely to show a response to MPAs and their zones. The residency data I have collected provides reasonable support to suggest that the marine park zones and the marine parks themselves are likely to be adequate to protect large numbers of bluespotted flathead for reasonably long periods of time (e.g. 12–18 months). Assessing juvenile movement in the future research could potentially show that these residency periods are considerably longer.

The large, rapid and direct movements made by a substantial proportion of the tagged flathead, complicate the residency picture somewhat for this species. Just over a quarter of the tagged fish (12 of 46) made movements of up 200 km within 2–3 weeks. When fish are moving over these distances it is unlikely that small-scale management will have much of an effect. Hence, it may be that no-take zones and marine parks may provide a substantial temporal refuge, but that many fish appear to move large distances, primarily to the north, which would reduce the apparent effect of MPA protection. This "spill over" of some adult bluespotted flathead is likely to reduce the magnitude of any effect on abundances within both NTZs and the MPA as a whole, and at the same time may be subsidising numbers in areas surrounding the reserve; an outcome that can be a fisheries benefit (Gell and Roberts 2003, Russ et al. 2004, Russ and Alcala 2011). These movement results demonstrate the complex movement patterns that need to be considered in order to determine the likely effectiveness of protection on fish species. It should be noted that prior to this study no evidence existed indicating that bluespotted flathead made such large-scale directional movements. The question then becomes whether the residency of these fish over periods of 12–18 months (and possibly longer as 2/3 of the tagged fish were not detected making large distance movements) are enough for the MPA to have an effect or whether the large-scale movements of a substantial proportion of these fish would be likely to erode any spatial patterns in the abundance and size of the fish in relation

to MPAs. Further research is clearly needed to determine this intriguing and complex movement pattern and its effect of spatial patterning in this species.

For logistical reasons, the acoustic tracking was carried out within an ocean-dominated embayment. An assessment of movement needs to be carried out to determine whether these residency patterns reflect that of fish on the open coast and wider shelf area, as it is possible that fish in these areas may be more mobile than those within the studied embayment. It should, however, be considered that prior to this study no movement estimates existed for this species and I see no reason why my results here would be inconsistent with those of fish on the wider coast. However, now with the experience of working on this species with acoustic telemetry technology, I would be confident in assessing their movement patterns on the open coast.

Also, of note is that all the bluespotted flathead detected of Bondi were in water deeper than 60 m and some were detected in at least 80 m of water. In BMP, with the zones extending out to 3 nautical miles, this depth would be covered. However, in JBMP, with the zones only extending 1.5 km out from the shore, this depth would not be included in the MPA. Hence, there would be some indication that the coverage of the depths may not be fully adequate at JBMP. Similarly, the narrower width and generally smaller zones in JBMP (i.e. generally 1.5 x 3km) compared to BMP (i.e. generally 5.56 x 6 km) means that stray fishing effort from poor position could have a greater effect in JBMP than in BMP. It should be noted, however, that the NTZ sampled in BMP was one of the smaller open coastal zones. Future sampling of the larger southern BMP no-take zones should be a priority but were beyond the scope of the current study.

There is also the real potential that lack of enforcement may be playing a role in limited PPA or NTZ effects. Commercial trawl operators have been observed and on occasions fined for fishing within the MPAs and within no-take zones. Currently, no estimates of compliance exist so it is difficult to determine how much illegal fishing is occurring and how much of this activity could effectively remove any biomass of fish in these areas. However if non-compliance is occurring in an MPA, it can

potentially negate any benefits of protection (Bergseth et al. 2015). Assessing this should be a future priority for Fisheries NSW (who are responsible for NSW MPAs). Doing so would provide a better indication of whether NSW Marine Parks are affecting soft sediment fish assemblages.

There is considerable expense and effort associated with protecting large areas of the ocean. If enforcement is required to ensure compliance, then protecting vast tracts of ocean is likely to be prohibitive in its cost. Soft sediments are likely to be particularly costly as fishing effort is often dispersed across wide areas and harder to monitor. This is in comparison to rocky reef for example, where effort is often concentrated on a few restricted locations (e.g Lynch 2006). If protection on soft sediment isn't effective, then perhaps it should be revisited, and resources redirected to management of habitats, where protection has been shown to be effective. Short falls in management capacity have been identified as the key reason limiting the success of MPAs (Gill et al. 2017), so it makes little sense to expend limited funding and effort on extensive areas of habitat if there is no demonstrable benefit. This is, however, different from not being assessed – which is currently the case with soft sediments.

The use of acoustic telemetry provides results that are useful to assist with the interpretation of complementary density, size and abundance data collected simultaneously using baited remoted under water video (BRUVs). For example, when assessing the first and second chapter in isolation, my results suggest that bluespotted flathead are relatively unaffected by no-take MPAs; either because i) fishing pressure is low relative to their fecundity (and therefore fishing has little impact on abundances in PPAs), or ii) they are so highly mobile relative to the zone sizes in the two MPAs that they are unaffected by zoning, or iii) fishing is occurring inside the NTZs. However, when the results are considered in light of the strong residency patterns in Chapter 5, it becomes much less likely that extensive mobility could be the reason behind the abundances patterns observed. The use of the second method, acoustic tracking, therefore eliminates a potential explanation for the results gained using BRUVS. Assuming compliance is high (i.e. there is little fishing occurring in NTZs), and inside Jervis Bay at least, it appears to be a reasonable assumption due to high

levels of enforcement, then it suggests that the impact of recreational fishing inside Jervis Bay is at a sustainable level. Whether low compliance can be assumed in offshore waters is not as clear (as discussed earlier).

The use of BRUVS and acoustic telemetry as complementary techniques to investigate movement and abundance/biomass in the same study, is uncommon. There are some examples on sharks (Bond et al. 2012, Acuña-Marrero et al. 2017, Papastamatiou et al. 2017) but other than Fetterplace (2011) and by extension this study, there are no examples on bony fish that I am aware of (though see studies where visual census data and acoustic telemtry were used as complementary methods e.g. Zeller and Russ 1998, Abecasis et al. 2015). Although their use together is novel, both methods improve our understanding of the ecology of fish, and in unison they provide an additional means of assessing results gained by either method. Overall, combining this information will allow better management of the bluespotted flathead, and other fish in the assemblage when movement data becomes available.

In conclusion, the effect of protection on demersal fishes inhabiting soft sediments is poorly understood. This is despite the dominance of soft sediment habitats in the ocean and the widespread inclusion of large areas of soft sediments in MPAs. Whether protection can have the same impact on demersal soft sediment fish assemblages as those on other habitats has rarely been assessed and is a critical gap in the understanding of MPA efficacy. My thesis represents an important step in filling this gap by providing one of the few assessments of soft sediment fish community response to MPA implementation. My results demonstrate that temperate demersal fishes found on marine soft sediments can show strong residency and that they can be influenced by protection within MPAs at a number of spatial scales. However, many species show no response and for those that do, the range of responses are highly variable. At the assemblage level responses were also varied with no response detected inside Jervis Bay, but clear differences in assemblages among all management zones in offshore waters. This study has broken new ground, providing strong spatial and temporal estimates of the relative abundance and size of the soft sediment fish assemblages along the temperate south-east coast of Australia. Furthermore, I have provided robust long-term (up to 618 days with more data to come) estimates of the

residency and large-scale movement patterns of one of the most important commercial and recreational soft sediment species, which prior to this study had been effectively unassessed. In a worldwide context, this study, together with my preliminary work (Fetterplace 2011, Fetterplace et al. 2016) represents a) the first comprehensive assessment of effects of MPA on soft sediment fishes across multiple years, NTZs, MPAs, and b) one of the few studies assessing the movement of a soft sediment fish in relation to MPAs across multiple years and c) a rare example of movement data on a soft sediment associated fish species based on a relatively large sample size (51 fish tracked in total). It is envisaged that the use of abundance and movement data together, will be more broadly adopted to improve the oft neglected assessment of one the most protected habitats and fauna assemblages—marine soft sediments and soft sediment associated fishes.

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Appendix A: Published Manuscripts and Presentations

During my PhD I have published; two journal articles as lead author, one as 2nd author and have one in review as 2nd author. I have also published 4 datasets and presented data from my thesis at national and international conferences.

Publications

Fetterplace, L., Davis, A., Neilson, J., Taylor, M. and Knott, N. (2016). "Active acoustic tracking suggests that soft sediment fishes can show site attachment: a preliminary assessment of the movement patterns of the blue-spotted flathead (*Platycephalus caeruleopunctatus*)." Animal Biotelemetry 4(1): 15.

Adams, K. R., Fetterplace, L. C., Davis, A. R., Taylor, M. D., Knott, N. A. (2018). Sharks, rays and abortion: the prevalence of capture-induced parturition in elasmobranchs. Biological Conservation 217: 11-27.

Fetterplace LC, Turnbull JW, Knott NA, and Hardy NA. (2018). The devil in the deep: expanding the known habitat of a rare and protected fish. European Journal of Ecology 4(1):22.

Gibbs L, Fetterplace LC, Rees MJ & Hanich Q (In Review). Determining the effects and effectiveness of shark hazard management: The Shark Meshing (Bather Protection) Program, NSW, Australia. People and Nature.

Datasets

Fetterplace LC, Taylor MD and Knott NA. (2016). Data from: 'Jervis Bay Marine Park: Active Tracking of Blue-spotted Flathead'. ZoaTrack.org.

Fetterplace LC and Knott NA. (2018): Dataset: Offshore Reef Fishes of South Coast NSW. figshare. Fileset. https://doi.org/10.6084/m9.figshare.5947012.v1

Fetterplace LC, Gibbs L and Rees, MJ. (2018): Shark Meshing (Bather Protection) Program, NSW, Australia: Catch and Effort Data 1950–2017. figshare. Fileset. https://doi.org/10.6084/m9.figshare.6222155.v2

Fetterplace LC. (2018): Acoustic and Satellite Telemetry Studies on Marine Fish: 1986–2013. figshare. Fileset. https://doi.org/10.6084/m9.figshare.6275312

Invited Seminars

Fetterplace LC (2018). Can MPAs conserve fish on marine soft sediments? Invited Speaker, Department of Ecology, Environment and Plant Sciences seminar series, Stockholm University.

Selected Presentations

Fetterplace L, Knott N, Adams K, Taylor M and Davis A (2018). Can MPAs conserve fish on marine soft sediments? The 5th International Marine Conservation Congress, Kuching, Malaysia.

Fetterplace L, Taylor M, Davis A and Knott N (2018). Are open coastal soft sediment fishes always highly mobile? Swedish meeting of PhD students in Fish Biology, Älvkarleby Fisheries Research Station, Sweden.

Fetterplace LC (2016). "Nobody wants to research fish that live on sand". Australian Society for Fish Biology Conference – Student Science Communication Three Minute Thesis Competition.

Fetterplace LC, Taylor M, Davis A and Knott N. (2015). "The conservation of soft sediment fishes: the vast unknown". Plenary session – Ecological Society of Australia Conference.

Fetterplace LC, Adams K, Taylor M, Davis A and Knott N (2015). "Are soft sediment fishes always highly mobile? An assessment of the movement patterns of the bluespotted flathead". Australian Society for Fish Biology Conference.

Fetterplace LC (2015). "Marine Science in Jervis Bay". The sea & me film premier Huskisson – Invited speaker.

Fetterplace LC (2015). Invited speaker at Talking Landcare 2015, Theme: marine debris – "The science behind the marine plastic issue"

Fetterplace LC (2014). "Fish on sand". Fish Habitat Workshop Wollongong 2014 (South East Wetland Carers Network) – Invited speaker.

Fetterplace LC (2013). "Fish diversity on soft sediments" Student Conference for Conservation Science Brisbane.

Reports
Knott N, Ferguson A, Fetterplace L (2014) Coastal Erosion Remediation: Environmental Baseline Study, Final Report – Initial Sampling Round. NSW Department of Primary Industries Fisheries Research, Marine Ecosystems. Huskisson, NSW.

Fetterplace LC and Knott NA. (2015). Standard Operating Procedure for Field Based Surgical Implantation of Acoustic Transmitters in Fish: Jervis Bay Marine Park. Huskisson, NSW.

Co-Author Presentations

Fetterplace L, Taylor M, Davis A and Knott N*. (2017) Are open coastal soft sediment fishes always highly mobile? An assessment of the movement patterns of the bluespotted flathead. International Conference on Fish Telemetry, Cairns, June 2017. *Presenting author.

Appendix B: Systematic Review of Acoustic Telemetry and Satellite Telemetry Based Studies

The following steps were undertaken to find acoustic tracking and satellite telemetry papers to explore (1) research effort undertaken by broad habitat type and (2) spatial patterns in the use of this technology globally.

To compile an initial list of marine and estuarine tracking publications, a web of science search using terms based on those used in Kessel (2015) was carried out and included papers up to 31st December 2013. The search terms for the Kessel (2015) dataset were "acoustic, ultrasonic, sonic, satellite, PSAT and SPOT proceeded by each of the words telemetry, tracking and tag". The search terms provided ~800 papers, however this did not replicate the study list of the original paper (including papers cited in the searched publications). I then cross referenced the list with the Kessel (2015) dataset and included all additional papers. The total number of papers after this combined search was 1170 and I included these in my initial database.

Papers in Kessel (2015) already had species from each publication defined, so I defined species for the remaining papers. I then removed any studies not tracking fish (note: if a publication tracked fish and non-fish species it was included). Following Kessel (2015) each species tracked was assigned a study number. If more than one species was tracked in a publication then each species was assigned a unique 'study' number (i.e. if a study

tracked Great Whites and Whip Rays then a study number was assigned to both and a point on the global maps was plotted for each). Similarly, on the few occasions where there were different tagging locations within a paper (e.g. fish tagged in Russian and fish tagged in the US) I assigned each group of fish tagged at each location a study number. If there were different tagging years within a paper, each was considered a different study and assigned a study number (e.g. if fish were tagged in one batch in 2009 and another in 2011 than they were both plotted).

This produced a list of 853 studies on fishes (bony and cartilaginous).

- 624 on Teleosts (494 acoustic, 130 Satellite).
- 229 on Elasmobranchs (126 acoustic, 103 Satellite).

I then used FishBase (2017) to assign an attribute(s) based on water type with categories marine, freshwater, brackish (fish were assigned to one or multiple categories). Freshwater only species were then removed (Table B1.1). Species classed as occurring in both freshwater and brackish categories were removed (Table B1.1) as they spend the majority of time in freshwater and rarely enter marine waters. I also removed catadromous species that spend most time in freshwater and only enter marine waters to spawn/breed (Table B1.1). During sorting of the remaining studies, I removed a number of papers included by Kessel (2015) that either weren't on fish or didn't using telemetry (Table B1.2)

Table B1.1: Species assigned both freshwater and brackish categories and catadromous species that spend most time in freshwater and only enter marine waters to spawn/breed that were removed from the data set.

Category	Common Name	Scientific Name
Freshwater	largemouth bass	*Micropterus salmoides*
Freshwater	lake trout	*Salvelinus namaycush*
Freshwater	South American Perch	*Percichthys trucha*
Freshwater	Bighead carp	*Hypophthalmichthys nobilis*
Freshwater	Silver carp	*Hypophthalmichthys molitrix*
Freshwater	Razorback sucker	*Xyrauchen texanus*
Freshwater	Mekong giant catfish	*Pangasianodon gigas*
Freshwater	Smallmouth bass	*Micropterus dolomieu*
Freshwater	Crucian carp	*Carassius cuvieri*
Freshwater	Crucian carp	*Carassius auratus*
Freshwater	Dark chub	*Nipponocypris temminckii*
Freshwater/Brackish	Lake whitefish	*Coregonus clupeaformis*
Freshwater/Brackish	Brown bullhead	*Ameiurus nebulosus*
Freshwater/Brackish	European perch	*Perca fluviatilis*
Freshwater/Brackish	Common carp	*Cyprinus carpio*
Freshwater/Brackish	European catfish	*Silurus glanis*
Freshwater/Brackish	Lake sturgeon	*Acipenser fulvescens*
Freshwater/Brackish	White sucker	*Catostomus commersonii*
Freshwater/Brackish	Taimen	*Hucho taimen*
Freshwater/Brackish	Lake sturgeon	*Acipenser fulvescens*
Freshwater/Brackish	Bull trout	*Salvelinus confluentus*
Freshwater/Brackish	Burbot	*Lota*
Freshwater/Brackish	Northern Pike	*Esox lucius*
Freshwater/Brackish	Australian bass	*Macquaria novemaculeata*
Freshwater/Brackish	Northern Pike	*Esox Lucius*
Freshwater/Brackish	Siberian Sturgeon	*Acipenser baerii*
Freshwater/Brackish	Common Bream	*Abramis brama*
Freshwater/Brackish	Northern Pike	*Esox Lucius*
Freshwater/Brackish	Australian bass	*Macquaria novemaculeata*
Catadromous	European silver eel	*Anguilla anguilla*
Catadromous	Japanese sea bass	*Lateorabrax japonicus*
Catadromous	American eel	*Anguilla rostrata*
Catadromous	Longfin eel	*Anguilla dieffenbachii*
Catadromous	Shortfin eel	*Anguilla australis*
Catadromous	European silver eel	*Anguilla anguilla*
Catadromous	Grey mullet	*Liza aurata*
Catadromous	Tupong	*Pseudaphritis urvillii*
Catadromous	American eel	*Anguilla rostrata*

Table B1.2: Papers removed from the dataset as they did not tag fish.

Paper	Category	Name		Reason Removed:
Edwards et al. 2007	Teleost	Gulf sturgeon	*Acipenser oxyrinchus desotoi*	Review of other studies, so no tags.
Wright et al. 2007	Teleost	Coho salmon	*Oncorhynchus kisutch*	They only tag seals
Watson et al. (b) 2009	Teleost	Horseshoe crab	*Limulus polyphemus*	Not a Teleost
James-Pirri 2010	Teleost	Horseshoe crab	*Limulus polyphemus*	Not a Teleost
Schaller et al. 2010	Teleost	Horseshoe crab	*Limulus polyphemus*	Not a Teleost
Watson and Chabot 2010	Teleost	Horseshoe crab	*Limulus polyphemus*	Not a Teleost
Cooke et al. 2011	Teleost	Atlantic sturgeon	*Acipenser oxyrinchus*	Review, so no tags
Lee et al. 2011	Teleost	Lingcod	*Ophiodon elongatus*	Paper not on lingcod
Wuneschel et al. 2013	Teleost	Weakfish	*Cynoscion regalis*	Based on fishery trawling data- no tagging
Wuneschel et al. 2013	Teleost	Striped bass	*Morone saxatilis*	Based on fishery trawling data-no tagging
Wuneschel et al. 2013	Teleost	Summer flounder	*Paralichthys dentatus*	Paper does not tag any fish

This resulted in the final dataset of 729 studies from 584 publications was complete.
FishBase (2017) was then used to allocate a habitat to all species based on the following criteria;

"Habitat- Indicates the particular environment preferred by the species, with the following choices (adapted from Holthus and Maragos 1995):

• pelagic: occurring mainly in the water column between 0 and 200 m, not feeding on benthic organisms;

• benthopelagic: living and/or feeding on or near the bottom, as well as in midwater, between 0 and 200 m;

- demersal: living and/or feeding on or near the bottom, between 0 and 200 m;

- reef-associated: living and/or feeding on or near reefs, between 0 and 200 m;

- bathypelagic: occurring mainly in open water below 200 m, not feeding on benthic organisms; and

- bathydemersal: living and/or feeding on or near the bottom, below 200 m."

There were only three bathypelagic studies on two species, the sharp-tail mola (*Masturus lanceolatus*) and opah (*Lampris guttatus*), and both are listed in the publications as occurring above 200m depth so there were included in the pelagic category. In a similar manner, only sixgill sharks (*Hexanchus griseus*) were included in the bathydemersal category and were listed in the publication as occurring in much shallower water so were included in the demersal category.

Lastly, the demersal category was split by habitat type into the following categories; soft sediment associated (n = 51), generalist (n = 76) or reef associated (these 11 papers were placed into the main reef associated category). Category was designated firstly by manually checking the publication, if not defined in the publication then the detailed ecology section of 1) FishBase (2017) and 2) the ICUN red list (2017) was consulted.

The final classification by habitat was pelagic (n = 226), benthopelagic (n = 175), reef associated (201), demersal soft sediment associated (n =51) and demersal generalist (n = 76).

Systematic Review Methods References

Froese, R. and Pauly, D. Editors. (2017). FishBase. World Wide Web electronic publication. www.fishbase.org. 1/03/2016.

IUCN. (2017). The IUCN Red List of Threatened Species. Version 2017-1. www.iucnredlist.org. 12/6/2016.

Kessel, S. T. (2015). Location-referenced Acoustic and Satellite Telemetry Papers (Version 1) [Data set]. Ocean Tracking Network (OTN). https://doi.org/10.14286/20150203/kessel

Systematic Review Final Publication List

The full list of 584 publication can be downloaded at:

Fetterplace, L. C. (2018): Acoustic and Satellite Telemetry Studies on Marine Fish: 1986–2013. figshare. Fileset. https://doi.org/10.6084/m9.figshare.6275312