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Assessment of the Sydney offshore artificial reef

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This report is based upon a series of research projects undertaken by staff at NSW Department of Primary Industries (DPI) as well as staff and students at University of New South Wales (UNSW). Scientists who were responsible for undertaken the primary research presented in Chapters 2 – 4 are listed under the title of that section and are acknowledged as the primary authors of that body of work.

A large number of people assisted in various aspects of the research projects presented in this report. Details of the assistance provided can be seen in each of the research papers listed in Appendix 1.

Non-technical summary

Assessment of the Sydney offshore artificial reef

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Objectives

Fisheries enhancement is a core element of the Fisheries NSW strategic plan, and central to the enhancement program is the deployment of artificial reefs. Global reviews of the success and failures of previous artificial reef deployments highlight the need for a thorough pre-deployment planning process, including the development of clear goals, followed by a long-term post-deployment monitoring period, spanning multiple years. This allows for a robust evaluation of the artificial reef deployment to be made, as well as providing data and information which may be fed into the development and deployment of future artificial reefs.

In collaboration with UNSW, the Department of Primary Industries (DPI) undertook both a detailed planning procedure, and a number of research and monitoring projects after the Sydney offshore artificial reef (hereafter referred to as OAR) was deployed in October 2011. Each of these research projects contributed to our understanding of the broader process in planning and monitoring artificial reefs. The objective of this report is to bring both the planning and post deployment research projects together, highlight the procedures which were undertaken by DPI, and detail the collective findings of the various research projects.

Key words

Offshore artificial reef; fisheries enhancement; reef fish; pelagic fish; productivity

Summary

On the 12th October 2011, NSW’s first purpose built offshore artificial reef was deployed off the coastline of Sydney. The primary goals of the Sydney OAR were to 1) improve offshore recreational fishing opportunities by creating new fish habitat, and 2) providing an additional fishing location. Prior to the deployment, a detailed environmental assessment provided the basis for approval of the project, including a thorough site selection process, ensuring the OAR was located in an optimum position, while having minimal effect on the existing environment or impacting on a variety of stakeholders.

The continued deployment of OARs along the NSW coast requires a significant level of planning, evaluation and assessment before any structure is lowered into the water. This consists of a constraint mapping exercise to identify suitable locations for deployment, a risk assessment and an environmental assessment (EA). Major constraints which limited the scope of potential sites included depth, exclusion zones (e.g. commercial shipping lanes, ACMA cable exclusion zones, marine protected areas), location of suitable substrata to support the structure, distribution of habitats, avoidance of locations supporting threatened species, and other coastal and

oceanographic processes. The constraint mapping process led to the identification of two potential locations in the Sydney region, one off Barrenjoey Headland and a second off South Head. Ultimately, the location off South Head was selected. The EA supported the constraint mapping exercise and contained a detailed assessment of the potential impacts of the OAR during both the short term construction phase, and the longer term operational phase. Impacts identified by the EA were broad and included noise and waste created during the construction, effects of the structure on seafloor characteristics, how the structure may influence local flora and fauna, its impact on the local fishing industry, interactions with threatened species and how it may affect the local oceanographic processes. The EA also outlined the scope for a long-term monitoring plan which formed the basis for much of the research and monitoring that eventuated in the period 2011 – 2014. A strategy was proposed in the EA which involved a multi-disciplinary approach to monitoring various aspects of the OAR. To assist in prioritising the research and monitoring conducted in light of funding constraints, the series of proposed monitoring objectives were prioritised. Most of the high priority monitoring objectives were undertaken and form the various data chapters of this report. These monitoring programs also allowed for a robust evaluation of the success of the OAR in meeting its original deployment goals. Following the deployment of the OAR, a number of research and monitoring projects was initiated covering a range of areas including; 1) the long term response of reef associated fish; 2) the response of pelagic fishes; 3) residency of fishes and their connectivity with nearby natural reefs; 4) angler participation rates; and 5) trophic pathways for fish production on the OAR.

Response of the reef fish assemblage

One principal monitoring objective was to assess the long term response of reef associated fish to the deployment of the OAR. This fish assemblage was compared to nearby natural reefs with sampling conducted on a monthly basis over four years following the OAR deployment. Stereo-baited remote underwater video (stereo-BRUV) was used to sample the fish assemblage and obtain length estimates of abundant species. A total of 53 species were observed, however the fish assemblage at the OAR showed distinct inter-annual variation, this was not observed at the natural reefs. This likely due to successional processes on and around the structure, such patterns have been observed in long-term monitoring programs at other artificial reefs around the world. Although the fish assemblage at the OAR was consistently changing, it remained distinct from the assemblages at the natural reefs and is likely due to the differences in physical structure between the OAR and natural reefs. Importantly, numerous species of recreational value were observed at the OAR, and length measurements indicated many of these were above minimal legal length and available for harvest by recreational anglers.

Distribution of the pelagic fish assemblage around the OAR

Pelagic baited remote underwater video was deployed in the waters around the OAR to assess the response of non-resident and pelagic species and their spatial association with the structure itself. Fish were generally only associated with the OAR on a small spatial scale, with abundances of fish rapidly declining at distances only 10’s of meters from the structure. This close association suggests there may be some trade-off between predation risk and foraging success. The composition of the assemblage itself was diverse and included coastal pelagic species (e.g. *Trachurus. novaezealandae*, *Seriola lalandi*), oceanic (e.g. *Isurus oxyrinchus*, *Makaira indica*), reef associated (*Pseudocarnx georgianus*, *Nelusetta ayraudi*) and soft bottom benthic species (*Platycephalus caeruleopunctatus*). The close spatial association of fish with the structure has implications for the design of future reef deployments which now include deploying clusters of concrete reef modules, thereby creating a ‘reef field’. These modules are positioned with a high degree of precision, with a key consideration of their layout being the distance between the concrete modules themselves. Findings from this study suggest reef units as close as 60 m will avoid overlapping distributions of associated fish, while promoting connectivity.

Multispecies residency and connectivity around the OAR

Connectivity between designed artificial and natural reefs is central to understanding the effect of an OAR within a coastal ecosystem. The location, size and proximity of designed artificial reefs to other reefs can facilitate the dispersion and recruitment of species to newly deployed artificial reefs. Residency, connectivity and general movement patterns of Eastern Fiddler Ray (*Trygonorrhina fasciata*), Bluespotted Flathead (*Platycephalus caeruleopunctatus*), and Port Jackson Shark (*Heterodontus portusjacksoni*), in association with the Sydney OAR were examined using acoustic telemetry. The three species showed varying levels of residency at the OAR including strong variation between individuals of the same species, particularly for Eastern Fiddler Ray. The site of tagging influenced the residency of all three species, where the proportion of days spent at either the OAR or Dunbar (a nearby natural reef site) was highest by individuals tagged from those reefs. Connectivity was evident between the OAR and nearby natural reefs, with all species exhibiting movements >5 km from their tagging reefs and visiting up to 5 or 6 other reef areas during the monitoring period. Despite this, most individuals remained within 2 km from their tagging reef (either the OAR or Dunbar).

Monitoring boat-based recreational fishing effort at the Sydney OAR

A key goal in the deployment of many artificial reefs is to provide enhanced fishing opportunities for recreational anglers. Using a shore-based camera, the recreational boat-based fishing effort at the OAR was estimated. Fishing effort was compared between June 2012 – May 2013 (year 1) and June 2013 – May 2014 (year 2). This time frame was further stratified into seasons. It was estimated there was 1765 fisher hours during year 1 and 2460 hours during year 2. There was some seasonal differences in fishing effort, for example, fishing effort in spring of year 2 was greater than during spring of year 1. All other pairwise comparisons of seasons between years showed no differences in fishing effort. This study demonstrated that shore-based camera systems are effective for monitoring changes in recreational fishing effort at near-shore artificial reefs. The seasonal pattern of fishing effort observed during the two-year survey period was influenced by both the length of time since the Sydney OAR was deployed, and more general seasonal patterns in fishing activity which is observed by fishers in the general region of NSW. Effort intensity recorded at the Sydney OAR was 31,525 and 44,116 fisher hours per square kilometre for years 1 and 2 respectively. This level of usage was up to 12 times greater than that recorded from many estuarine fisheries in NSW. Monitoring of recreational fishing effort at future artificial reefs will form a central aspect of any evaluation process and camera based technologies provide a cost effective solution to monitor these fisheries.

Zooplanktivory as a pathway for fish production on the OAR

Zooplanktivores can be extremely abundant on artificial reefs and their capacity to continuously access zooplankton supplied by prevailing currents highlights that the provision artificial reef habitat may allow for increased production of zooplanktivorous fishes. Mado (*Atypichthys strigatus*) is among the most abundant species found at the OAR, evaluation of its diet revealed 93% consisted of zooplankton. The density of *A. strigatus* around the reef was estimated from camera deployments and their food consumption was then calculated. The supply of zooplankton to the OAR was estimated using plankton tows. A numerical model was developed to assess the depletion of zooplankton caused by *A. strigatus* predation at the Sydney OAR and predicted the depletion of various size classes of zooplankton. A general model was then developed to identify how the size of the Sydney OAR influences the availability of food relative to the availability of habitat for reef-resident zooplanktivorous fish such as *A. strigatus*. Despite an estimated 3800 *A. strigatus* with a biomass of 130 kg populating the Sydney OAR, their total consumption depleted less than 0.5% of the prevailing supply of zooplankton. Given this, it might seem logical that increasing the size of the OAR would support a greater biomass of zooplanktivores, ultimately leading to greater benefits for recreational fishers. However, this study shows that doing so would provide more refuge volume for zooplanktivores, but relatively less foraging volume to support the increased consumption by a larger population, ultimately limiting its size. This suggests that an optimum reef size exists that can successfully trade-off between food and refuge. Understanding the limitations of artificial reefs that are either too small or overly large is essential for designing reefs that effectively facilitate the important trophic link between zooplankton and reef-resident fishes. This is important as larger reefs cost more to construct, yet may not optimise the transfer of energy from zooplankton to reef-resident zooplanktivorous fishes.

Findings from the work presented in this report indicate the Sydney OAR successfully met the original goals of the deployment. This is supported by a number of findings, including that fish rapidly colonised the structure, with this community undergoing change over time as expected. Importantly the community included species of key recreational importance, with length data revealing many were at lengths allowing harvest by anglers. Participation rates by recreational anglers appear to indicate the structure is now a popular fishing location for boat based fishers in Sydney. Additionally, modelling has identified trophic pathways on the reef and shown how the design of reefs can facilitate a balance between foraging and refuge for reef associated species. Collectively this information is important for the future planning, deployment and monitoring of artificial reefs and demonstrated the considerable value of a thorough planning phase, followed by a long term comprehensive monitoring program.

Introduction

History of artificial reef development

The construction of artificial reefs to enhance fishing opportunities has a long history extending back to the 18th century, and has been particularly prolific in South-East Asia , North America and more recently in Europe (Baine 2001; Bohnsack and Sutherland 1985). Artificial reefs were initially constructed from a wide variety of materials including car tyres, trams, aircraft, decommissioned ships and oil platforms among others, and are collectively referred to as 'materials of opportunity' (Brickhill *et al.* 2005; Krohling *et al.* 2006). These materials were cheap and readily available and often had a perceived added advantage of being a novel and environmentally beneficial way of dealing with unwanted waste (Pollard 1989).

Waste materials however, are not normally designed to persist in aquatic environments, and there is a history of the materials not only breaking down and therefore no longer providing the structure for which they were deployed, but also leaching pollutants into the surrounding waters (Collins *et al.* 2002; Kellison and Sedberry 1998). Problems with early artificial reefs were further compounded by a lack of clear objectives and often little monitoring pre- and post-reef deployment (Svane and Petersen 2001; Wilding and Sayer 2002). As such, the outcomes in terms of fisheries enhancement were difficult to determine. Ecological research of fish on natural reefs showed clear patterns of increasing fish diversity and abundance with increasing complexity of reef structure (Roberts and Ormond 1987; Robertson and Sheldon 1979). This led to a shift in the use of design specific materials which are structurally complex, including internal spaces, which did not pose any environmental risk and were often constructed from concrete or steel and are described as 'purpose built reefs' (Kellison and Sedberry 1998; Sherman *et al.* 2002). Along with a shift in the design of reefs, came an increasing awareness of the importance of the location and configuration of the reef, and clear objectives which could be evaluated with a rigorous monitoring program (Seaman 2000; Strelcheck *et al.* 2005; Svane and Petersen 2001). It is now acknowledged that successful artificial reef projects should be able to demonstrate the project meets pre-deployment goals through the evaluation of ecological, physical and socio-economic variables (Folpp 2012). This includes making comparisons to nearby natural reefs, comparisons with reference sites, as well as suitable temporal 'before and after' monitoring (Fabi and Fiorentini 1994; Lowry *et al.* 2014)

Initially, artificial reefs were perceived as fish aggregation structures, which would increase fishing revenue (Santos and Monteiro 2007). However it quickly became evident that simply attracting fish to a location only increases fishing efficiency and has no long term fisheries enhancement, due to local population loss. It was acknowledged for any artificial reef to have any real fisheries enhancement capability, it must increase local productivity in order to augment natural fish production and thereby support local fisheries (Osenberg *et al.* 2002). The issue of 'attraction vs production' has remained prominent in the development of artificial reef design, management and evaluation to this day (Bohnsack 1989). There are now increasing signs that artificial reefs which are well planned and implemented, do increase local productivity and enhance, rather than attract fish populations (Johnson *et al.* 1994; Smith *et al.* 2016).

Within an Australian context, the development of artificial reefs follows a similar trend to that which has been observed on a global scale. Early reefs employed 'materials of opportunity', for example, car tyres were regularly used along relatively low energy coastlines which contained little natural structure such as Port Phillip Bay in Victoria and Gulf St Vincent and Spencer Gulf in South Australia (Branden *et al.* 1994; McGlennon and Branden 1994; Pollard 1989). As awareness of the negative environmental impacts of waste material grew, deployments and research into artificial reefs stalled during the mid-1980's. Like in other parts of the world, deployments of artificial reefs generally lacked any clearly defined goals or monitoring regimes, and so in addition, it was not possible to demonstrate these reefs were actually enhancing local fish productivity.

Recent initiatives for artificial reefs in NSW and the development of the Sydney offshore artificial reef

The introduction in NSW of recreational fishing licence fees in 2001 generated revenue for recreational fishing enhancement initiatives (Lowry *et al.* 2010). In 2004, NSW DPI gained funding through this revenue stream for the deployment and monitoring of artificial reefs. Following a significant consultation process, a number of large estuaries along the NSW coast were chosen as sites for the new reefs (Lowry, Folpp *et al.* 2010). These systems had recently been declared recreational fishing havens, where commercial fishing was no longer permitted. Clear qualitative objectives for the program were outlined and were complimented by a number of quantitative measures which were closely monitored, including fish abundance, size and community composition, as well as changes in the benthic community. The monitoring program itself consisted of a combination of baited remote underwater video (BRUV) and SCUBA diver surveys of the fish community, photographic surveys of the benthic community, and independent angling surveys to determine the utility of the reef (Lowry, Folpp *et al.* 2010). Surveys were undertaken prior to the deployment of the reef and included comparisons with nearby natural reef and bare sand habitat. The reefs consisted of Mini-Bay Reef Balls® and were deployed at sites specifically selected following a mapping exercise to identify the most suitable locations. The well planned and conducted approach taken to the deployment of artificial reefs in NSW estuaries demonstrated the benefit of these reefs through sustained recruitment of species which were highly regarded among recreational anglers (Lowry, Glasby *et al.* 2014). This four year program established a foundation for future projects in refining the planning and construction, and also the evaluation of ecological, physical and socio-economic factors associated with artificial reef systems.

Following the success of the estuarine artificial reef program, NSW DPI began planning the deployment of an offshore artificial reef which was recognised by the then Ministerial Advisory Council for Recreational Fishing (ACoRF) and the recreational fishing community as a high priority. Funding was secured from the Recreational Fishing Saltwater Expenditure Committee to investigate the potential for the establishment of three OARs along the NSW coast in shallow waters accessible by trailer-boat recreational anglers.

Sydney Offshore artificial reef design and installation

The offshore artificial reef (OAR) unit design is 12 m x 16 m x 12 m (height x length x width) with the bulk of the internal structure in the lower 4 m (Figure 1). The OAR unit is manufactured from square hollow sections, rectangular hollow sections and plates, and weighs approximately 42 tonnes (dry weight). Four concrete anchor blocks are connected to each corner to ensure stability of the OAR.

The OAR unit has design certification to withstand a 1/100 year storm event (a wave height of approximately 18 m – H_{Max}) and will have an operational lifespan of 30 years. The deployment site is approximately 1.9 km (1.3 nm) south-east of South Head (Sydney Harbour) at a depth of 38 m.

The OAR was lowered into position on the morning of 12th October 2011 (Figure 2), this was followed by the attachment of moorings and inspection by divers prior to commissioning on the 13 October 2011.

Figure 1 Design of the Sydney offshore artificial reef showing the four concrete anchor blocks, twin towers and main structure.

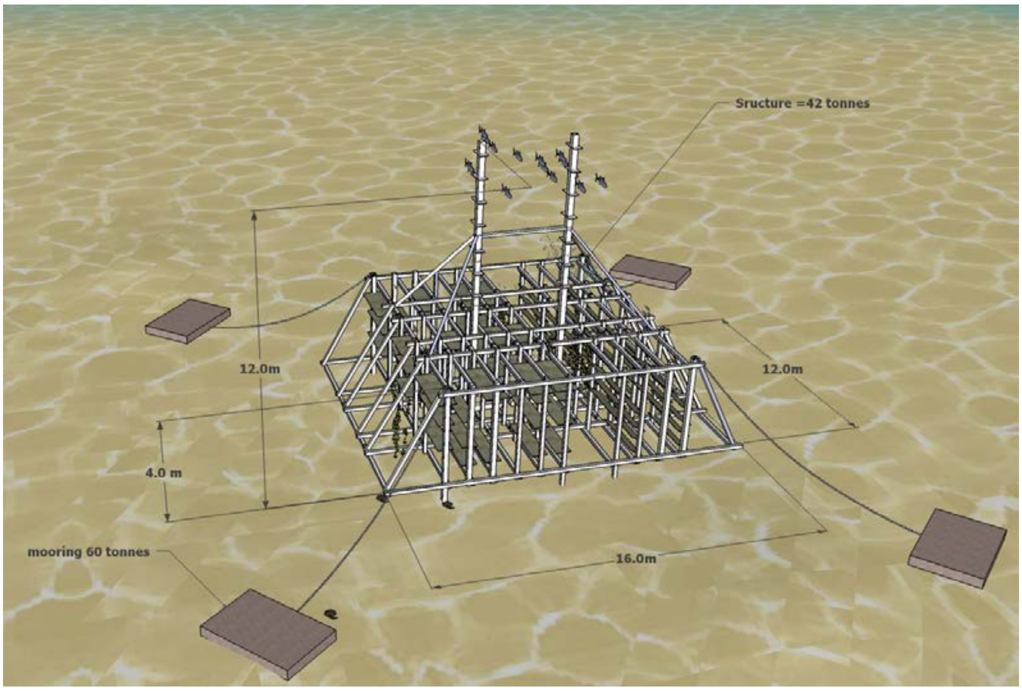


Figure 2 Deploying the Sydney offshore artificial reef on the 12th October 2011.

Chapter 1 – New South Wales offshore artificial reefs – constraint mapping, risk assessment and final environmental assessment

Introduction

The deployment of OARs along the NSW coast by NSW DPI requires a significant level of planning, consultation, evaluation and assessment before the structure is deployed. Statutory approvals, licences and permits are spread across State and Commonwealth agencies and are consistent with number of international agreements including the London Convention, which was principally set up for the prevention of marine pollution by dumping of wastes. Constraint mapping is designed to identify suitable locations in which the OAR can be placed by layering areas which preclude their installation. A multidisciplinary Risk Assessment framework is used to focus the aims of the environmental assessment (EA) that forms the backbone of the proposal and subsequent application for project approval. Before deployment of the Sydney OAR, a constraint mapping and risk assessment analysis was conducted by Cardno consultants, this together formed part of the broader Environmental Assessment (EA). An Environmental Management and Monitoring Plan (EMMP) was also included in the EA and listed monitoring priorities based on key risks identified by the EA, most of which were subsequently undertaken and the outcomes are detailed in chapters of this report.

In the initial phase of the OAR program, three potential regions were considered for placement of New South Wales's first offshore artificial reef, and included the metropolitan areas of Sydney, Newcastle and Wollongong. Initially, one location was to be selected and subsequent deployments at the remaining locations would be dependent upon the outcome of the first, and funding availability. While Sydney was selected as the region for the first OAR, detailed planning was carried out for all three regions. This chapter details the process and major findings only for the Sydney OAR, with the aim of highlighting the planning, evaluation and assessment process required prior to lowering structures into the water in a responsible approach to OAR deployments and ongoing management.

Constraint Mapping

The locations of artificial reefs are subject to numerous constraints which will limit the positions in which a reef may be placed. Poorly positioned artificial reefs may have negative impacts, such as becoming a navigation hazard or cause environmental damage by modifying coastal processes. Identifying these constraints is an important initial step in the site selection process, as it eliminates particular sections of coast and provides a focus to areas in which more detailed evaluation and consultation can be undertaken.

In 2007, at the beginning of the process a number of major constraints that restricted the potential positioning of the artificial reef (and are also relative to any future reef deployments) were identified prior to more detailed investigations. The proposed region considered for the placement of the Sydney OAR ranged from Barrenjoey Head to Sydney Harbour (Figure 3 & Figure 4). The major constraints considered in the site selection process are as follows:

- <u>Depth</u>:
  As the tower structures of the OAR reach 12 m in height, a depth range between 32 m – 40 m based on the lowest astrological tide (LAT) on the continental shelf was a major requirement to avoid creating a navigational hazard, with a minimum of 20 metres LAT considered as appropriate based on the potential depth draw of large coastal vessels. Suitable depth is also important for the stability of the structure (in terms of ability to withstand certain hydrodynamic forces of sliding and overturning), accessibility to recreational fishers (via boat) and will also influence the type of fish which will aggregate around the structure.

The depth contours in the Sydney region do not strictly follow the shape of the coastline, particularly in the region around South Head (Figure 4). In some sections of the Sydney region there are areas of rocky outcrops where the seabed profile is relatively steep. Generally, due to the slope of the seafloor, depth was not a major limiting factor in the constraint mapping process for the Sydney OAR.

- Exclusion zones, including:
  - ACMA cable exclusion zones
  - Deepwater ocean outfalls
  - Recognised commercial fishing areas (e.g. trawl and trap)
  - Designated commercial shipping lanes/port restrictions
  - Marine protected areas (state and commonwealth)
  - Mining exploration leases
  - Historical ship wrecks
  - Spoil grounds

There are two large ACMA (Australian Communications and Media Authority) cable protection zones along the Sydney coast, one off the northern beaches at Narrabeen (Figure 3) and the second around Tamarama and Clovelly Beaches (Figure 4). Each of these zones represented large areas which could not be considered for the OAR placement. Three deep-water ocean outfalls are located at North Head, Bondi and Malabar and extend to depths between 60 – 80 m (3 – 4 km offshore) and also represented constraints.

There are a number of state fisheries which operate within the Sydney region consisting of the Ocean Trap and Line Fishery, NSW Southern Fish Trawl and Lobster Fishery. The lobster fishery predominantly operates in waters greater than 100 m within the Sydney region, so was not considered to be a major constraint given the targeted depths for the OAR. The NSW Southern Fish Trawl Fishery operates within both the northern and southern regions proposed for the Sydney OAR. Purse seining for Eastern sea garfish (*Hyporhamphus australis*), yellowtail scad and blue mackerel is conducted in the northern sector of the proposed region.

Two large commercial ports in Botany Bay and Sydney Harbour result in high volumes of commercial shipping traffic within the proposed region, including container carriers, car carriers, and general cargo and passenger cruise ships. This is an important consideration when investigating safe clearance depths as discussed above. Bulk liquid carriers are generally restricted to only the port in Botany Bay. Both ports have limits which extend in a radius 4 nm out to sea (Figure 3 & Figure 4). Commercial ships are prohibited from anchoring in these limits. Initial consultation with Sydney harbour masters indicated they would consider the OAR to be placed within these limits with a preference to the more extreme angles from which shipping rarely approaches the port.

There are no Marine Protected Areas (MPAs) within the proposed region, however there are 10 Aquatic Reserves, which are managed by DPI and are designed to protect aquatic biodiversity, protected species, populations and ecological communities (Table 1).

Table 1 List of Aquatic Reserves within the Sydney region proposed for the OAR.

Location	Area (km^2)
Barrenjoey Head	0.3
Narrabeen Head	0.1
Long Reef	0.8
Cabbage Tree Bay	0.2
North Sydney Harbour	2.6
Bronte – Coogee	0.4
Cape Banks	0.2
Towra Point	14
Boat Harbour (Kurnell)	0.7
Shiprock (Port Hacking)	0.02

Most of these Aquatic Reserves are close to the shoreline (within 100 m of the mean low water mark), because of this, these reserves posed no major constraint to the OAR position as their waters were too shallow.

There is a Petroleum Exploration Licence which covers the NSW coastline extending from Port Stephens to Wollongong, however this only extends from 3 nm offshore. In the northern region, sand is permitted to be dredged from the mouth of Narrabeen Lagoon to replace sand lost from Collaroy beach due to coastal processes. This is only completed periodically and was not considered to be a major constraint due to its location and depth.

There are 105 known shipwrecks within the Sydney region. Of these however, only a few occur in depths suitable for the OAR deployment. Furthermore, some of these wrecks also occur in ACMA cable zones, so these areas were already not considered as suitable locations. Approximately 10 wrecks were identified as posing constraints to the OAR position. There are also a number of wrecks for which no accurate positional information is available and future analysis to identify these locations would be valuable.

A large spoil ground is located south of Port Jackson, offshore from Macquarie Lighthouse, it is in 80 m of water and was not considered a major constraining feature. There are smaller spoil grounds in the northern area off Curl Curl Head, between 35 – 50 m of water and another just off North Head which needed to be avoided.

- Other restricting features:
  - E.g. FADs (Fish Attraction Devices), desalinisation plants

There are five FADs offshore from Sydney, four of these are positioned in water considerably deeper than was proposed for the Sydney OAR. However, one is located closer to shore off Sydney Harbour in 50 m of water, which did fall within the depth range of the OAR. There are also wave rider buoys located off the northern metropolitan region however these are at depths of approximately 100 m.

The Kurnell desalinisation plant on the southern headland between Cape Solander and Cape Bailey has an exclusion zone around the water intakes and outlets (Figure 4), eliminating these areas for consideration.

- Distribution of substrata and suitability of substrata to support artificial reef structures:

  Artificial reefs are typically designed to maintain their structural and functional integrity for many years without deteriorating or being permanently covered by sediment. Underlying sediments need to have the ability to easily support the reef structure and generally soft sediments such as clays, silts and loosely packed sands should be avoided. Reliable and detailed seabed characterisation data are therefore necessary to properly site the OAR.

Detailed benthic maps of the Sydney region were sourced from existing charts and digitised and included information on the distribution of various sized sediments, natural rocky reef and artificial reef (e.g. shipwrecks). Additionally, multibeam swath mapping in proposed areas was also undertaken to confirm depth and slope, while inferring sediment characteristics and sediment depth. The location of rocky reef was considered a significant constraining feature to the location of the Sydney OAR. This is because a key goal of the OAR program is to provide recreational fishing opportunities in new areas, natural rocky reef is perceived to already provide these services, and therefore the Sydney OAR was to be positioned away from these areas. Furthermore, rocky reef is unlikely to provide a stable flat platform on which to deploy the structure and the actual deployment itself would likely damage existing fish habitat. The coast off Sydney consists of large areas of rocky reef, although it is generally patchily distributed and often separated by large areas of coarse grain bare sand (Figure 3 & Figure 4). In particular the area between South Head and Ben Buckler was identified as an area in which rocky reef was confined to a narrow strip along the shoreline cliffs (Figure 4).

- Distribution of habitats, flora and fauna:

  The locations of OARs need to avoid existing reef habitats, habitats unique within an area, or locations known to support diverse benthic/epibenthic communities. Areas to be avoided should also include beds of macroalgae, oyster reefs and mussel beds. Habitats critical to the survival of a particular species are generally protected under NSW legislation and therefore must be avoided. It should also be considered that protected habitats may require an additional buffer zone around them where fishing or development activity is restricted. In addition, it has been specified by DPI that placement of any OARs should be at least 500 m away from existing reefs in order to create new habitats where there was originally none, rather than adding to existing fishing areas.

Identification on the distribution of habitats, flora and fauna relied on previous research which found the sandy habitats off Sydney support diverse and abundant assemblages of fish and invertebrates. Bare sandy communities are known to be resilient to disturbance (e.g. deployment of an OAR) and capable of rapid recolonisation. Communities show zonation in regards to depth, with kelp and turfing macroalgae found in shallow waters and sponge gardens in deeper waters (Underwood *et al.* 1991). Ideally the OAR was to be positioned on bare sandy substrate, disturbing only benthic invertebrates which would recover quickly due to their ability to rapidly recolonise areas.

- Threatened species issues, including fish, invertebrates, marine mammals and marine reptiles:

  Information on the occurrence and distribution of threatened species is generally sparse and may be limited to predictions based on presence of suitable habitat and/or records of a species occurring at near-by locations.

Within NSW, threatened species and communities are protected under three legislative acts which include the *Fisheries Management Act*, *Threatened Species Conservation Act* and the *Environment Protection and Biodiversity Conservation Act*. There are 15 endangered, vulnerable or protected species known to occur in the Sydney region which includes three species of marine reptile, seven marine mammals and five species of fish.

Grey nurse sharks (*Carcharias taurus*) is listed as critically endangered and has been observed off the Sydney coast. Aggregations of grey nurse sharks have been mapped (Otway *et al.* 2003; Otway and Parker 2000) which show that South Maroubra (Magic Point) was a significant aggregation site believed to contain 3.5% of the NSW population. A 200 m area of critical habitat was declared off Magic Point and was mapped as a potential constraint. An endangered population of the Little Penguin (*Eudyptula minor*) is known to occur just north of Smedley's Point near Manly, and is the only known breeding population on mainland NSW. There is a 50 m restriction zone extending from the shore at this location, however the depths were too shallow to consider placing an OAR and it was no constraint.

Migrating mammals are known to occur in the Sydney region. In particular, numbers of the Southern Right Whale (*Eubalaena australis*) are known to be increasing in shallow inshore waters off the NSW coast (Allen and Bejder 2003) and Humpback Whales (*Megaptera novaeangliae*) migrate along the NSW coast. Calving Southern Right females remain close to the shore within a depth range between 5 – 10 m. It was recommended placement of the OAR avoid these depth ranges to minimise interactions between the structure and Southern Right Whales.

- <u>Coastal and Oceanographic processes</u>

  Artificial structures placed on the ocean floor are subject to the forces of currents (variable in speed and direction), waves, tides and hydrostatic changes in water levels. Combined, such currents can produce stresses on the seafloor and artificial reef structures (Sheng 2000). Excessive physical forces may cause erosion of benthic sediments, leading to instability or movement of the reef structures. Spatial information on near-shore hydrodynamics is therefore essential in order to avoid potentially high energy areas with bottom stresses unsuitable for reef placement.

A number of processes cause coastal circulation in the Sydney region including the East Australian Current (EAC), coastal trapped waves, winds, internal waves and outflows from large estuarine systems. These are dynamic forces and show high temporal variation (Middleton *et al.* 1997). A typical EAC cannot be defined, although it generally breaks away from coast at Seal Rocks. This can result in warm or cold core eddies breaking from the main current in a southward direction. The EAC is generally strongest during summer. Trapped coastal waves generally move northwards from Bass Strait and impact the Sydney region about 2 days after being generated by strong winds. Tidal outflows from major estuaries in the region (Hawksbury River, Sydney Harbour, Botany Bay and Port Hacking) may also have significant effects on current patterns.

The wave climate across the Sydney region can be considered similar. Long term data from 'Waverider' buoys located in 80 m of water off the Sydney coast shows the average H_s (wave height trough to crest) is approximately 1.6 m and storm conditions with H_s greater than 4.5 m occur less than 1% of the time. The predominant wave direction offshore of the Sydney region is SSE, with 31% of the swell coming from this direction, 19% of the swell comes from the S and 16% from the NE (Kulmar *et al.* 2005). Waves also generate orbital velocities and currents in the waters below, resulting in complex strong horizontal currents. The potential strength of these currents and its effects on the OAR were evaluated in an independent study commissioned by NSW DPI but not detailed in this report.

Sediment transport generally occurs along the coast between breaking waves and the coast. However, during storms cross shore movement can occur and is often referred to as storm erosion, shifting sand from the beaches to deeper offshore areas. In depths between 25 – 40 m sediment movement is small and unlikely to bury or scour large structures such as an OAR.

Taking information regarding major coastal processes, a number of recommendations were made regarding the position of the Sydney OAR, namely that depths between 25 – 40 m would be generally favoured because:

- the artificial reef would be less likely to affect wave refraction patterns and the like;
- wave-driven currents are smaller and therefore so are dynamic forces on the structure;
- navigation is less likely to be restricted
- oceanographic currents are likely to be smaller
- sediment transport is likely to be negligible

Outcomes of the constraint mapping process in the Sydney region

Having taken into consideration all the previously listed constraints, two potential locations within the Sydney region were proposed. The first was in the northern metropolitan region in an area extending from Barrenjoey Headland south to Palm Beach (Figure 3). This location covered an area of 6 km^2 and was selected due to the lack of a network of rocky reefs which characterise this section of the coastline. The location also contained suitable substrate and was not constrained by the presence of any shipwrecks. The second location was in the southern metropolitan region offshore of South Head and Ben Buckler (Figure 4). This location was only considered following approval from Sydney Port Authority, as it is located within the Sydney Harbour Port Limits. This location lacked any rocky reef other than a thin strip of reef extending seaward of the shoreline cliffs. Ultimately, this southern location off South Head was selected as the optimum location, and the OAR now sits at 33° 50.79’ S, 151° 17.98’ E in 38 m of water.

Figure 3 **Major constraining features in the northern section (Barrenjoey to Port Jackson) of the Sydney region. Sub regions are marked by red dotted boundaries.**

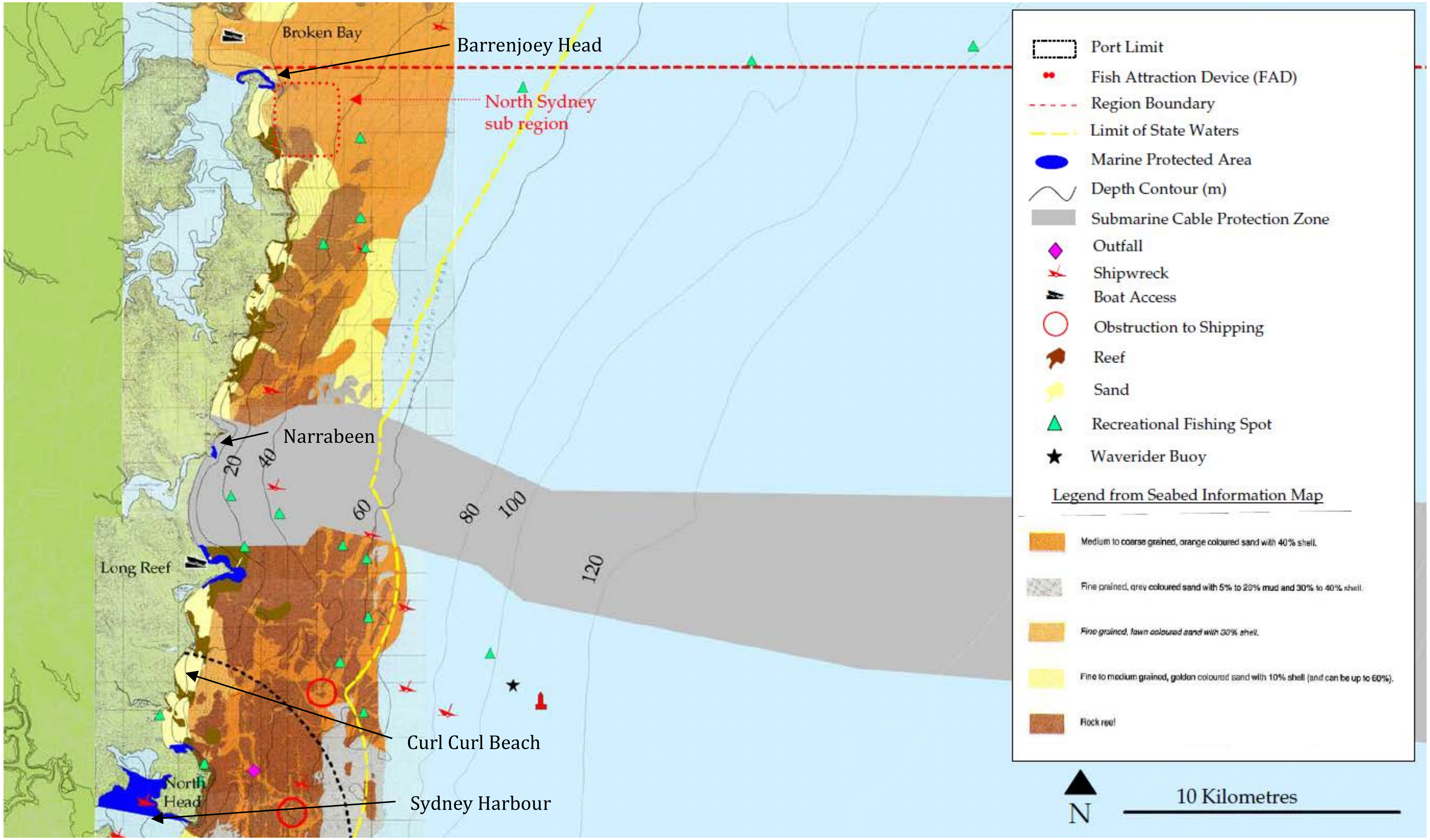
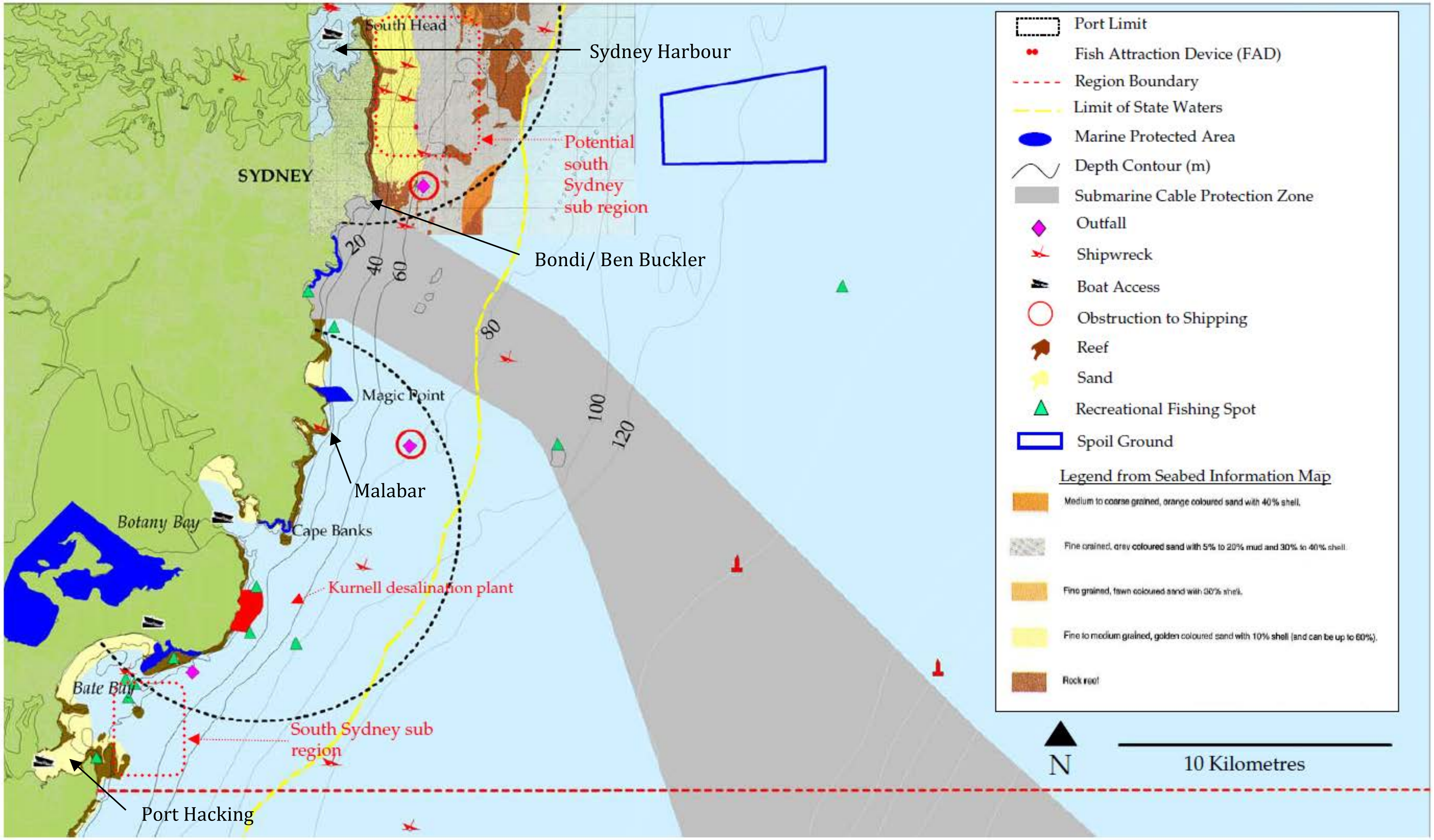


Figure 4 **Major constraining features in the southern section (Port Jackson to Bundeena) of the Sydney region. Sub regions are marked by red dotted boundaries.**



Risk Analysis

The risk analysis formed the basis for the EA and considered potential impacts relating to coastal processes and oceanography, ecosystem processes, contamination, climate change, recreational and commercial fisheries and interference with existing coastal infrastructure, obstructions and exclusion zones. The constraints mapping process described earlier in this chapter was designed to minimise or eliminate a number of potential risks by selecting proposed areas away from factors such as port restrictions, spoil grounds, shipwrecks and deep-water outfalls among others. This process also minimised risks associated with protected species, critical habitats and marine protected areas and identified areas with stable substrates.

Following a risk assessment workshop that included industry and related experts from outside DPI, a total 50 receptors were identified in a risk assessment (Figure 5). These were broken down into a spatial scale, with risk being either small (< 1 km) or large (> 10 km). Participants identified most of the receptors as “low significance” (shown in green). Within the category ‘Coastal Infrastructure’ a high significance assessment was given for the stability of the structure (Figure 5). It should be pointed out that assessment was made prior to the development of final designs for the OAR, and that these designs largely overcame any of these potential risks. Stability assessments showed there was a low risk of the structure either sliding or overturning during extreme weather events.

Many of the receptors in the ‘Ecosystem Processes’ group were identified as likely to have a consequences of medium significance but only at small spatial scales (< 1 km). In particular local changes to fish and invertebrate communities and some habitat disturbances as well as potential interaction with threatened species (Figure 5). It was proposed at the time that long-term monitoring work would help to understand exactly how the deployment of the Sydney OAR would impact on these communities.

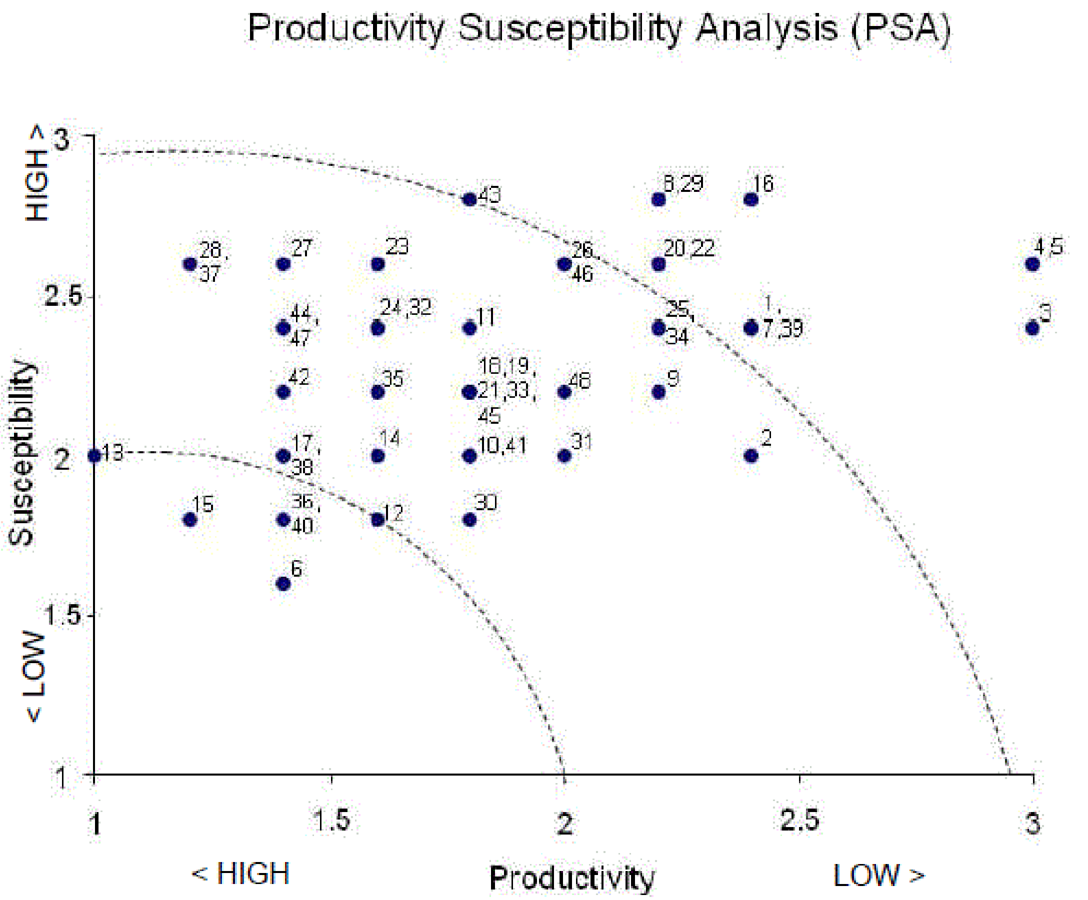
Figure 5 Results of risk assessment workshop for long term OAR deployments. Orange cells = High significance; yellow cells = moderate significance; green cells = low significance. L = likelihood of risk occurring, C = consequence of risk occurring.

Receptor/Issue	Hazard/Negative Impact	Scale: Small (≤1 km) L	Small (≤1 km) C	Large (1-10 km) L	Large (1-10 km) C
1. Coastal Processes & Oceanography:					
Nearshore coastal	Inshore wave climate	D	5	D	4
	Change to beach erosion/deposition	N/A	N/A	D	3
Local Seabed	Local scouring/deposition around units	A	5	N/A	N/A
2. Structural Integrity and Stability					
	Loss of structural integrity e.g. from corrosion or excessive marine growth	C	2	N/A	N/A
	Risk of sliding or overturning	E	3	N/A	N/A
3. Flora and Fauna					
Benthos	Direct loss of habitat	A	4	E	4
	Change to sedimentary characteristics	A	4	B	4
	Changes to benthic assemblages	A	4	C	4
	Increased predation by fishes from the OAR on benthos	A	4	C	4
	Commercial trawling in areas not previously trawled	B	5	E	4
Proximal natural reef	Changes to benthic assemblages	C	3	B	3
	Change to fish assemblages	B	3	D	3
Pelagic Environment					
Plankton	Concentration of plankton	B	5	E	5
Fish	Loss of habitat (benthic species)	A	5	E	5
	Attraction/aggregation	A	4	B	4
	Increased fishing effort	B	4	C	4
	Increased mortality (from aggregation)	A	3	B	3
Threatened and protected species					
Fish	Incidental capture	B	2	C	2
	Aggregation of threatened or protected species	B	2	C	2
	Harm from marine debris and pollution (KTPs)	C	2	C	2
	Interruption of movement corridors (e.g. GNS)	D	3	D	3
	Increased predation	C	4	D	4
	Loss of habitat	A	5	E	5
	Introduction of harmful marine pests	D	3	D	3
Marine Turtles	Harm from marine debris and pollution (KTPs)	C	2	C	2
	Incidental capture	C	2	C	2
	Boat strike	C	2	C	2
	Acoustic disturbance	C	5	E	5
	Interruption of movement corridors	D	3	D	3
	Loss of habitat	D	4	E	4
	Introduction of harmful marine pests	E	2	E	2
Cetaceans	Harm from marine debris and pollution (KTPs)	C	2	C	2
	Incidental capture	C	2	C	2
	Boat strike	C	2	C	2
	Acoustic disturbance	B	4	B	4
	Interruption of movement corridors	C	4	C	4
	Loss of habitat	E	2	E	2
	Introduction of harmful marine pests	E	2	E	2
Pinnipeds and Sirenians	Harm from marine debris and pollution (KTPs)	C	2	C	2
	Incidental capture	D	2	D	2
	Boat strike (sirenians only)	D	2	D	2
	Acoustic disturbance	D	4	E	4
	Interruption of movement corridors	E	2	E	2
	Loss of habitat	E	2	E	2
	Introduction of harmful marine pests	E	2	E	2
Seabirds	Harm from marine debris and pollution (KTPs)	C	2	C	2
	Incidental capture	C	2	C	2
Invasive Marine Pests					
	Introduction of invasive marine pests	C	4	E	3
The Commonwealth Marine Environment					
	Impact on and activities within Commonwealth Marine	N/A	N/A	E	3
Areas of Conservation Significance	Impacts on Nature Reserves	C	2	C	2
	Impacts on Aquatic Reserves	E	2	E	2
	Impacts on Critical Habitats	E	2	E	2
4. Recreational and Commercial fishing					
	Loss of commercial fishing ground	A	4	N/A	N/A
	Conflict between user groups	A	4	A	4
	Risk OAR does not achieve goals	D	3	D	3
	Gear hook-up	A	4	N/A	N/A
	Collision from crowding	C	3*	E	3*
	Impacts on commercial fish stocks	E	3	E	2
	Increased encounters with dangerous marine animals	C	5*	E	5*
	Injury or drowning (spear fishing)	C	1*	N/A	N/A
5. Sediments and Water					
	Leaching of contaminants	A	4	E	4
6. Navigation and Safety					
	Clearance	B	2	B	2
	Anchor fouling	B	3	N/A	N/A
	Increased vessel traffic				
7. Infrastructure					
	Conjestion/crowding at boat ramps	B	5	B	5
	Lack of amenties	D	5	D	5
8. Heritage					
	Impacts on historic shipwrecks	E	4	N/A	N/A
	Impacts on submerged aboriginal deposits	E	4	E	4
	Conflict with areas of spiritual significance/dreamings	B	4	D	3
	Negative impacts on aesthetic amenity	C	4	C	4
9. Occupational Health and Safety					
	Injury or loss of life during diver maintenance or monitoring inspections	C	1*	C	1*

Aggregation of fish at the Sydney OAR is a central issue, and as a primary goal of the OAR was to increase fishing opportunities, fishing mortality must also occur. The key to the success of artificial reefs is to balance this fishing mortality with increased fish production. The risk assessment also included a Productivity Susceptibility Analysis (PSA). A PSA is commonly used in fisheries as part of a process to determine how vulnerable different species, communities or components of habitats are to impacts from certain fisheries, and to assess the sustainability of a fishery. The PSA approach assumed that vulnerability depends both on the susceptibility of a species to capture on the OAR, and the productivity of the species, as this will determine the rate at which it can recover from any fishing mortality. Susceptibility was considered to depend upon behavioural factors such as the attraction towards artificial structures, site fidelity and depth range. Other factors taken into consideration included whether species are recreationally or commercially important, and the exploitation status of a species. Productivity depends on life history traits, with long lived, slow growing species with low fecundity likely to have low productivity.

The PSA was conducted on 48 species which were considered most likely to occur at the proposed OAR location (Figure 6). For each species, criteria was ranked 1 – 3, with 1 indicating low susceptibility and high productivity (low risk) and 3 indicating high susceptibility and low productivity (high risk). The average productivity and susceptibility scores for each of the 48 species were then plotted on a 2D PSA graph (Figure 6). High risk species included Wobbegong Sharks (*Orectolobus* spp), Eastern Shovelnose Ray (*Aptychotrema rostrate*) Long-fin Pike (*Dinolestes lewini*), Sergeant Baker (*Aulopus purpurissatus*), Silver Trevally (*Pseudocaranx georianus*), Kingfish (*Seriola lalandi*), Mulloway (*Argyrosomus japonicus*) and Moray Eel (*Gymnothorax prasinus*). This classification resulted from these species either having low fecundity or being a key targeted recreational species. The risk analysis recommended these species be considered a high priority for future monitoring and management of the artificial reef. It was also noted that some of these species in the high risk group are territorial reef species (e.g. Blue Grouper *Achoerodus viridis*) and only likely to be at high risk if drawn away from natural reef to the Sydney OAR. Locating the structure at least 500 m from any natural reef would minimise the likelihood of this occurring. Species with low risk mostly included those associated with sandy habitat and high productivity such as flatheads (Platycephalidae) and small pelagics such as herrings, sardines and pilchards (Clupeids). It was determined there would be some loss of habitat for some sand dwelling species caused by the placement of the unit, but the effect of this would be insignificant in relation to the amount of similar habitat nearby.

Figure 6 **Productivity Susceptibility Analysis of recreationally and commercially important species likely to occur at the Sydney OAR.**



1	Shovelnose ray	17	School whiting	33	Blue morwong
2	Fiddler ray	18	Tailor	34	Red morwong
3	Spotted wobbegong	19	Cobia	35	Bastard trumpeter
4	Banded wobbegong	20	Silver trevally	36	Sand mullet
5	Ornate wobbegong	21	Yellow-tail scad	37	Striped sea pike
6	Herrings, sardines, pilchards	22	Kingfish	38	Maori wrasse
7	Moray eel	23	Common dolphinfish	39	Blue grouper
8	Sargeant baker	24	Australian salmon	40	Blue mackerel
9	Nannygai	25	Snapper	41	Australian bonito
10	John dory	26	Tarwhine	42	Striped marlin
11	Red rock cod	27	Yellow-fin bream	43	Sawtail
12	Eastern blue-spotted flathead	28	Silver biddy	44	Yellow-finned leatherjacket
13	Long-spine flathead	29	Mulloway	45	Chinaman leatherjacket
14	Dusky flathead	30	Luderick	46	Six-spined leatherjacket
15	Tiger flathead	31	Black drummer	47	Southern calamari
16	long-fin pike	32	Silver sweep	48	Arrow squid

Environmental Assessment

Overview

Deployment of OARs in New South Wales requires approval under statutory requirements of both State and Commonwealth legislation including the then enacted Part 3A of the NSW *Environmental Planning and Assessment Act 1979* (EP&A Act), the Commonwealth *Environmental Protection and Biodiversity Conservation Act* 1999 (EPBC Act) and the *Commonwealth Environmental Protection (Sea Dumping) Act* 1981. Guidelines were issued for the deployments of OARs to meet both State and Commonwealth legislation, it was proposed that all these requirements be met in a single Environmental Assessment (EA) document. The EA also provided the opportunity to present the views of various stakeholder groups, provided background regarding the need for the project, and some basic designs for potential reefs as well as a review of existing information on artificial reefs.

The EA provided a detailed outline of the existing environment in the Sydney region in which the OAR was to be placed. Much of this information was collected during the desktop constraints mapping process and included assessments on topics such as the distribution of existing flora and fauna, coastal processes (e.g. wave climate, currents, sediment transport) and existing levels and locations of commercial and recreational fishing effort. Importantly, the EA built upon this existing information by also identifying gaps in information and included additional studies in an effort to bridge these gaps. For example an Acoustic Doppler Current Profiler (ADCP) was deployed to study current flows. Detailed seabed mapping was undertaken covering 4.36 km^2 at proposed OAR locations using sidescan sonar, providing detailed bathymetry, backscatter, slope and aspect data, and flora and fauna surveys, such as BRUV (Baited Remote Underwater Video) deployments were also conducted.

In addition to providing an outline of the existing environment, the EA contained detailed assessment of the potential impacts of the Sydney OAR as identified in the risk analysis workshop both during the short term construction phase, and the longer term operational phase. In response to the identification of these potential environmental impacts, the EA outlined a series of recommendations to mitigation strategies. The EA also outlined the scope for a long term monitoring plan which formed the basis for much of the research and monitoring that eventuated from 2011 – 2014. A summary of the assessment of impacts, recommendations and mitigation and the proposed environmental management and monitoring plan (EMMP) is provided in the following sections.

Assessment of impacts

The environmental impacts of the Sydney OAR were broken down into the construction, transport and deployment phase, and then the long term operation phase. During the construction phase, impacts included factors such as noise pollution from the actual construction of the structure due to the use of power tools and generators, air pollution from dust emissions from the use of cutting and grinding tools, and waste materials generated during the manufacture. Other identified potential impacts included increased turbidity at the site during the actual deployment of the structure, the potential for migratory mammals to become entangled during the deployment, as well as the hazard the structure posed to the navigation of other vessels as it was towed to its location and lowered. While these issues were identified along with a number of minor others, the likely impact of each of these factors were all considered to be minimal.

Long term impacts were considered to be both broader as well as more likely than those during the initial construction and deployment phase. This covered potential local and broad scale changes to both the abiotic and biotic environments following the deployment of the Sydney OAR.

Large underwater structures have the potential to transform local wave patterns, particularly refraction, which in turn could affect nearshore coastal processes. This potential was examined in a mathematical modelling exercise (the SWAN model) comparing inshore coastal processes before and after the deployment of an artificial reef. This modelling work showed the deployment of reefs was unlikely to have any discernible influence on the inshore wave climate, this is most likely due to the relatively small size of the structure compared to typical offshore wave lengths experienced off the coast of NSW.

Placement of the structure on the seabed may lead to changes in the sediment conditions of the immediately surrounding area. Scouring is possible as currents are diverted and increase in speed as they pass around the structure. Modelling revealed the profile of the artificial reef itself could affect the level of sediment movement, but found that scouring would likely be less than 15 cm in even extreme current events, and under normal conditions, be unlikely to exceed 2 cm. This level of scour was considered minor and unlikely to undermine the stability of the structure.

The Sydney OAR was designed to have a lifespan of 30 years. The materials from which it was constructed were selected due to their ability to cope with extreme storm conditions (i.e. a 1 in 100 year storm event) and allow a rate of corrosion for similar marine structures under Australian standards. Also taken into account was the extra stress and weight due to the build-up of marine growth over 30 years. The risk of greater than expected marine growth, or faster than expected corrosion was considered, but this was deemed manageable with a suitable monitoring program.

The EA included a detailed investigation of the potential impacts of the Sydney OAR to local flora and fauna. The structure would sit and directly occupy an area of 180 m^2. Sites included in the constraint mapping consisted of bare sandy substrate. While it was acknowledged some loss of infaunal communities directly under the structure would occur, this was considered negligible when taken into context of the extensive areas of similar habitat in the direct and wider areas. It was considered the structure would result in changes to the sediment size characteristics in the immediate area around the structure and that this may result in changes to the infauna community structure. The increase in fish drawn to the reef was also considered to have the potential to alter infauna communities through increased predation. This effect is known as a 'feeding halo' and a monitoring plan outlined later in this chapter was designed to address this impact.

The deployment of the OAR has the potential to affect nearby natural reefs, although the impact of artificial reefs on nearby natural reefs is not well understood. The EA identified the possibility of predators, competitors and graziers drawn to the Sydney OAR, and the potential for them to then move to, and impact nearby natural reefs. Larval settlement process and the potential for the OAR to alter or interrupt these processes on natural reefs were also identified. Changes to recruitment processes are only evident over relatively long time periods, and it was acknowledged that even long term monitoring programs may not detect differences if they were to occur. The potential for 'draw down' of fish from natural reefs to an artificial reef has been well documented (Bortone 2006), with the assumption that fish are more susceptible to capture on artificial reefs. Recommendations in shifting the OAR location so that it is at least 500 m from any natural reef in order to reduce drawn down was made in the EA.

There has been a long running debate within the scientific literature as to whether artificial reefs increase regional productivity through the supply of increased habitat and food resources, or simply attract fish into a localised area where fishing mortality may be increased, this is sometimes referred to as the 'attraction vs production' debate. Although it is often pitched as a dichotomous debate, the reality is a gradient of effects, with evidence reefs increase productivity, but also, some fish species are attracted to the structure, suffering increased fishing mortality. There is growing evidence that productivity on artificial reefs can be considerable, placing them among the most productive systems in the world (Claisse *et al.* 2014; Smith, Lowry *et al.* 2016). The EA acknowledged the deployment of the OAR has potential to increase fishing mortality, particularly for recreationally important species, but also this fishing mortality may be balanced by an increased production of fish, indeed the design of the OAR incorporated aspects which would concentrate plankton to serve as a food source, essentially kick starting production on the structure. A balance between increased production and fishing mortality is a key goal for the deployment of artificial reefs.

The deployment of the Sydney OAR had the potential to affect threatened species, and this issue was raised in the EA. The constraint mapping process identified some areas which contain threatened species (e.g. grey nurse sharks) and so these areas were avoided. Because of this, the OAR was not considered to impact directly on any threatened species. However, the structure was identified as having a potential positive impact on threatened species by relieving fishing effort at natural reefs where threatened species are susceptible to hook and line fishing. Indirect effects from fishing activity such as the loss of fishing gear and harmful marine debris is considered a Key Threatening Process (KTP) in NSW under the *Threatened Species Conservation Act*. Some threatened species which transit past the OAR may be vulnerable to ingestion of these marine debris. The EA considered this to be a low risk but recommended regular monitoring of the structure, including underwater inspections to remove any fouled fishing gear, debris or litter.

The EA identified a number of potential impacts of the OAR on marine mammals. Increase boating activities places these animals at greater risk of boat strike. Additionally, the structure itself may hinder migration patterns for some species such as humpback whales (*Megaptera novaeangliae*). Existing restrictions on the distance boats must stay from whales was seen to minimise the risk of boat strike, but the EA encouraged the creation of mechanisms to the reporting of any threatened species including whales.

The OAR structure could provide a substratum or habitat suitable for invasive marine pests (also referred to as' introduced', 'alien' or 'non-indigenous' species). Of those species known to occur in NSW, the European Fan Worm (*Sabella spallanzani*) could potentially occur on the OAR. The species is known to inhabit depths to 30 m and colonize artificial structures such as marinas, submerged wrecks and pontoons. The species can compete with native species for food and space, thus inhibiting their settlement. Whilst the OAR is at risk from colonisation by invasive marine pests, the scale of the potential impacts is small and was considered unlikely to have any significant impact on the marine environment.

Both the Southern Fish Trawl and Ocean Trap and Line fisheries are potentially impacted by the deployment of artificial reef units in NSW through the loss of fishing grounds, as an area extending 100 m from the structure would need to be avoided by fishers to prevent losing fishing gear. Prior to the EA, considerable consultation with commercial fishing groups was undertaken and included in the constraint mapping exercise. Based on this, the EA considered the impact on commercial fishing not to be a significant issue.

The constraint mapping process used a combination of techniques such as sidescan sonar and towed video to identify the presence of anything with heritage value including shipwrecks within the vicinity of the proposed site as described in the constraint mapping section. These surveys did not detect any objects representing sites of cultural or heritage significance, so the EA concluded the Sydney OAR project was unlikely to have a negative impact in this regard.

Recommendations and Mitigation

The EA made a number of recommendations to mitigate some of the potential issues identified in the assessments of impacts. In the short term construction phase, most of these recommendations focused on reducing the impact of waste materials, including sediments and water, on the environment. It was recommended the contractor develop a detailed 'waste management plan' and 'water quality management plan' in accordance with existing State and Local Government guidelines. Regarding flora and fauna, it was also recommended that during the actual deployment, all activity should stop if there are reported sightings of migratory marine mammals.

For the long term operation of the structure, it was recommended it be placed at least 500 m from any existing natural reef, to minimise potential drawn down effects. User groups (i.e. recreational fishers) should be informed and educated on general saltwater fishing rules, approach distances to whales and mechanisms for reporting incidents of conflict.

Impacts on threatened and protected species were to be minimised by ensuring there are mechanisms to report the sighting or incidental capture of these species such as online forms, provide education on the identification of these species, as well as best practice for returning incidentally captured fish. The EA also recommended education in the prevention of spreading invasive marine pests and the potential impacts of harmful marine debris including the best ways to dispose of litter and discarded fishing gear.

In order to minimise any potential hazards to navigation and safety, a number of recommendations were made including reporting the coordinates of the structure to the Australian Hydrographic Office so they could prepare a ‘Notice to Mariners’ and amend official hydrographical charts. A minimum of 20 m clearance between the top of the structure is maintained at LAT to avoid collision with ships. A code of conduct and guidelines should be published to promote awareness of boating safety in the vicinity of the structure.

Environmental Management and Monitoring Plan (EMMP)

In order to better understand how the installation of OARs will impact upon significant components of the marine environment, evaluate mitigation strategies (e.g. waste/debris removal) and evaluate their overall effectiveness in relation to the project objectives, the EA identified a monitoring strategy was required. The construction contractor was also required to produce a contract specific Construction Environmental Management Plan which is not detailed in this report. The EMMP summarised reporting requirements to various consenting authorities including annual monitoring reports to the Department of the Environment and Energy and additional reporting requirements under the London Convention. It was noted that ideally monitoring would incorporate a multidisciplinary approach in order to understand how the deployment of the structure impacts upon significant components of the marine environment, and evaluate the overall effectiveness in relation to the project objectives. Furthermore, the EA suggested that sampling be conducted before and post deployment, in a sampling regime often referred to as ‘beyond BACI’ (Before, After, Control, Impact), incorporating multiple controls both spatially and temporally. In particular the EA noted the need for a long term sampling program spanning multiple years, to evaluate the effects on fish assemblages, long term seasonal variations and trends in relation to the age of the OAR as being crucial. Many previous monitoring programs evaluating the response of fishes have been conducted over insufficient time frames to fully detect responses, as these may take a number of years to eventuate. The EA also noted that any EMMP needed to include sampling conducted on a sufficiently regular basis so that the early dynamic stages of succession could be documented following deployment of the structure, with a maximum of 3 monthly intervals in the first 12 months of operation.

While the EMMP recommended a large suite of environmental and social factors be included both before and after the deployment of the structure, it did acknowledge funding and resources may not be available to cover all of these monitoring objectives. To assist in the most appropriate allocation of the funding and resources available, the EMMP developed a series of monitoring objectives listed below, which were determined as either high or low priority.

Biological Priority 1 (Monitoring objectives strongly recommended)

A) Investigate movements of high priority species within the direct study area
B) Assess effectiveness in terms of catch rates, species composition and fish stocks
C) Investigate occurrence of threatened/protected species on the OAR

Biological Priority 2 (Monitoring objectives to be addressed given available funds/resources)

A) Assess influence of the OAR on benthic assemblages (soft sediments) including potential halo effects
B) Assess influence of the OAR on benthic assemblages of proximal natural reefs (benthos)
C) Document colonisation of the reef structures by macroinvertebrates including pest species

Socio-Economic Priority 1

A) Assess effectiveness in terms of popularity with recreational fishing groups

Socio-Economic Priority 2

A) Identify issues of conflict between user groups

Physical Priority 1

A) Assess structural integrity
B) Remove fouled gear

Physical Priority 2

A) Assess influence of OAR on sediment characteristics
B) Assess concentration of heavy metals in adjacent sediments to the OAR
C) Assess water quality

In order to meet the monitoring objectives, the EMMP outlined a series of suggested methods, sampling frequency and sampling locations (Table 2). The original EMMP was comprehensive; Table 2 outlines only those sections which were directly relevant to priority 1 objectives.

Table 2 **Relevant sections of the OAR monitoring plan as suggested in the EMMP.**

Factor To Monitor	Objectives	Location	Monitoring Frequency	Method	Review Period	Management Procedures If Negative Impact Detected
Fish	Investigate the movements of high priority species within the wider study area	OAR, proximal natural reef sites (impact sites), and reference natural reef sites (controls)	At least every 3 months in the first 12 months post deployment, twice during winter and summer thereafter, allowing for seasonal comparisons	BRUVS, biotelemetry and visual diver census	Annual	• Determine acceptable level of impact: • Continue monitoring • Consider temporary closure and/or further monitoring • Limit to seasonal operation • Removal of structures
Fish	Assess effectiveness in terms of catch rates, species composition and fish stocks	OAR	At least every 3 months in the first 12 months post deployment, twice during winter and summer thereafter, allowing for seasonal comparisons	Stereo-BRUVS, visual diver surveys, on-site surveys and/or charter boat log book data	Annual	• Determine acceptable level of impact: • Continue monitoring • Consider temporary closure and/or further monitoring • Limit to seasonal operation • Removal of structures
Threatened Species	Investigate occurrence of threatened and/or protected species on the OAR	OAR	At least every 3 months in the first 12 months post deployment, twice during winter and summer thereafter, allowing for seasonal comparisons. Also continuous mechanism for feedback	Stereo-BRUVS, visual diver surveys and acoustic listening stations	Annual	• Determine acceptable level of impact: • Continue monitoring • Consider temporary closure and/or further monitoring • Limit to seasonal operation • Removal of structures
Recreational and Commercial Fisherman	Assess effectiveness in terms of popularity with recreational fishing groups	N/A		Stakeholder questionnaires	Annual	• Analyse feedback from user groups against project • Implement necessary changes • Consider temporary closure and/or further monitoring • Limit to seasonal operation • Removal of structures

Most of the priority 1 monitoring objectives were conducted by NSW DPI between 2011 and 2014, sometimes in collaboration with tertiary education facilities, generally following the methods and sampling frequency outlined in the EMMP. The outcomes of these monitoring and research programs are detailed in the following chapters of this report.

Chapter 2 – Monitoring the response of fishes and connectivity with natural reefs following the deployment of the Sydney OAR – Alistair Becker, Michael Lowry, Matthew Taylor, Iain Suthers, Molly Scott, Krystle Keller, James Smith

Introduction

The bulk of the monitoring work undertaken on the Sydney OAR was aimed at addressing the Biological Priority 1 objectives outlined in the EA (see Chapter 1). Primarily, this concerned evaluating the response of fishes to the deployment of the structure and determining how assemblages develop on the reef over time, their movements and interactions with nearby natural reef and sandy habitats, and ultimately if the reef is providing enhanced opportunities for recreational anglers. This was achieved through a number of complimentary monitoring and research programs undertaken by DPI staff, and scientists from the University of New South Wales. Essentially, this involved making observations of fish on and around the Sydney OAR using underwater cameras, and monitoring their movements around the structure and broader region using acoustic telemetry.

The goals of the Sydney OAR deployment were to create high relief, complex fish habitat which would provide quality fishing opportunities for anglers as well as providing additional fishing locations. An ability to demonstrate responses of fish assemblages to the deployment of artificial reefs is vital in any critical evaluation and was also a key requirement of the EMMP. Following the deployment of artificial reefs, fish assemblages undergo a long period of change prior to forming stable communities (Coll *et al.* 1998; Relini *et al.* 2002). Therefore a monitoring period extending across multiple years is required to properly assess fish responses. Unfortunately, monitoring programs running for long time frames are rare, which inhibits our understanding of the successional response of fish and the controlling ecological processes.

The need for comparisons between artificial reefs and control sites are also essential (Baine 2001). Historically, nearby natural reefs were used as comparisons as research often focused on the attraction vs production debate, and artificial reef construction also attempted to mimic natural reefs (Carr and Hixon 1997). Modern designs now differ considerably from natural reefs, but incorporate features (e.g. void spaces and towers) aimed at providing habitat for a range of species. Consequently, fish assemblages associated with artificial reefs may not directly mimic the assemblages found at natural reef control sites (Burt *et al.* 2009; Folpp *et al.* 2013; Thanner *et al.* 2006). In addition to making direct comparisons between artificial and natural reefs, connectivity between designed artificial and natural reefs is central to understanding the effect of a designed artificial reef within a coastal ecosystem. The location, size and proximity of designed artificial reefs to other reefs can facilitate the dispersion and recruitment of species to newly structures (Cenci *et al.* 2011), these can function as ecological stepping stones by increasing the connectivity between existing habitats. Connectivity can also contribute to the overall productivity of a designed artificial reef by increasing total fish abundance and biomass available for harvest (Koeck *et al.* 2013). Studying the movements of fish between artificial and natural habitats is therefore useful for assessing the suitability of artificial habitats, developing fisheries management zones (e.g. marine parks), as well as the potential for contribution to overall fisheries productivity.

While there has been much focus on the response of reef associated fish species to artificial reef deployments, larger modules are now deployed which can extend vertically 10s of meters from the seafloor. Due to their provision of habitat in the mid-water column, these reefs may also serve to provide resources for non-resident pelagic species (Scott *et al.* 2015). Pelagic species are often associated with remote structures such as fish aggregation devices (FADs), floating objects and isolated reefs (Hobday and Campbell 2009). Attraction towards such structures may be due to a number of ecological processes such as predator avoidance, school formation, feeding and parasite cleaning (Castro *et al.* 2001; Consoli *et al.* 2013). The specific process is likely to vary among species as well as determine the spatial scale at which fish may be associated with the OAR. Unique features of the Sydney OAR are the twin ‘towers’ which extend 12 m vertically into the water column, which were included to attract pelagic fishes.

Fish tend to show a close spatial association around OARs, with high abundances directly over the reef, with numbers rapidly declining 20 – 50 m away from the structure (Boswell *et al.* 2010; Stanley and Wilson 1997). Quantifying the distance at which fish interact with artificial reefs is essential for evaluating their effectiveness as fish habitat, or as targets for recreational fishing, and could be used to determine distances between reef units when designing future artificial reef fields (Jordan *et al.* 2005). Studies of the response of pelagic fish to artificial structures are rare, probably because they are difficult to conduct, yet can provide valuable additional information for reef managers.

The response of fish to the deployment of the Sydney OAR is outlined in this chapter which incorporates the findings of three studies. Each of these studies focused on different aspects of the community and behaviour of fish after the structure was in place. The design of these studies was largely based upon recommendations made in the EMMP described in Chapter 1. While the EMMP did recommend a BACI based design, this was not possible and biological sampling presented in this report was conducted only after the OAR was deployed. However, the combination of approaches adopted in conjunction with the length of the monitoring period of reef associated fishes makes the evaluation of the Sydney OAR both unique and comprehensive. Collectively this monitoring also provided evidence of the occurrence or interaction of the Sydney OAR with threatened and/or protected species through direct observations on camera deployments, or detections on acoustic receivers which was also a priority 1 objective detailed in the EMMP.

Long term monitoring of reef associated fishes

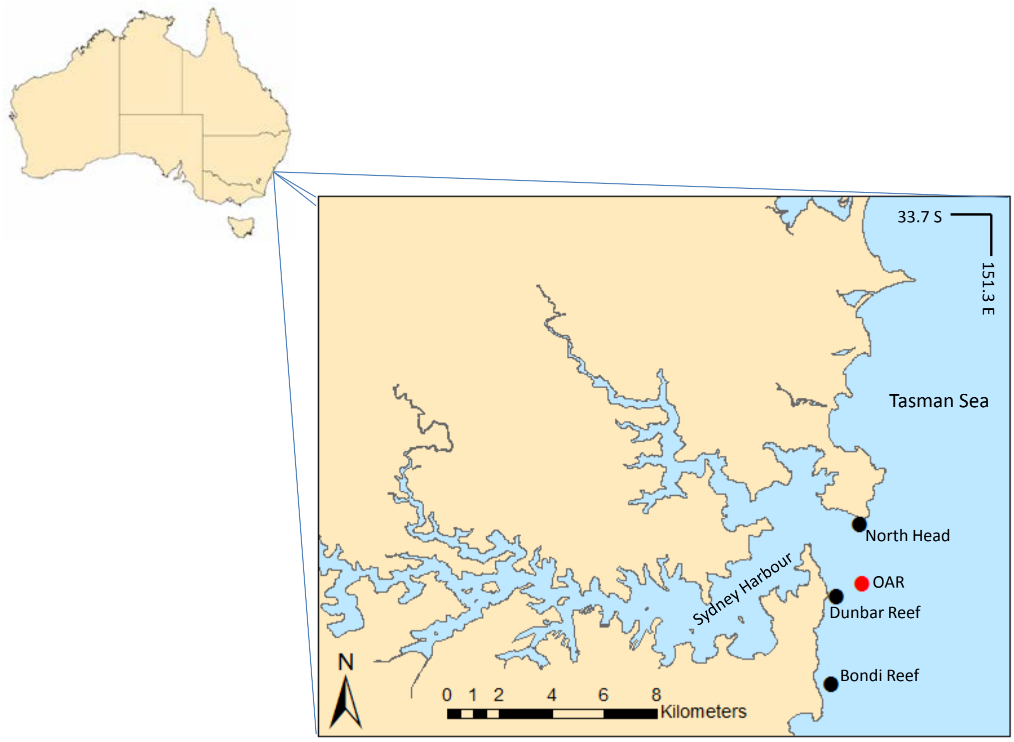
A key recommendation of the EMMP was the inclusion of a long term sampling regime to monitor the development of the fish assemblage on the OAR. Monitoring of fishes on artificial reefs for periods spanning more than 12 months is surprisingly rare, and the need for such studies has been identified in a number of global reviews on artificial reefs (e.g. Baine 2001). The primary aims of this aspect of the fish evaluation was to 1) examine the establishment and inter-annual variation of reef associated fish assemblages at the Sydney OAR compared to nearby natural reefs; 2) compare the length structures of key species between artificial and natural reefs.

Methods

Reef Associated Fish Assemblage

Between September 2011 and August 2014, the reef-associated fish assemblages at the OAR and three control reefs at North Head, Bondi and Dunbar (Figure 7) were surveyed using stereo baited remote underwater video (stereo-BRUV). Stereo-BRUVs provide accurate length measurements of fish (Harvey *et al.* 2004), and are well suited to collecting relative abundance and size data from deeper water reefs (Watson *et al.* 2009). At each location, stereo-BRUVs were deployed from a boat, with each deployment lasting 30 minutes following contact with the seafloor. The bait consisted of 500 g of *Sardinops sagax* (Pilchard) which was placed into a closed mesh bait container positioned centrally in the field of view, 1.2 m from the cameras. One deployment was made at the OAR and control reefs in a single day each month, with each deployment considered a replicate. The minimum distance between stereo-BRUV sites was 950 meters, at these distances over the deployment times used, no overlap of bait plumes occurred (Taylor *et al.* 2013), allowing multiple deployments in a single day that can be considered independent. Between 2011 and 2014, a total of 119 stereo-BRUV deployments were made at the four locations, some deployments were excluded due to equipment failure.

Figure 7 **Location of the Sydney Offshore Artificial Reef (Red Dot) and three control reefs (Bondi, Dunbar and North Head) off the coast of central Sydney.**



Video Analysis

Stereo-BRUV footage was analysed using the EventMeasure™ software (SeaGIS 2011). Relative abundance of fish was estimated using MaxN, which avoids repeated counts of fish by using the most number of individuals observed at any one time during a deployment (Cappo *et al.* 2004). A separate MaxN was generated for each fish species identified within a deployment. When a MaxN estimate was obtained, the fork length (FL) of each fish that contributed to the estimate was measured, this avoided generating length data that included multiple measurements of the same individual. For fish with truncate or round caudal fins, total length was measured.

Statistical Analysis

Reef associated fish assemblages sampled with the stereo-BRUV were analysed using Permutational Analysis of Variance (PERMANOVA) in the software package PRIMER v.6 (PRIMER E™, Plymouth, UK). Raw data was square-root transformed and from this a Bray-Curtis dissimilarity matrix constructed. An asymmetrical design described by Anderson *et al.* (2009) was set up and directly yields the correct test without the need to build it in steps. To conduct this test we used a fixed factor "Controls *verses* OAR" (*CvOAR*; 2 levels: OAR, Control), a nested random factor "Sites (Controls *verses* OAR)" (*Si(CvOAR)*; 3 reefs, Bondi, North Head and Dunbar were nested within the Control level, and 1 reef, OAR, nested within the OAR level). We also included the fixed and orthogonal factors "Season" (*Se*; 4 levels: Summer, Autumn, Winter, Spring) and "Year" (*Ye*; 4 levels: 2011, 2012, 2013, 2014). Pair-wise *a posteriori* comparisons were made for significant sources of variation. Where tests were based on low numbers of permutations, Monte-Carlo Pseudo-P values were used. To visualise patterns in the fish assemblages over time, principal coordinated ordination (PCO) analysis based on the yearly centroids for each reef was conducted, using square-root transformed, Bray-Curtis dissimilarity measures. PCO, otherwise known as metric multi-dimensional scaling, performs a principal coordinate analysis of any symmetric distance matrix.

For each year, fish diversity among reefs was compared based on Simpson's diversity index. As sampling was not equal across all sites the lowest number of samples collected for a site each year was used as the maximum level of replication, and the remaining data randomly subsampled to yield the same number of replicates (thus giving equal samples sizes between years). Simpson's diversity was then calculated using the `diversity` function in the Vegan package of R.

Four abundant species including Snapper (*Pagrus auratus)*, Silver Trevally (*Pseudocaranx georgianus*), Yellowtail Kingfish (*Seriola lalandi*) and Mado (*Atypichthys strigatus*) were common across all reefs, and length-frequency data were compiled by pooling across all years. Comparisons in length-frequency were made between fish observed at the OAR with the other three control sites. There were often limited length measurements from each control site, so length data was also pooled across these sites for comparisons with the OAR. Length-frequency samples were compared with Kolmogorov-Smirnov (KS) two-sample tests. We selected a bootstrap version which allows the test to be conducted with data that contains ties. The test was run using the `ks.boot` function with 100,000 simulations in the Matching package of R (Sekhon 2011).

Results

Inter-annual variation at the OAR and control sites

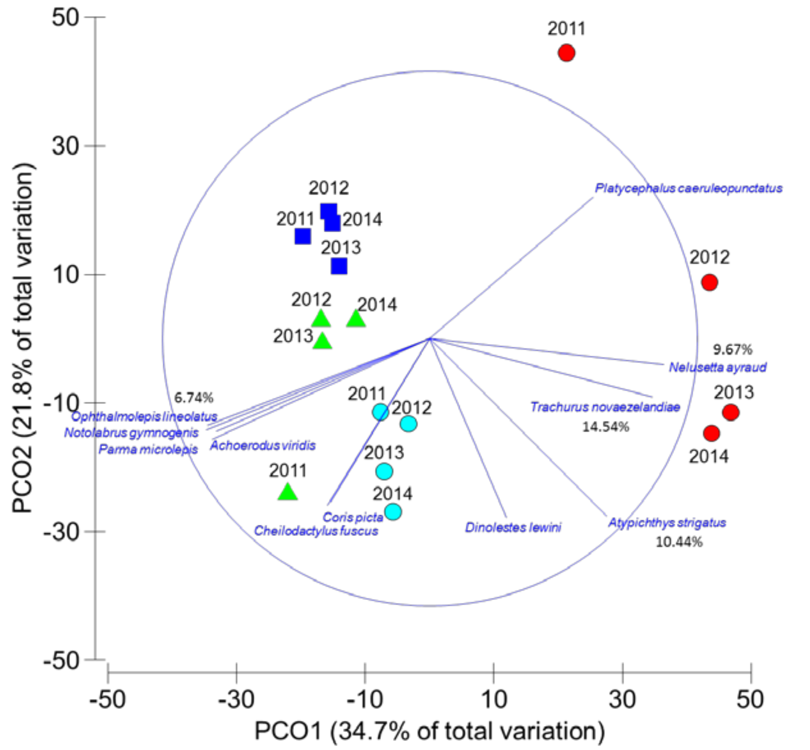
A total of 53 species of fish were recorded at the OAR during the sampling period, with new species recorded each year (Table 3). PERMANOVA analysis for fish assemblages observed in stereo-BRUV footage showed a significant interaction between sites, seasons and years (Table 4). Post-hoc analysis of this interaction showed the three control reefs did not differ among years or seasons. For the OAR, differences were detected between years, although these were not consistent across all seasons, and also between seasons, but only during 2013. A significant effect was also detected for the *CvOAR* term (Table 4) highlighting the differences between the three control reefs and the OAR. The PCO analysis based on the annual centroids for reefs, shows a pattern of distinct differences between the OAR and control reefs along PCO1 (Figure 8) and is consistent with the PERMANOVA analysis. SIMPER analysis showed that higher abundances of Yellowtail Scad (*Trachurus novaezelandiae*), *A. strigatus* and Ocean Jacket (*Nelusetta ayraud*) had the greatest contribution to differences between the OAR and control reefs (Figure 8). The pattern of yearly variation at the OAR which resulted in the significant interaction term (Table 4) is also evident along PCO2 with the OAR following a more linear trajectory (Figure 8). For 2011 to 2012, SIMPER analysis indicates differences w due to the addition of a number of new species to the OAR such as *A. strigatus*, *N. ayraud* and *P. auratus* (Table 5). While there is evidence of new species continuing to be observed at the OAR in the remaining two years, differences between these years were more influenced by fluctuations in the abundances of fish species already present on the structure. In particular, the abundance of *T. novaezelandiae* greatly increased between 2012 and 2013, while the abundance of *A. strigatus* increased from 2013 to 2014 (Table 5).

Table 3 **Presence of species observed in BRUV footage at the OAR across the sampling period.**

Species	2011	2012	2013	2014
Acanthopagrus australis			•	
Anoplocapros inermis		•		
Asymbolus analis		•		
Atypichthys strigatus		•	•	•
Aulopus purpurissatus		•	•	•
Austrolabrus maculatus		•		
Chaetodon guentheri			•	
Cheilodactylus fuscus			•	
Cheilodactylus vestitus				•
Coris picta				•
Dasyatis brevicaudata		•		
Dinolestes lewini			•	•
Enoplosus armatus				•
Eubalichthys mosaicus		•	•	
Gymnothorax prasinus			•	•
Heterodontus portusjacksoni	•	•	•	•
Hypoplectrodes maccullochi				•
Meuschenia flavolineata			•	
Meuschenia freycineti	•	•	•	•
Meuschenia scaber	•	•	•	•
Meuschenia trachylepis		•		
Myliobatis australis			•	
Nelusetta ayraud		•	•	•
Nelusetta ayraudi			•	•
Nemadactylus douglasi				•
Nemadactylus douglasii	•	•	•	•

Ophthalmolepis lineolatus			•	
Orectolobus maculatus			•	•
Orectolobus ornatus			•	
Pagrus auratus		•	•	•
Paracaesio xanthura				•
Parma microlepis				•
Parupeneus spilurus	•			
Platycephalus caeruleopunctatus	•	•	•	
Plotosus lineatus				•
Pseudocaranx dentex				•
Pseudocaranx georgianus	•	•		•
Rhabdosargus sarba		•		•
Scobinichthys granulatus			•	
Scorpaena jacksoniensis				•
Scorpis lineolata			•	•
Seriola lalandi		•	•	•
Trachurus novaezelandiae	•	•	•	•
Trygonorrhina fasciata	•	•	•	•
Upeneichthys lineatus	•	•	•	•
Zeus faber		•	•	

Figure 8 **Principal coordinates analysis showing centroids of reef sites for each year. ● = OAR, ● = Dunbar, ■ = North Head, ▲= Bondi. Vectors show Pearson correlations, a limit was set for variables with values ≤ r = 0.7, percentage values adjacent to species represent percent contribution to differences between control reefs and OAR from SIMPER analysis.**



Diversity of fish assemblages at the OAR was lower than each of the control reefs each year with the exception on Dunbar during 2012 (Figure 9). At the OAR, diversity showed the greatest variation with an increase during 2012, however it dropped again during 2013 and remained similar during 2014. Diversity was greatest each year at either North Head or Bondi (Figure 9).

Figure 9 **Simpson diversity index for each year at the OAR and three control reefs.**

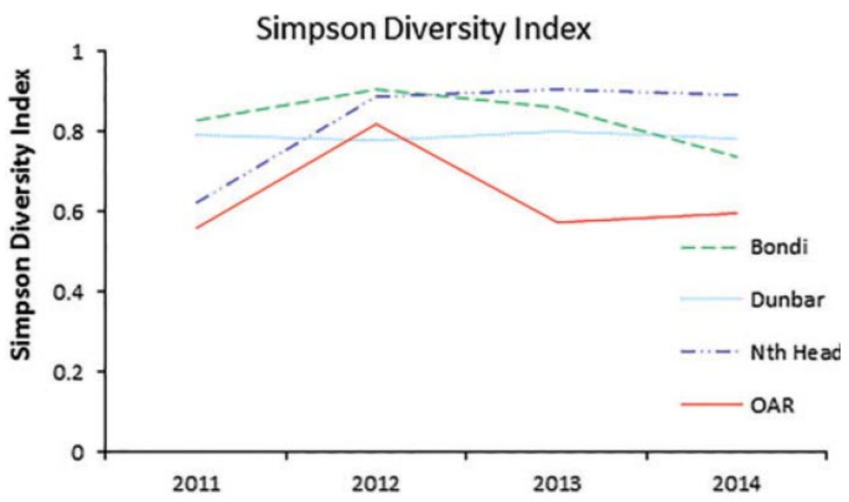


Table 4 **Results of permutational analysis of variance (PERMANOVA) of fish assemblages observed in stereo-BRUV footage. Significant sources of variation shown in bold.**

Source	df	MS	Pseudo-F	P (perm)
Controls vs OAR (CvOAR)	1	39726	2.974	**0.034**
Year (Ye)	3	4562.4	2.378	**0.039**
Season (Se)	3	2665.6	1.099	0.381
Sites(CvOAR) (Si(Cv))	2	11217	6.853	**0.001**
CvOAR x Ye	3	3292.8	1.728	0.179
CvOAR x Se	3	2281.2	0.947	0.524
Ye x Se	6	2962.5	1.419	0.154
Si(Cv) x Ye	6	1859.3	1.136	0.236
Si(Cv) x Se	6	2361.8	1.443	**0.017**
CvOAR x Ye x Se	6	2557.5	1.225	0.268
Si(Cv) x Ye x Se	10	2089.6	1.277	**0.042**
Res	65	1636.8		
Total	114			

Table 5 **SIMPER results from stereo-BRUV deployments showing species which contributed to changes in the benthic fish assemblage at the OAR between years. Measures are based on square root transformed data.**

Species	Mean Abundance (MaxN) 2011	Mean Abundance (MaxN) 2012	Average Dissimilarity	% Contribution
Atypichthys strigatus	0	4.79	12.76	21.37
Nelusetta ayraud	0	4.57	12.19	20.42
Pagrus auratus	0	3.08	8.22	13.77
Trachurus novaezelandiae	0.82	2.86	5.46	9.14
Trygonorrhina fasciata	0	1.18	3.15	5.28
Seriola lalandi	0	1.1	2.92	4.89
	Mean Abundance (MaxN) 2012	Mean Abundance (MaxN) 2013		
Trachurus novaezelandiae	2.86	11.94	13.84	34.12
Pseudocaranx georgianus	3.11	0	4.75	11.7
Pagrus auratus	3.08	1.14	2.96	7.3
Seriola lalandi	1.1	2.24	1.74	4.29
	Mean Abundance (MaxN) 2013	Mean Abundance (MaxN) 2014		
Atypichthys strigatus	5.28	9.3	5.17	19.76
Nelusetta ayraud	4.37	2.5	2.41	9.2
Plotosus lineatus	0	1.77	2.28	8.69
Pseudocaranx georgianus	0	1.06	1.37	5.22
Acanthopagrus australis	0.84	0	1.08	4.11

Pelagic Fish Response

In addition to reef associated species, non-resident pelagic species are also likely to be common in the water column surrounding the OAR. The spatial scale at which these fish are associated with the structure is likely to vary among species and may be influenced by abiotic factors linked to changing oceanic conditions. Specifically, this study aimed to: (1) test the effect of distance from the OAR on the abundance and composition of the fish assemblage; (2) determine the influence of oceanographic variation on the observed distribution of fish around the OAR; and (3) assess the efficacy of pelagic-BRUVs (PBRUV) as a sampling tool for these environments.

Methods

Sampling was conducted between October 2011 and March 2012, covering the Austral Spring and Summer, with sampling days haphazardly selected during this time. The timing of the commencement of fieldwork relative to the deployment of the reef was considered sufficient to allow recruitment and settlement of fish onto the OAR. Both PBRUV and unbaited camera drops were used in this study as they were determined to be the most suitable methods to collect data at the depths encountered, and the patchy distribution of pelagic fish. Each PBRUV contained a high-definition GoPro camera (Woodman Laboratories Inc., CA, USA) inside a waterproof casing, with a small bait canister mounted to the end of a PVC tube, 1m horizontally from the camera. Each PBRUV was suspended from a surface buoy, which was moored to the seafloor using an anchor (see Heagney *et al.* 2007). The PBRUV was designed to face downstream, enabling fish to be observed swimming up-current into the bait plume. The PBRUV was baited with 100 g of a mixture of minced pilchards, bread, and tuna oil, in an 8:1:1 ratio. The PBRUV could not be deployed directly above the OAR, so an unbaited drop camera was used (Seaviewer Sea-drop 650, FL, USA). The drop camera allowed species to be quantified around the two towers but not cryptic species inside the structure itself. However, this was not considered to hinder this study, as the aims were to assess the spatial scale at which a fish assemblage associated with the structure could be detected, rather than provide a census of the resident fish. A detailed study of reef associated resident fish is outlined in previous sections of this chapter.

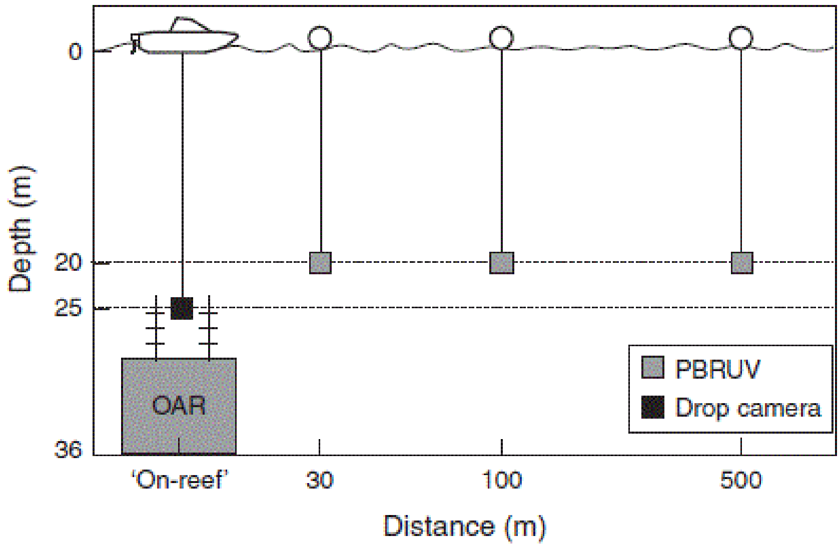
Experimental Design

Four distances from the OAR were sampled (Figure 10) which included ‘on reef’ (<5 m), ‘near’ (30 m), ‘mid’ (100 m) and ‘far’ (500 m). The ‘on reef’ camera was deployed down to a similar depth to the two towers (25 m). The PBRUVs were deployed at depths of 20 m, equal to the top of the structure. The order in which each of the distances were sampled on any given day was haphazard but the total number of deployments during a day varied between 4 and 6, mainly due to weather conditions. Based on previous research, deployments lasted for 45 mins as this was considered sufficient time to survey pelagic fish distribution and abundance (Heagney, Lynch *et al.* 2007).

All fish observed in the footage were identified to species and relative abundance (MaxN) was estimated for each species in each PBRUV and drop camera deployment. Summing these values together gives ‘total MaxN’. MaxN may not accurately estimate density, especially for baited cameras, as only a proportion of the fish that detect the bait plume may respond by moving up-current towards the food source; but, assuming this proportion is constant, MaxN is sufficient to enable a test of a treatment effect on relative abundance.

At the start of each deployment vertical water quality profiles were done using a SeabirdCTD™ (Sea-Bird Electronics, Inc., Washington, USA). Parameters measured included temperature, salinity, dissolved oxygen, fluorescence, pH and turbidity. Current speed was calculated using the Ocean Reference Station (ORS), managed by Integrated Marine Observing System (IMOS) which was moored in 65 m of water ~ 5 km SSE of the OAR.

Figure 10 **The sampling design. Pelagic baited remote underwater video (PBRUV) was deployed at three distances from the OAR, suspended from buoys to a depth of 20 m. The 'on-reef' distance was sampled using an unbaited drop camera deployed from a boat to a depth of 25 m. The PBRUVs were deployed shallower than the drop camera to remain >5m above the shallowest parts of the sampling area.**



Data Analysis

The ability of the PBRUV to attract and detect fish (i.e. effort) can be influenced by visibility and current velocity. Essentially, visibility limits the distance at which fish can be identified, thereby influencing the sampling effort, so effort can vary among deployments, requiring MaxN estimates to be standardised among deployments. Turbidity (ntu) was subsequently used as a surrogate for visibility. Currently velocity can also affect MaxN by altering the size of the bait plume, which influences the number of fish approaching the BRUV (Jones *et al.* 2003; Taylor, Baker *et al.* 2013). Bait plume area can be calculated as

$$A = \frac{(t\ x\ v)^2}{6}$$

Where *A* is the triangular bait plume area (m^2), *t* is the duration of the deployment (s), and *v* is the average current velocity during the deployment ($m\ s^{-1}$). Thus there were three estimates of relative abundance analysed: MaxN (raw abundance), MaxN ntu^{-1} (standardised to turbidity) and MaxN ntu^{-1}, m^{-2} (standardised to turbidity and bait plume area). MaxN ntu^{-1} was used for most of the analyses to account for visibility, and allowed the PBRUVs and drop cameras to be analysed together, while current speed was otherwise included as a covariate (Taylor, Baker *et al.* 2013).

Based on Bray-Curtis similarity measures, PERMANOVA was used to test the effect of distance on the fish assemblage for all three datasets (MaxN, MaxN ntu^{-1}, MaxN ntu^{-1} m^{-2}) to evaluate the consistency of the result. Where significant differences were observed, similarity percentages (SIMPER) was used to identify which species mostly contributed based on the MaxN ntu^{-1} dataset.

Analysis of variance (ANOVA) was used to test the effect of distance on total MaxN, and to test the association of the main species with the OAR. ANOVA was done using only the MaxN ntu^{-1} standardisation. All data was fourth-root transformed to improve normality and homogeneity of variance.

Analysis of the three datasets were completed twice, the first run of analyses included the three PBRUV distances (30m, 100 m and 500 m) as well as the drop camera ('on reef'), despite there being different methods used to collect data 'on reef' compared to the three PBRUV distances. The second run of analysis consisted of only the PBRUV derived data (i.e. the 30 m, 100 m and 500 m distances). As the on-reef sampling differs from the PBRUV method (unbaited and shorter duration), results including all four treatment distances should be treated with caution. It is, however, likely that the unbaited drop cameras underestimated abundance compared with the baited PBRUVs (Hardinge *et al.* 2013; Harvey *et al.* 2007) and the same could be said of their shorter duration. Thus, the difference in abundance between the on-reef distance and the other three distances is probably greater than reported in this study.

Results

Distribution of pelagic fish around the Sydney OAR

Of the 76 PBRUVs deployed during the project, fish were observed on 42 (55%) with an average time of 7.4 minutes before the first fish was observed. Fish <5 cm in length were too small to be positively identified, so these were grouped together as 'juvenile fish'. From the PBRUV deployments a total of 2768 individuals consisting of 14 taxa were observed, while 1547 individuals were observed in the drop camera, including 4 species which were not seen in the PBRUV footage (Table 6). The average MaxN for the on-reef deployments were greater by an order of magnitude than average MaxN values at each of the distances from the OAR (Table 6; Figure 11). Within the pelagic zone around OAR, *T. novaezelandiae* and *N. ayraudi* were by far the most abundant species (Table 6).

The PERMANOVA showed a significant effect of distance, when all four distances were analysed, for both MaxN and MaxN ntu^{-1} (

Table 7). There was no effect of distance for any data standardisation when on-reef data were removed. This shows that the fish assemblage within 5 m of the OAR ('on-reef') is significantly different from the surrounding pelagic assemblage. A SIMPER analysis of MaxN ntu^{-1} data reported that the most abundant species contributed most to the significant distance effect identified using PERMANOVA, with *N. ayraudi*, *T. novaezelandiae*, *A. strigatus*, *P. dentex*, and *S. lalandi* explaining an average of 21, 21, 20, 17 and 11% respectively of the dissimilarity between the on-reef assemblage and the surrounding distances.

Figure 11 Average (± S.E.) MaxN ntu^{-1} between the four treatment distances (on-reef, 30 m, 100 m, 500 m), for all species observed (Total) and the four species observed both on the reef and in the surrounding pelagic environment: *T. novaezelandiae* (Tn), *P. dentex* (Pd), *N. ayraudi* (Na), and *S. lalandi*(Sl). The results of Tukey's HSD test are shown (distances within species that do not share a letter are significantly different).

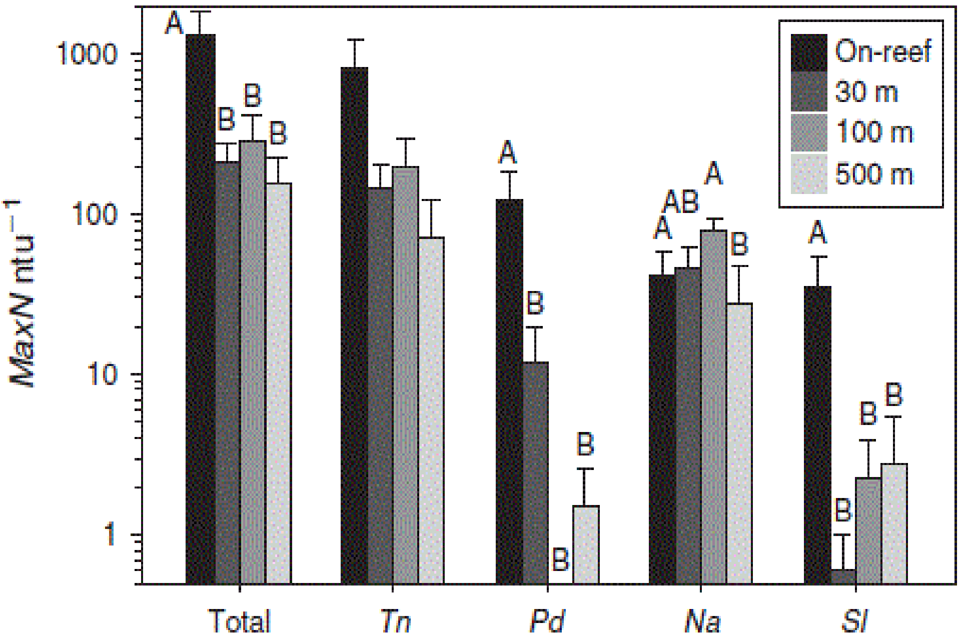


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Table 6 Species observed during the study, and their total MaxN, reported separately for each distance (on reef (<5 m); near, (30 m); mid, (100 m); far, (500 m). There was a total of 76 PBRUV (near, mid and far distances) and 6 drop camera (on reef) deployments. Also reported are the number of deployments in which the taxa were observed (sightings), and the single largest MaxN observed for each species during the study (Largest MaxN).

		Total *MaxN*					
Species	**Common Name**	**Distance**				**Sightings**	**Largest *MaxN***
		On-reef	**Near**	**Mid**	**Far**		
Nelusetta ayraudi	Ocean leatherjacket	49	220	284	158	29	178
Trachurus novaezelandiae	Yellowtail scad	979	739	596	332	24	450
Seriola lalandi	Yellowtail kingfish	42	2	8	16	10	25
Pseudocaranx dentex	Silver trevally	143	66	0	8	8	65
Meuschenia freycinetti	Sixspine leatherjacket	0	1	1	1	4	3
Platycephalus caeruleopunctatus	Bluespotted flathead	0	3	0	2	3	2
Lagocephalus inermis	Smooth pufferfish	0	0	5	1	3	4
Chromis hysilepis	One-spot puller	0	0	0	254	1	254
Scomber australasicus	Slimy mackerel	0	0	2	0	1	2
Sarda australis	Australian bonito	0	1	0	0	1	1
Scorpis lineolata	Silver sweep	0	0	0	1	1	1
Makaira indica	Black marlin	0	0	1	0	1	1
Isurus oxyrinchus	Shortfin mako	0	1	0	0	1	1
	Juvenile fish (<5 cm)	0	30	2	33	9	20
Atypichthys strigatus	Mado	315	0	0	0	4	150
Dinolestes lewini	Longfin pike	17	0	0	0	1	17
Meuschenia scaber	Velvet leatherjacket	2	0	0	0	1	2

Table 7 Results of PERMANOVA tests. These test the effect of distance from the OAR on the composition of the fish assemblage. Reported are the degrees of freedom (d.f.), mean squares (m.s.), the pseudo-F statistic, and the pseudo-P value. A pseudo-P< 0.05 indicates a treatment has a significant effect (shown in bold). Analyses were done on raw abundance data (MaxN), on abundance data standardised to turbidity (MaxN ntu^{-1}), and on abundance data standardised to both turbidity and plume area (MaxN ntu^{-1} m^{-2}). All four distances were analysed and compared with separate analyses of the three distances sampled using only pelagic baited remote underwater video (PBRUV) (30, 100, 500 m). The on-reef distance was sampled using unbaited cameras, so the final data standardisation is not applicable (NA).

		d.f	MaxN			MaxN (ntu^{-1})			MaxN (ntu^{-1} m^{-2})		
			m.s.	*F*	P	m.s.	*F*	P	m.s.	*F*	P
On-reef, 30 m, 100 m, 500 m	Distance	3	5526.5	3.843	**0.001**	6216.5	3.4662	**0.002**		N/A	
	Residual	78	1438			1793.4					
30 m, 100 m, 500 m	Distance	2	1925	1.3486	0.223	2406.3	1.3389	0.242	204.69	1.7242	0.121
	Residual	73	1427.6			1797.3			118.72		

One-factor ANOVA reported a significant effect of distance on total fish abundance (total MaxN ntu^{-1}, $F_{3,81}$ = 5.61, P <0.01) (Figure 11), but there was no difference in abundance between the three off-reef distances (Figure 11), nor when the on-reef data were removed from the analysis (total MaxN ntu^{-1}, $F_{2,75}$ = 1.18, P = 0.32). One-factor ANOVA testing the abundance of the four species observed both on-reef and off-reef, found distance significant for three of these (Figure 11) – *N. ayraudi* (MaxN ntu^{-1}, $F_{3,81}$ = 3.08, P = 0.03); *P. dentex* (MaxN ntu^{-1}, $F_{3,81}$ = 8.27, P <0.01), and *S. lalandi* (MaxN ntu^{-1}, $F_{3,81}$ = 10.53, P <0.01) – and non-significant for *T. novaezelandiae* (MaxN ntu^{-1}, $F_{3,81}$ = 2.26, P = 0.09). Distance was not significant for any of these species when the on-reef data were removed.

Connectivity of fish with nearby natural reefs

While the stereo-BRUV reef associated study did evaluate fish assemblages on nearby natural reefs, the degree to which move among the OAR and the surrounding habitats could not be determined. Acoustic telemetry however, allows fish to be tracked and movement patterns recorded, thereby gaining an understanding of the time spent at the OAR, as well as movements to other habitats. The broad objective of this study was to examine the site residency, connectivity and general movement patterns of Eastern Fiddler Ray (*Trygonorrhina fasciata*), Bluespotted Flathead (*Platycephalus caeruleopunctatus*), and Port Jackson Shark (*Heterodontus portusjacksoni*), in association with the OAR. The Bluespotted Flathead was a focal species for this study, as it supports a state-wide recreational harvest of between 320 and 450 tonnes. The Fiddler Ray and Port Jackson Shark were also selected as they are common benthic predators often incidentally captured by recreational fishers. The specific aims of this study were to: 1) determine the site residency of three benthic species at the OAR and at nearby natural reefs using acoustic telemetry; 2) determine the connectivity between the designed artificial reef and natural reefs for these species, and 3) use this information to infer the effect of this designed artificial reef on the local distribution of these species.

Methods

Acoustic Array

A VR2W receiver (Vemco Ltd, NovaScotia, Canada) was deployed on the OAR from 2011 to 2013, after which it was then replaced by a VR4 receiver (Vemco Ltd, NovaScotia, Canada) to provide remote uploading capability. The receiver was tethered to a cross-beam on the artificial structure approximately 8 m from the seafloor (Figure 12). Two VR2W receivers were also deployed on areas of scattered reef at Dunbar, one north (33°50'47.76"S, 151°17'27.60"E) and another south (33°51'4.32"S, 151°17'27.60"E) at approximately 25 m depth. All receivers were coated with a copper-based antifouling paint to prevent possible signal occlusion due to biofouling. Receivers were downloaded every 3 to 6 months over a period of 2 years. Broad-scale detections of tagged individuals detected in the greater Sydney area (Figure 13), and further afield, were downloaded from the Integrated Marine Observing System (IMOS) Animal Tracking acoustic telemetry array. This array consists of over 350 receivers in the region, and is publically available online (https://aatams.emii.org.au/aatams).

Figure 12 **Vemco VR4 acoustic receiver attached to the Sydney OAR. One of the two towers that rise from the main structure of the OAR can be seen in the background.**

Acoustic Tagging

Eastern Fiddler Ray, Bluespotted Flathead and Port Jackson Shark are three large-bodied species that were identified (prior to tagging) from BRUV footage collected at both the OAR and nearby natural reef at Dunbar. These three species were captured for tagging using various fishing methods, including rod and reel, fish trap, or modified long-line. Individuals collected by rod and reel were captured with circle-style hooks on monofilament line, baited with pilchard or squid. Modified long-lines comprised of a 10 m bottom set, 8 mm mainline rope, weighted on each end (3 kg), with one end attached to 20-30 m (depending on depth) 8 mm nylon float line tied to a surface buoy. Each long-line had 3 gangions which consisted of 1 m of monofilament line and a circle hook (6/0) baited with pilchard or squid, attached 2 m apart with a shark clip. Two long-lines were usually deployed simultaneously, depending on weather conditions and soaked for approximately 30 min. Upon capture, individuals were placed into a 20-30 L tub containing fresh seawater that was continuously aerated, and tagging was conducted either on the boat at the site of capture, or transported to a nearby private jetty. Prior to surgery, captured animals were anaesthetised with 0.5 ml Aqui-S (AQUI-S New Zealand Ltd) per kg of fish, except for Port Jackson sharks which were inverted to induce tonic immobility and operated on without anaesthetic (Wells *et al.* 2005). All captured individuals were measured to the nearest cm (total length – TL), sexed, and surgically implanted with a 69 kHz Vemco acoustic transmitter (Vemco Ltd, NovaScotia, Canada) into the peritoneal cavity (Figure 14). A variety of acoustic transmitters were used (Table 8), the type of transmitter depended on animal size and availability.

Figure 13 **Study area showing receiver locations. AR= OAR, NR= Dunbar receivers (north and south), SG= Sydney harbour gate receivers (1-4), SH= Sydney Harbour, BSH= Between Bondi and South Head, BL= Bondi line receivers (1-4), BCAR= Bronte-Coogee Aquatic reserve, SLB= South Long Bay. Insert shows location of designed artificial reef (AR), nearby natural reef, and the two Dunbar receivers (north and south). Numbers are depths in metres. Note that Dunbar reef (NR) is the area of natural reef that projects out from the coast between the two receivers. Bathymetry information is from acoustic surveys by the NSW Office of Environment and Heritage.**

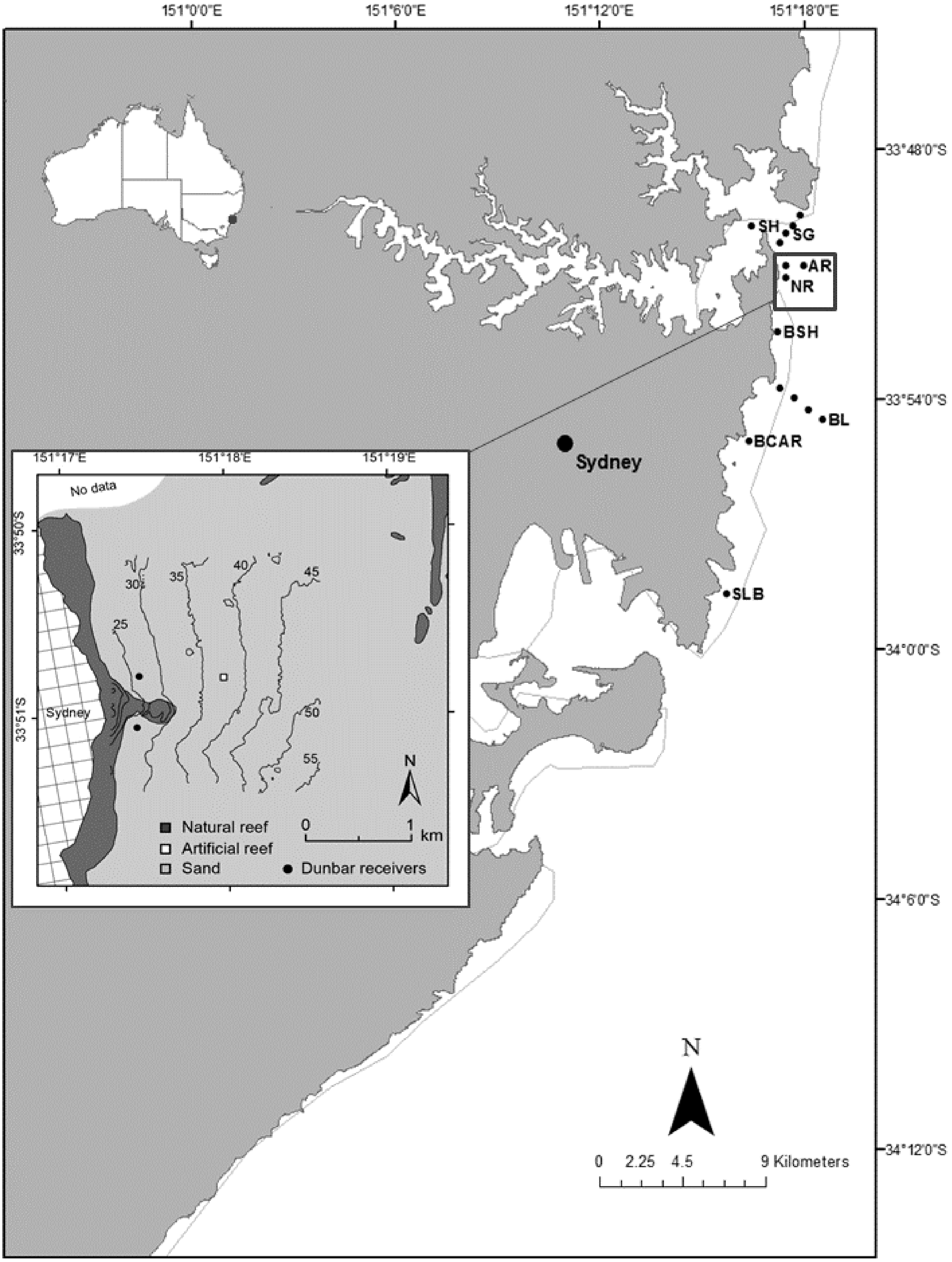
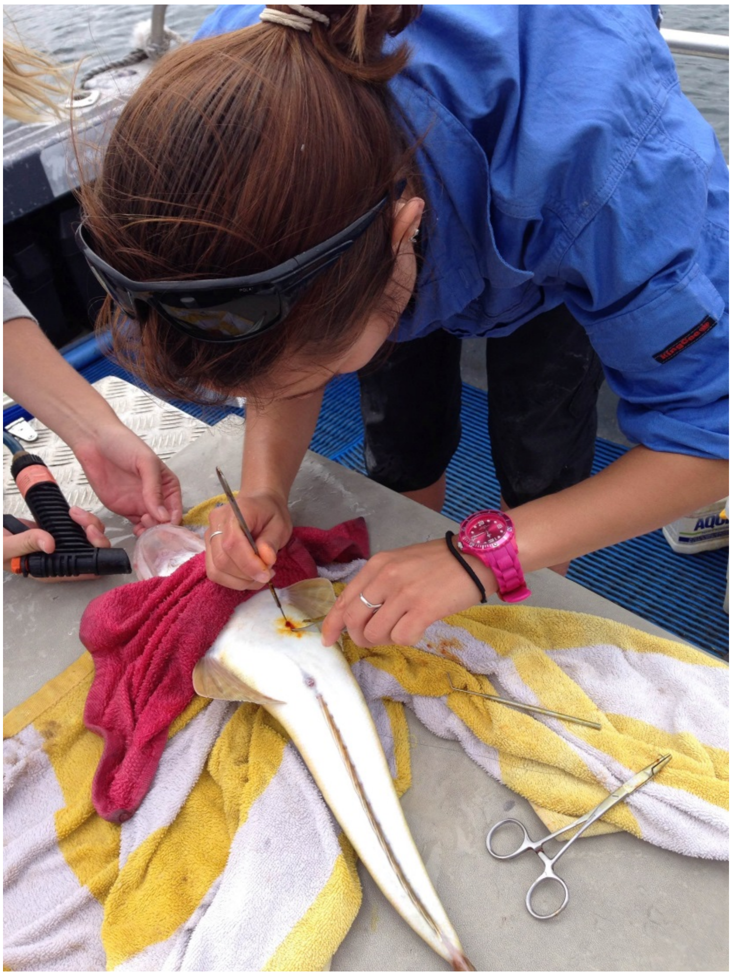


Figure 14 An anesthetised Bluespotted Flathead undergoing surgery to implant a Vemco acoustic transponder.



The approximately 3 cm-long incision was sutured with one or two stiches using non-absorbable polyamide monofilament with a curved cutting needle. Each individual was injected with Engemycin 100 (50 mg kg^{-1} body weight) to prevent infection and allow fast recovery. All Bluespotted Flathead were individually tagged with externally visible T-bar anchor tags (Hallprint, Hindmarsh Valley, SA, Australia, http://www.hallprint.com/) in the dorsal fin musculature and marked with contact details in case of recapture by fishermen. Sterile surgical methods and Betadine® were used throughout the procedure.

After surgery, individuals were returned to a tub containing fresh seawater and transported by boat back to the site of capture for release. Fish were released only after full recovery and normal signs of activity were observed. In order to avoid mortality after release, individuals which did not recover fully within 20 min were euthanized and were not released. A total of 9 Eastern Fiddler Ray, 25 Bluespotted Flathead and 9 Port Jackson Shark were tagged at the Sydney OAR; and 1 Fiddler Ray, 1 Bluespotted Flathead and 8 Port Jackson Shark were tagged at the Dunbar natural reef (Table 8).

Table 8 Summary of acoustically tagged animals and corresponding tagging reefs. AR = Sydney OAR, NR= Dunbar reef; TL= total length; M= male; F= female; I= fish of indeterminate sex. L= Low power, H= High Power.

Animal	Tagging reef	Number tagged	Sex	TL range (mm)	Tag types	Power output	Min-max tag delay (s)	Min-max Monitoring period (d)
Eastern Fiddler Ray	AR	9	M	742-840	V9-1x,	L	400-800	65-670
					V9A-2x	H	25-290	
					V9AP-2L	L	220-500	
	NR	1	M	820	V9AP-2L	L	220-500	475
Bluespotted Flathead	AR	25	I	250-550	V7-4x	L	130-230	61-651
					V9-2L	L	170-310	
					V9A-2x	H	25-25	
					V9AP-2L	L	220-500	
	NR	1	I	415	V9A-2x	L	25-25	157
Port Jackson Shark	AR	9	M+ F	640-1150	V9AP-2L	L	220-500	602
	NR	8	M+ F	880-1900	V9AP-2L	L	220-500	592-602
					V13-1x	L	140-240	

Site Residency

Data collection spanned approximately 2 years, with receiver downloads from June 2013 to June 2015 at the OAR, and from June 2013 to November 2014 at the Dunbar natural reef. Detections within the first 24 h after release of tagged individuals were excluded to ensure that behaviour was not influenced by the tagging procedure and data were filtered to remove potential spurious detections. Single transmitter detections were considered false detections and removed from the analyses (Reubens et al. 2013). Heavy rainfall events (>30 mm) measured from the Dover Heights weather station (BOM 2015), were compared with detections from each species; however there was no obvious relationship between individual absence/presence at the OAR or natural reef observed during the monitoring period.

Total residence time was calculated for every tagged individual and was defined as the total number of days that an individual was detected at a specific receiver. Total residence time was often the sum of multiple residence periods. A residence period for an individual began when a minimum of two detections on a specific receiver were recorded within a 24 h period (Campbell *et al.* 2012). Non-residence time was calculated as the duration between consecutive residence periods. A residence period ended when that individual was either detected at another receiver or was not detected within 24 or 48 h after the last detection, depending on the minimum number of days that an individual was considered resident. In this study, the minimum number of days that an individual was considered resident was determined by calculating a cumulative percentage of detections, as days between consecutive detections for each species in the study area during the monitoring period (Figure 15). For Bluespotted Flathead, 85% of the total detections were recorded with 3 or fewer days between them, whereas 97% and 93% of total detections of Eastern Fiddler Ray and Port Jackson Shark were recorded with 2 or fewer days between them, respectively (Figure 15). Hence, Eastern Fiddler Ray and Port Jackson Shark were considered resident if there was no more than 1 day between consecutive detections, and Bluespotted Flathead were considered resident if there were no more than 2 days between consecutive detections.

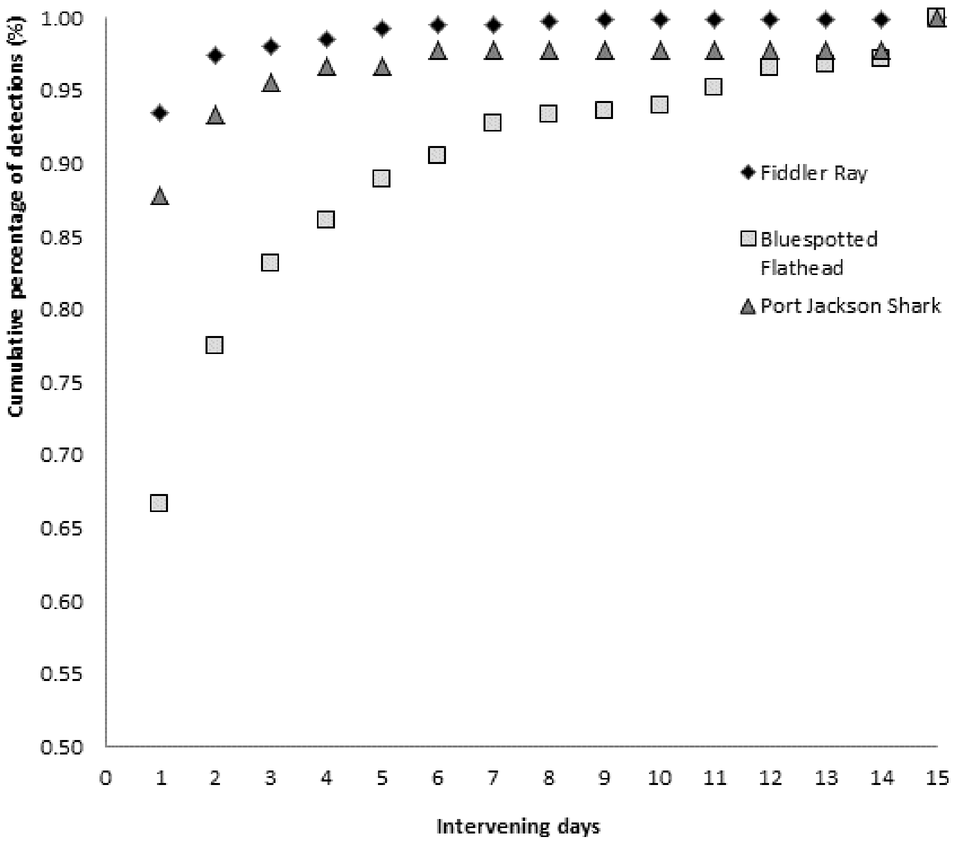
Definitions of residency for each species were considered to be appropriate since increasing the number of days between detections made negligible difference to the overall residency calculations. The rationale behind the 24 or 48 hour residency ‘grace period’ was that receivers can have a fairly short range, assumed here to be 200 m radius from the receiver, which was a conservative estimate based on a maximum detection range of 500 m (e.g. Espinoza *et al.* 2015; Lee *et al.* 2015; Villegas-Ríos *et al.* 2013). This grace period therefore incorporated a period of absence (i.e. of 24 or 48 hours) if an individual is located just outside the detection range of a receiver. All residency analyses were conducted using the ‘VTrack’ R package (Campbel *et al.* 2012) with the R 2.15.1 software (www.rproject.org).

A residency index (I_R) was determined for each individual for each tagging reef (OAR or Dunbar). I_R was calculated by dividing the total residence time at a tagging reef by the total monitoring period (Reubens *et al.* 2013; Villegas-Ríos, Alós *et al.* 2013):

$$\mathrm{I_R} = \frac{t_\mathrm{R}}{t_\mathrm{M}}$$

where t_R is the total number of days an individual was detected during the monitoring period (total residence time) and t_M is the total monitoring period for each fish, from the first day an individual was detected till the last day of the monitoring period (monitoring time). The monitoring period depended on the battery life of the transmitters, known captures of tagged fish, and the date of last receiver download (June 2015), which meant there was some variation in monitoring periods among individuals (Table 8). Due to these various monitoring periods, t_R was also compared between all individuals to determine the maximum residence times spent at each tagging reef. $I_R = 0$ indicates no residency and $I_R = 1$ indicates permanent residency. Differences in residency at the AR between the three species were tested using ANOVA on the I_R proportions, after a logit-transform to improve normality and homogeneity of variance (Warton and Hui 2011). The minimum non-zero proportion across all species was added to the numerator and denominator in the logit function to allow for values of zero (Warton and Hui 2011). The ANOVA tested the residency across the entire monitoring period for each individual, as there was insufficient data to explore seasonal effects.

Figure 15 **Cumulative percentage (%) of detections of Eastern Fiddler Ray, Bluespotted Flathead and Port Jackson Shark in study area during monitoring period.**



The proportion of total days spent at the OAR, Dunbar and other sites in the study area by all tagged individuals was determined to examine the effect of the site of tagging on these individuals. The number of days detected at each receiver was summed together for all individuals, and the proportion of total days detected at that receiver was examined separately for individuals tagged from either the OAR or Dunbar. Receivers in the study area were classified into 6 main sites (Figure 13): AR (Sydney OAR receiver), NR (north and south Dunbar receivers), Sydney Harbour (SH- SYT233 receiver), Sydney gate (4 receivers at entrance to Sydney harbour – SG1-4), Bondi (Bondi line receivers – BL1-3) and between Bondi and South Head (BSH-SYT409 receiver). A seventh site was also included; Narooma (5 receivers– N2-6), located south of Sydney (36°15'38.60"S, 150°10'57.70"E).

Site Connectivity

Connectivity between the OAR and natural reefs was determined for each species, by comparing the proportion of individuals detected at each receiver in the study area (Figure 13). A distance matrix (in metres) between all sites in the network of receivers was calculated using a 200 m radius for each receiver with the 'VTrack' R package. Mean and median distance travelled from the tagging reef (either the OAR or Dunbar) were determined for each species by calculating the proportion of individuals per species detected within 8 km, at 1 km intervals, and those detected at distances greater than 8 km.

Results

An ANOVA on logit-transformed I_R values showed that Eastern Fiddler Rays had significantly higher average residency ($F_{2,34}$ = 7.806, P = 0.002) than the other two species during the monitoring period (Table 9). Fiddler Ray exhibited the highest residency at the OAR with an average I_R of 0.32, while Bluespotted Flathead and Port Jackson Shark exhibited low average residency ($I_R \leq 0.1$) both at the OAR and Dunbar (Table 9). Nine Fiddler Ray tagged and released at the OAR were detected for periods longer than 2 days. Three individuals were only detected at the OAR with all others detected at up to 5 additional sites (Figure 16). 61% of tagged Bluespotted Flathead were detected at the AR for 2 or more days (Figure 16). 52% of Bluespotted Flathead were only detected at the OAR, of the other flathead, 43% were detected at up to 6 additional sites (Figure 16). All Port Jackson Sharks were detected at 1 – 5 additional sites from their original tagging location. 76% of Port Jackson Shark were detected at their tagging reefs but only irregularly from the beginning of the monitoring period when they were first tagged and detections ceased after October 2013. 33% of individuals were detected again at their tagging reefs between 282 to 334 days after the last date of detection (Figure 16).

Table 9 Summary of residency information at corresponding tagging reef for Eastern Fiddler Ray, Bluespoted Flathead and Port Jackson Shark. t_R = total number of days an individual was detected over the monitoring period (total residence time), I_R = Residency index.

Animal	N	t_R (min-max)	Average tR	I_R (min-max)	Average I_R	Tagging reef
Fiddler Ray	8	2-402	108	0.01-0.77	0.32	OAR
	1	28	28	0.06	0.06	Dunbar
Bluespotted Flathead	20	0-130	20	0-0.47	0.07	OAR
	1	15	15	0.1	0.1	Dunbar
Port Jackson Shark	9	4-77	25	0.01-0.13	0.04	OAR
	6	1-19	10	0-0.03	0.02	Dunbar

The proportion of time spent at the OAR and Dunbar was highest by individuals that were tagged from each corresponding reef (Figure 17, Figure 18). The total number of days detected at the OAR by all three species was highest at this reef compared to other sites. Individuals tagged from the OAR also spent time at other sites in the study area during the monitoring period. In addition, Port Jackson Shark tagged from the AR spent a proportion of time over summer and autumn near Narooma, located approximately 280 km south of Sydney (Figure 18).

Regarding the effects of environmental variables, there was no effect of either time of day (hour), water temperature, rainfall or wave height on the detection frequency of control tags placed 50 m from the receiver. However for a control tag placed 200 m from the receiver an effect of hour (P = 0.007) and wave height (P = 0.002) on detection frequency. It was shown that detection frequency at the 200 m distance was lower from 0700 – 0900 hours, with increasing wave height also reducing the amount of detections. It was subsequently determined that detection efficiency for the 50 m control tag was 78%, and this efficiency dropped to 68% for the 200 m control tag.

Connectivity

The majority of Eastern Fiddler Ray, Bluespotted Flathead and Port Jackson Shark were detected up to 2 km from their tagging reef, with 62%, 81% and 71% of individuals detected respectively (Figure 19). The maximum distance that individuals were detected from their tagging reef was 7 to 8 km for Fiddler Ray, 5 to 6 km for Bluespotted Flathead and >8 km for Port Jackson Shark. The greatest distanced detected by a tagged individual was ~ 280 km on the Narooma array (Figure 18).

Figure 16 a-c **Presence plot of animals monitored in the Sydney region from June 2013 to June 2015, a) Eastern Fiddler Ray, b) Bluespotted Flathead and c) Port Jackson Shark. Black dots indicate detection at the designed artificial reef (AR), other colour symbols indicate detection at other receivers (SG= Sydney gate, NR=Dunbar, BCAR= Bronte-Coogee marine reserve, BL=Bondi line, BSH= between Bondi and South Head, SH= Sydney Harbour, SLB= South long Bay, NAR= Narooma). Grey crosses indicates end of tag life.**

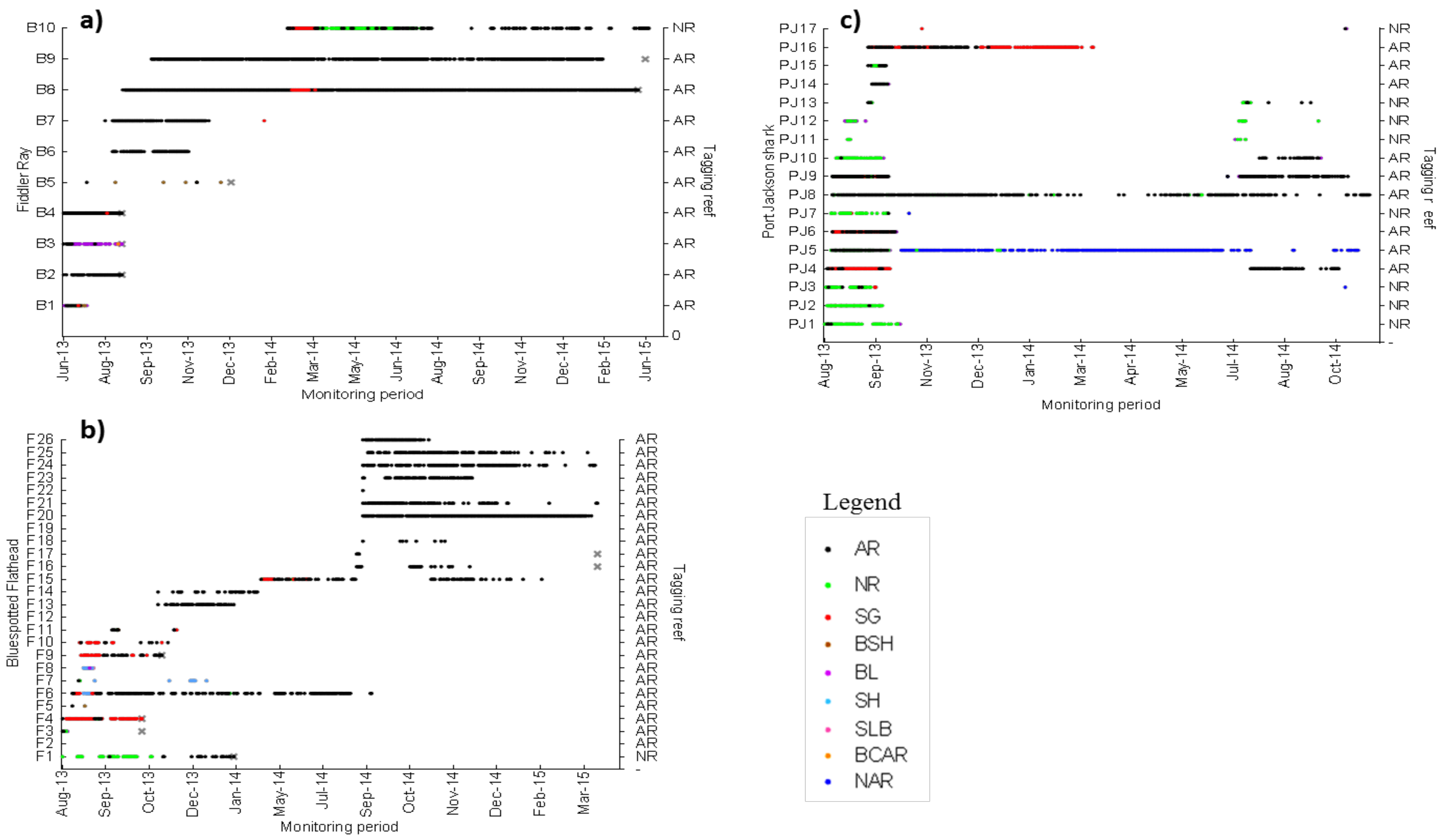


Figure 17 Proportion of total days that a) Eastern Fiddler Ray and b) Bluespotted Flathead tagged from either the AR (designed artificial reef) or NR (Dunbar reef) were detected at receivers in the study area during the monitoring period. DBHN=north Dunbar reef, SG1-4= Sydney gate, SYT233= Sydney harbour, SYT409=Between Bondi and South Head, BL1-3= Bondi.

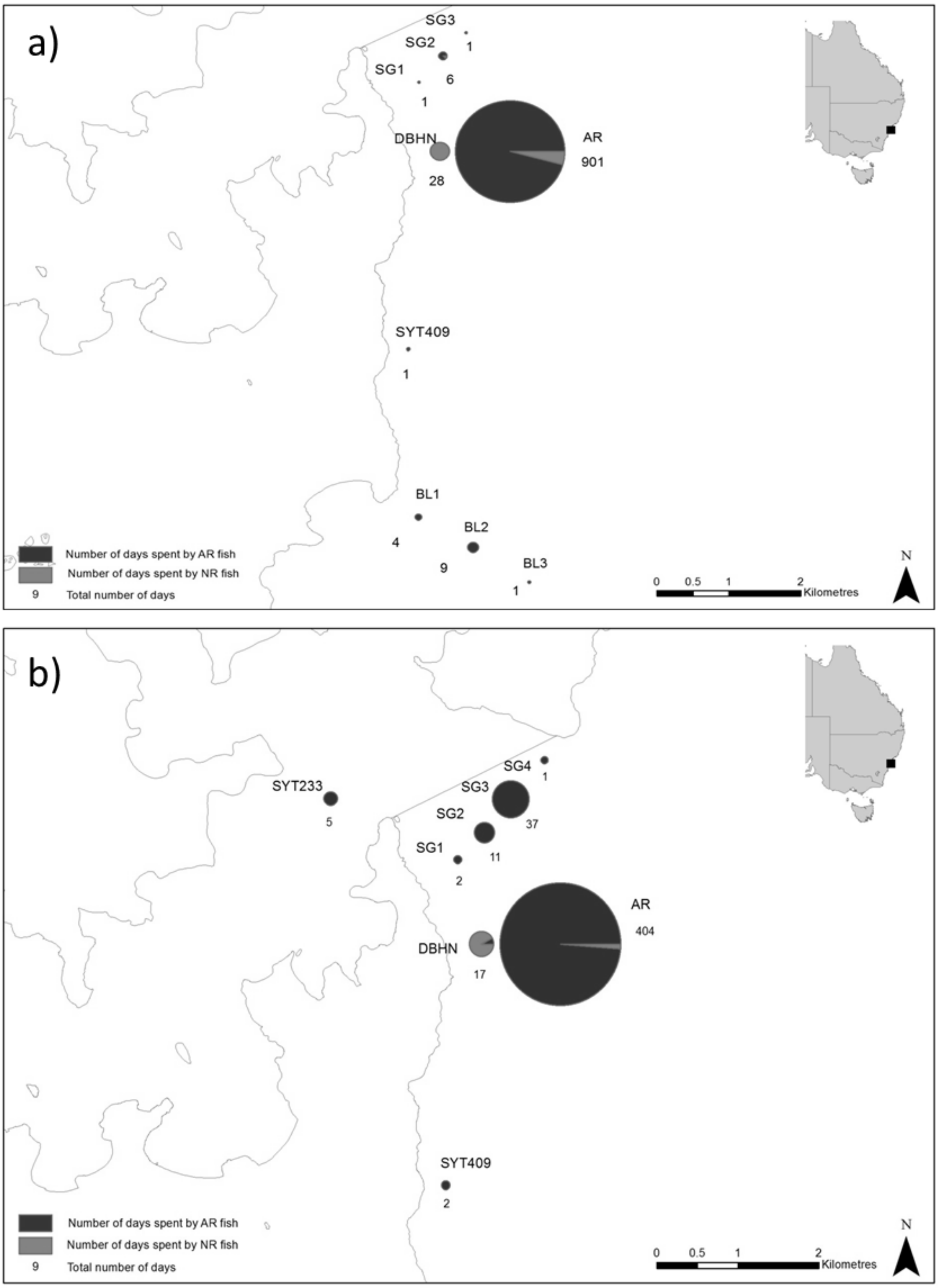


Figure 18 **Proportion of total days that Port Jackson Shark tagged from either the AR (designed artificial reef) or NR (Dunbar reef) were detected at receivers in the study area in greater Sydney area (top panel) and south coast (bottom panel) during the monitoring period. DBHN=north Dunbar reef, DBHS=south Dunbar reef, SG1-4= Sydney gate, SYT409=Between Bondi and South Head, N2-6= Narooma.**

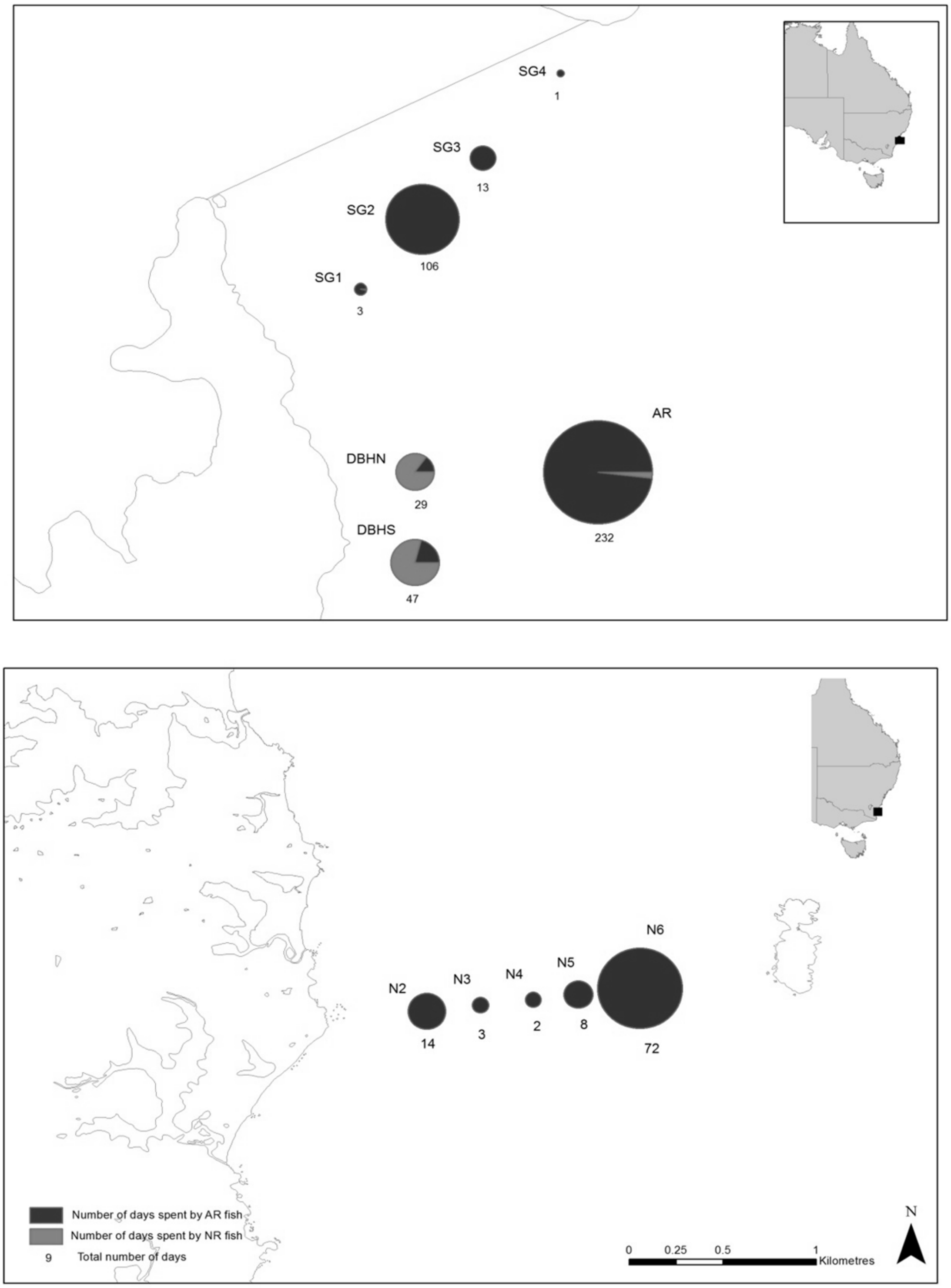
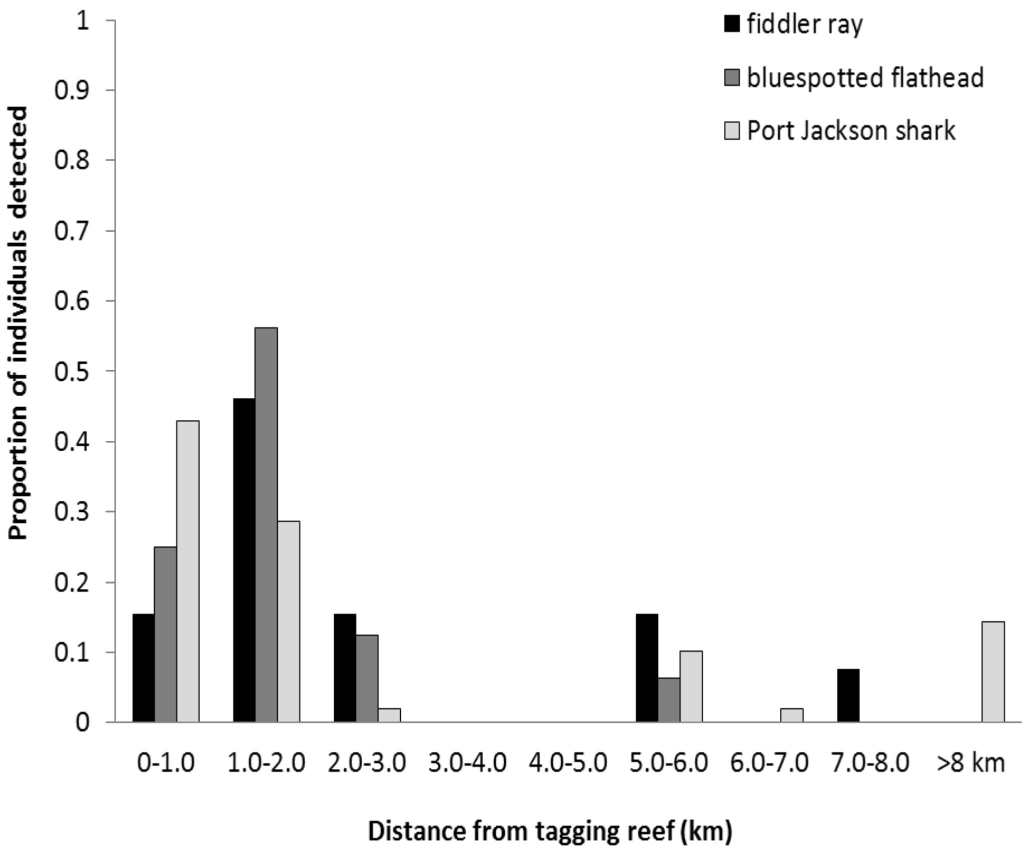


Figure 19 Proportion of individuals per species detected and the corresponding distances from their tagging reef during the monitoring period.



Discussion

The goals of the Sydney OAR were to create fish habitat which would provide additional quality fishing opportunities to anglers. Achieving this goal was reliant upon a complex fish assemblage becoming established. This was evident, with the reef resident stereo-BRUV study observing 53 species, including key recreational species such as *S. lalandi* and *P. georgianus* and their prey species (e.g. *T. novaezelandiae*). Similarly, the pelagic BRUV study also recorded a number of key recreational species in the water column around the structure. The OAR is able to achieve its primary goal despite differences in fish assemblages with control reefs. While early artificial reef development attempted to mimic natural reefs, in the belief that creating an identical assemblage would support similar goals, modern reef design attempts to create habitat for a range of species, but not necessarily establishing an assemblage similar to nearby natural reefs.

Unlike the control reefs, most modern artificial reefs such as the Sydney OAR, are high relief structures, and the vertical orientation has been shown to result in differing benthic as well as pelagic fish communities (Clynick *et al.* 2008; Perkol-Finkel *et al.* 2006; Rilov and Benayahu 2000). In the water column around the towers, the pelagic BRUV study revealed a diverse fish assemblage including representatives of coastal pelagic species (e.g. *T. novaezealandae*, *S. lalandi*), oceanic (e.g. *Isurus oxyrinchus*, *Makaira indica*), reef associated (*P. dentex*, *N. ayraudi*) and soft bottom benthic (*Platycephalus caeruleopunctatus*). Most species were rare, but the four most common species were observed both 'on-reef' as well as the three increasing distances away from the structure. Neither study sampled the pelagic zone around natural reefs, so comparisons with the pelagic fish assemblage at the Sydney OAR are not possible.

The stereo-BRUV study focusing on reef fish ran across four years, during which time the fish community at the OAR showed distinctive inter-annual variation which was not evident at each of the three control reefs. As this temporal variation was restricted to the OAR, successional processes on and around the structure (Connell and Slatyer 1977), rather than broader regional effects, are likely attributable to these changes. Large shifts in fish assemblages following the deployment of artificial reefs is a common response recorded in numerous locations (Bohnsack *et al.* 1994; Leitão *et al.* 2008; Thanner, McIntosh *et al.* 2006). The arrival of new species can be the result of the recruitment and growth of larvae, or the arrival of adults (Bohnsack, Harper *et al.* 1994; Herrera *et al.* 2002). Underpinning these processes is the requirement of suitable habitat and the provision of protection and food it provides to succeeding assemblages. No single group of fish initially dominated the OAR, with initial colonization by a variety of demersal species (e.g. *Trygonorrhina fasciata*, *Parupeneus spilurus*), small pelagics (e.g. *T. novaezelandiae)* and reef associated fish (e.g. *Meuschenia freycineti*). Early colonization by benthic feeding reef associated species is likely a function of the shelter the artificial reef provides, rather than other ecological functions such as a food source, because the development of sessile communities takes time (Perkol-Finkel and Benayahu 2007; Relini *et al.* 1994; Woodhead and Jacobson 1985). The diversity of early species indicates the design of the module immediately provides habitat for different groups of species. Deployment of the structure changes local-scale hydrological conditions, providing refuge from current flow, while also congregating small prey items (Hobson and Chess 1976). This may explain the appearance of small pelagics such as *T. novaezelandiae*. The OAR would provide these services regardless of age, but the interactions between artificial reef habitat, currents and fish distribution, is still a developing area of research.

Although fish assemblages at the OAR showed distinct inter-annual change, there was no obvious convergence in the fish assemblage on the OAR and control reefs. The differences observed after four years were similar to what was observed soon after the reef was deployed, and our data indicates monitoring over multiple years is required before an evaluation of community levels effects can be undertaken. Indeed, it is possible the community continued to change at the completion of the field component of this project, and may continue to change up to 10 years post deployment (Relini, Relini *et al.* 2002). At the very least, this project suggests that post-deployment monitoring over multiple years is required to understand fish assemblage responses in the face of ecological succession, and even then convergence with natural reef assemblages should not be expected when the physical reef structure is so different.

Despite inter-annual changes in fish composition, species diversity at the conclusion of the study was similar to the beginning. Given that distinct annual changes in the community were observed, a relatively similar diversity between these years suggests successional changes in the assemblage composition at the OAR did not necessarily lead to an overall increase in diversity. Essentially, one group of species may be replacing another, and such patterns are well documented in successional studies (Connell and Slatyer 1977). Longer time periods may be required for an increase in overall diversity to occur.

Interestingly, there was evidence of differences among the control reefs themselves, although these appear to be consistently less than comparisons between these reefs as a whole, and the OAR. A number of species were responsible for these differences, however some patterns were evident. For example, abundance of *Chromis hypsilepis* was generally higher at Bondi compared to other control reefs, while abundances of *A. strigatus* were consistently highest at Dunbar. This further highlights how varying complexity, size and isolation can affect local fish assemblages. Bondi and North Head were most similar and this is likely because both these reefs are large and continue from shoreline cliffs, while Dunbar is smaller and isolated from the cliffs by 500 m.

The height of purpose-built artificial structures appears to be an important feature in increasing the overall diversity of fish assemblages around the reef, including the bottom section. Key species which were responsible for the differences between the OAR and control reefs include pelagic species such as *T. novazealandie*. It is likely that the towers initially provided mid-water habitat for these species and they subsequently moved to deeper sections of the reef and were detected in stereo-BRUV deployments. The pelagic study showed these fish were generally only associated with the OAR on a small scale, with abundances of fish rapidly declined with distance (< 30 m) from the reef. There was no significant difference in total abundances between 30 m, 100 m and 500 m from the reef, indicating any reef effect only exists within a zone less than 30 m around the structure. This is consistent with other research which has found most fish tend to be associated with artificial reefs within an area of between 20 and 50 m (dos Santos *et al.* 2010; Fabi and Sala 2002). This 'boundary effect' has also been observed around larger structures such as oil platforms (Boswell, Wells *et al.* 2010; Stanley and Wilson 1997). This relatively close association with artificial structures may suggest some form of trade-off between predation risk and foraging success, at least for reef associated fishes (Biesinger *et al.* 2011). The Sydney OAR can be considered large, for a purpose built single unit, however there are larger reefs elsewhere in the world (Boswell, Wells *et al.* 2010; Stanley and Wilson 1997). Importantly, the 'area of effect' of artificial reefs is not linear with reef size (Jordan, Gilliam *et al.* 2005). Therefore, doubling the size of an artificial reef will not double the area of this boundary effect.

Fish generally have a close spatial association with artificial reefs, and it appears even pelagic species are unlikely to be detected at distances greater than 30 m. A number of factors may influence this scale including the size of the artificial structure and the composition of the fish assemblage, as well as the distance of the reef from other structures. More isolated structures may have higher species diversity and be used by a larger abundance of pelagic fish (Jordan, Gilliam *et al.* 2005; Vega Fernández *et al.* 2008; Walsh 1985). The design of artificial reefs in NSW now includes dropping clusters of concrete reef modules, creating a 'reef field'. Typically these modules are positioned with a high degree of precision, therefore a key consideration when planning the layout of a reef field is the proximity between individual modules themselves. New deployments are also increasingly incorporating steel towers extending above these concrete structures in an attempt to also provide habitat for pelagic species.

Although fish appeared to show a close association with the Sydney OAR structure, the telemetry study indicated the three species tagged moved among the OAR and nearby natural reefs. It must be noted however that these were all benthic species. This connectivity was evident between the OAR and nearby natural reefs, with all species exhibiting movements >5 km from their tagging reefs and visiting up to 5 or 6 other reef areas during the monitoring period. Despite this, most individuals remained within 2 km from their tagging reef (either the OAR or Dunbar). The nearest natural reef to the OAR is only 550 m away (Dunbar Reef). Closely-spaced habitats can promote the dispersal of mobile species, particularly those which have the ability to move over relatively large distances (Chin *et al.* 2013). Some Port Jackson Sharks travelled over 8km from their tagging reef, with the furthest location this species detected being 280 km south of the OAR at Narooma, highlighting their large scale movements and high dispersal frequency among reef areas. For more resident species with limited movements, bare sandy areas can act as a barrier to movement. Artificial reefs offer the potential to alter the seascape connectivity and habitat use of individuals by facilitating movement among habitat patches. Regular movements by all three species in this study however indicate that connectivity between this AR and natural reefs can occur on a localised scale. Additional telemetry work, with a focus on pelagic species and additional reef associated fishes would help to further understand how artificial reefs contribute to the broader seascape.

Considering both the telemetry and camera work, it appears fish maintain a close association with the structure while resident on the artificial reef, however, there is evidence that connectivity with natural reefs will occur. This has important implications for the planning and design of future reef deployments, certainly it suggests that proximity to natural reef, or more broadly, its location within the existing seascape will likely have implications in both the fish which utilise the artificial reef and ultimately the overall assemblage.

Finally, the three studies also provided important evidence that the Sydney OAR was not providing any interaction with threatened species. Monitoring the occurrence of threatened/protected species on the OAR was a priority 1 monitoring objective outlined in the EMMP. Despite approximately 50 hours of footage being collected and processed, no threatened/protected species were identified. Furthermore, an increasingly number of threatened Chondrichthyes have been acoustically tagged, none were detected on receivers deployed on the structure.

Chapter 3 – Monitoring boat-based recreational fishing effort at the Sydney OAR with a shore-based camera – Krystle Keller, Aldo Steffe, Michael Lowry, Iain Suthers

Introduction

Assessing the effectiveness of the Sydney OAR in terms of popularity with recreational anglers was the single socio-economic priority 1 objective identified in the EMMP. New camera technology linked with Internet Protocol (IP) based systems now allow recreational fishing effort to be monitored at well-defined access points to a fishery, such as boat ramps, jetties, wharfs and rock groynes (Ames and Schlindler 2009; Smallwood *et al.* 2012). Such systems can also be used to monitor the surface area above artificial reefs which are within the focal range of land based cameras. As technology develops, it is likely shore based cameras will have the ability to observe boating activity at increasing distances in the future. The advantage of camera systems is they offer the opportunity to store a permanent record of fishing activity which can be accessed at any time after the sampling period is complete (Smallwood, Pollock *et al.* 2012). Such shore-based cameras have been used effectively to monitoring passing boats (Ames and Schlindler 2009), fishers on rock walls (Smallwood, Pollock *et al.* 2012) and angler effort in rural lakes (Van Poorten *et al.* 2015).

A key goal in the deployment of artificial reefs is to provide enhanced fishing opportunities for recreational anglers. Therefore, monitoring fishing activity on these structures is important in determining if they are effectively meeting this goal. An understanding of temporal patterns of fishing at the OAR is also important because fishing effort is positively correlated to the levels of fishing-related mortality (i.e. harvest and release-induced mortality). Despite the global widespread deployment of artificial reefs, there are few studies which assess recreational fishing effort patterns on these reefs (McGlennon and Branden 1994; Tinsman and Whitmore 2006).

Figure 20 **View from the shore based camera system, showing numerous boats in the waters around the Sydney OAR (Red X shows the location of the OAR).**

The aim of this study was to estimate recreational boat-based fishing effort at the Sydney OAR using a shore-based camera (Figure 20). A second aim was to quantify patterns of recreational boat-based fishing effort at the OAR for each season over two years and standardize the fishing effort by area to allow comparisons with other Australian artificial reefs and estuarine fisheries. These comparisons are important for determining the usage by recreational anglers and for assisting managers to determine the economic benefits of implementing additional reefs in the future.

Methods

Artificial reef survey area

The OAR monitoring area was calculated using time and position data, with a boat mounted Global Positioning System (GPS). The effective fishing area was defined as a spatial zone at and adjacent to the reef which would be used by fishers to target recreationally important species.

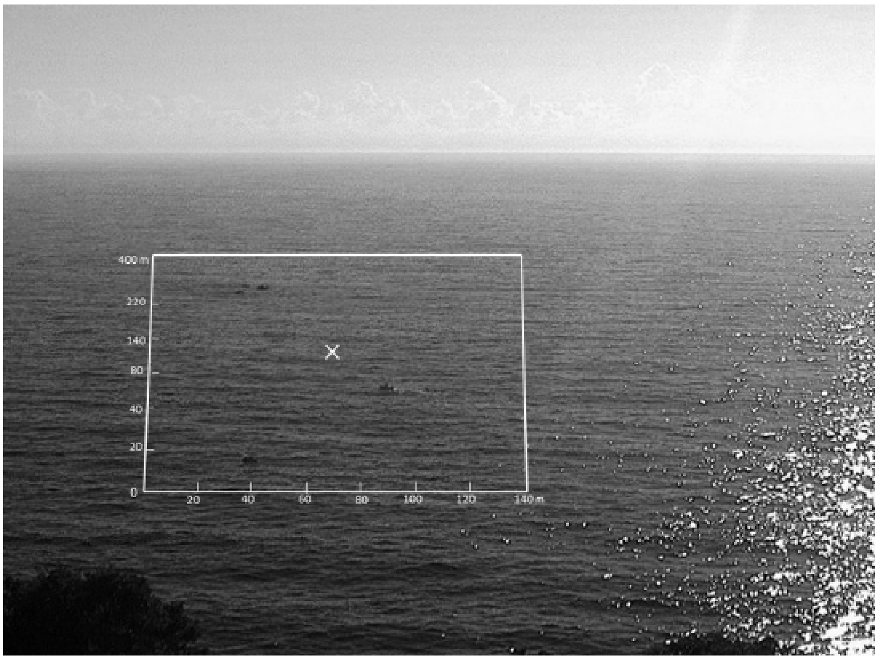
Camera imagery

A mobotix M24, 3 megapixel (2048 x 1536 pixel resolution, 8 x digital zoom, 45° horizontal lens, 8 mm focal length, 2.0 aperture) twin-head camera (www.anso.com.au) was used to monitor the recreational fishing effort at the OAR. The camera was fixed to the Old South Head Signal Station (a lighthouse) (33° 51' 1.47" S, 151° 17' 12.41" E) which is 85 m above sea level and has a clear view of the water surface above the reef from a distance of 1.3 km (Figure 21). Photographs were taken continuously every minute during pre-defined period of daylight (06:00 – 18:00 h) for two years between 1st June 2012 – 30th May 2014 resulting in a total of 188,370 images.

To eliminate vessels which were simply steaming past the OAR location, a pilot study was conducted to determine the time it took for non-fishing vessels to transit the area. It was subsequently determined any vessel which remained within the area directly above the OAR for 5 min or longer was fishing either by anchoring, drifting or trolling. From this, a fishing event was defined as a vessel remaining in the OAR area for at least 5 min (i.e. 5 frames). All types of vessels regardless of size, were counted and included in the fishing effort estimation so long as they remained in the vicinity of the OAR. The fishing effort data generated from the digital photos was in the units of fishing events and boat hours (i.e. the number of hours of boat-based fishing in the OAR area) with all photos analysed using Microsoft Office picture manager (Figure 22). A reference set of digital images was used to train and standardize the image interpretation.

Figure 21 The motobox M24, 3 megapixel camera, mounted on the Old South Head Signal Station.

Figure 22 **Camera view of the 0.056 km^2 monitored OAR area extending 140 m north to south x 400 m east to west. 'X' indicates the location of the OAR. Note the two different scale bars, the vertical scale bar indicates the increasing distance to 400 m.**

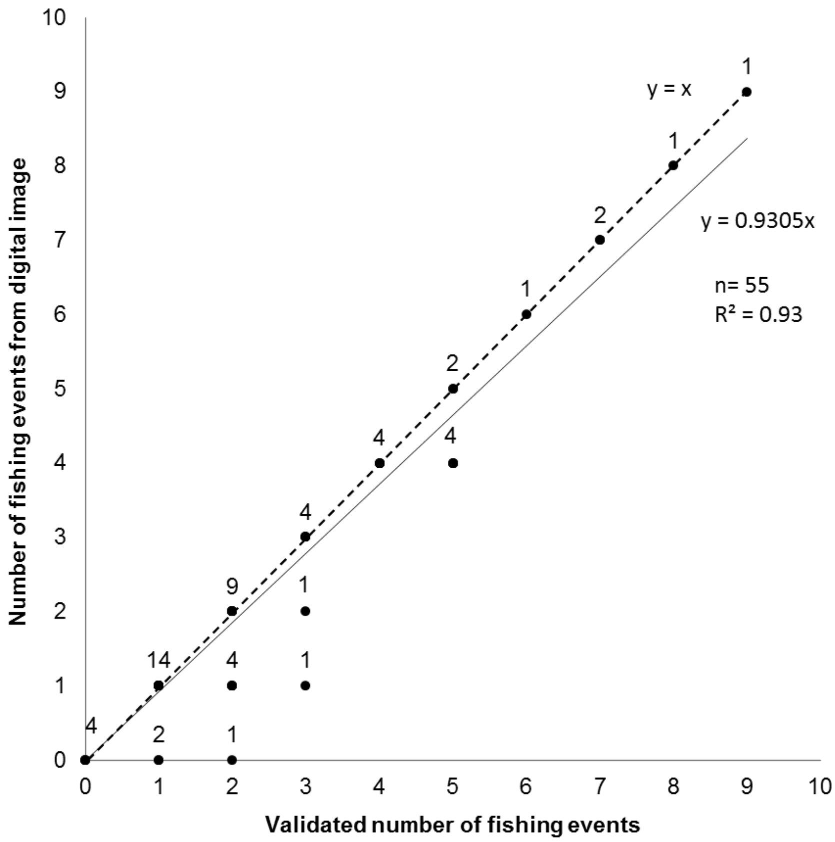


Validation of data derived from digital images

To investigate the potential bias and errors resulting from using digital images, comparisons were made with independent counts of fishing events made by observers standing on the shore using standard binoculars (7 x 50 Tasco marine series, model 222YRZ). Observers were members of the local marine rescue volunteer group and they were trained to observe and count the numbers of vessels that remained at least 5 min within the previously described monitoring area. These observers also recorded the time each vessel arrived and departed. Observations made by the volunteers were regarded as accurate measures of fishing effort in the OAR area. Volunteer observations of fishing effort were done on 55 randomly selected days between November 2012 and June 2014 and covered all weather conditions.

A linear regression forced through the origin was fitted using estimates of daily fishing events derived from digital images on the y-axis and the observer validated number of daily fishing events on the x-axis. This regression analysis can provide evidence of bias in estimates of fishing events derived from the digital images if the slope of the regression line differs significantly from 1.0. To determine if any bias existed, a two tailed *t*-test was used to test whether the sample value of b (i.e. the slope) as different from the expected value of 1 (Sokal and Rohlf 1981). When bias is detected, it is possible to derive a correction factor using the regression coefficient and its variance derived from the "variance of a quotient" equation. This correction factor was used to adjust the estimates of fishing effort (fishing events) for all strata.

Figure 23 **Regression equation describing the relationship between the number of fishing events counted from the camera digital images and from field validated observations at the OAR. Number of overlaying observations is provided above each data point. The dotted line denotes the y = x equation.**



Effort estimation from digital images

Stratified random sampling methods were used to select a sample of daily digital images for processing. Days were the primary sampling unit for all strata. A survey day started at 0600 h and ended at 1800 h. Years were stratified into Austral seasons and day types (weekdays or weekend days) within seasons. Sample sizes for each base level are given in Table 10. Whenever possible, each month within a season was allocated an equal number of each day-type. Mean daily fishing effort values (i.e. daily mean of the mean number of fishing events per day) and variances for each day-type stratum within each season were calculated. The first survey year spans from June 2012 to May 2013 and the second survey year lasted from June 2013 – May 2014. The total fishing effort for each day-type stratum was calculated by multiplying the mean daily fishing effort values by the number of possible sample days (N). Day-type stratum totals were added together to obtain seasonal totals and the seasonal totals were summed to obtain annual estimates.

Table 10 The number of days sampled (n) and stratum size (N) within day-type, weekdays (WD) and weekend days (WE), during the survey period.

Season/Year	Day-Type	Days sampled (*n*)	Number of days in stratum (*N*)
Winter 2012	WD	10	64
	WE	10	28
Spring 2012	WD	15	64
	WE	15	27
Summer 2012-13	WD	15	60
	WE	15	30
Autumn 2013	WD	15	63
	WE	15	29
Winter 2013	WD	15	64
	WE	15	28
Spring 2013	WD	15	64
	WE	15	27
Summer 2013-14	WD	15	61
	WE	15	29
Autumn 2014	WD	15	62
	WE	15	30

Fishing effort was converted from fishing events into boat hours and finally into fisher hours for comparisons with other studies. We multiplied the estimated total stratum effort (i.e. number of fishing events) independently for each stratum by the daily mean of the mean number of boat hours per fishing event for that stratum to obtain estimates of fishing effort in units of boat hours. To convert fishing effort in terms of boat hours to actual fisher hours we used data from a survey of coastal marine fishing outside the Port Hacking estuary between March 2008 and February 2009 (Steffe and Murphy 2011). The conversion was done by multiplying the boat hour estimates to the daily mean of the mean number of fishers per boat within each base level stratum. Pairwise comparisons of fishing effort (fisher hours) were made between seasons and years.

Standardized comparisons of effort intensity

Standardized comparisons of effort intensity per unit of area were made between the Sydney OAR and three other artificial reefs in coastal waters of South Australia (McGlennon and Branden 1994) and 14 estuarine fisheries in NSW (Table 11). Standardized values of fishing intensity were calculated for each fishery by dividing the total effort (fisher hours) by area in square kilometres. This allowed the boat based fishing intensity at the OAR to be benchmarked against these other Australian recreational fisheries. It was assumed that the patterns of fishing effort and the average number of fishers per boat per day within the study area had not changed.

Table 11 Study site survey periods, habitat types, distance from shore and location (latitude and longitude) and measured areas (km^2).

Survey location	Survey period	Habitat type	Distance from shore & depth	Latitude	Longitude	Area (km²)	Source
Sydney artificial reef	Jun 2012-May 2013; Jun 2013-May 2014	Untreated steel designed reef	1.2 km, 38 m	33°50.80'S	151°17.99'E	0.06	This study
Grange artificial reef	Sep 1990-Aug 1991	Tyre modules (1,200)	4.3 km, 15 m depth	34°55.1'S	138°24'E	0.08	McGlennon & Branden, 1994
Glenelg artificial reef	Sep 1990-Aug 1991	Tyre modules (900)/ sunken vessels (2)	5 km, 18 m depth	34°58.8'S	138°26.4'E	0.19	McGlennon & Branden, 1994
Port Noarlunga artificial reef	Sep 1990-Aug 1991	Tyre modules (650)	2.5 km, 18 m depth	35°05.2'S	138°26.5'E	0.07	McGlennon & Branden, 1994
Northern Lake Macquarie	Mar 1999-Feb 2000; Dec 2003-Nov 2004	All estuarine habitats	-	33°02.0'S	151°37.0'E	60.73	Steffe *et al.*, 2005b
Southern Lake Macquarie	Mar 1999-Feb 2000; Dec 2003-Nov 2004	All estuarine habitats	-	33°06.0'S	151°35.0'E	43.10	Steffe *et al.*, 2005b
Swansea channel	Mar 1999-Feb 2000; Dec 2003-Nov 2004	All estuarine habitats	-	33°04.35'S	151°38.40'E	3.23	Steffe *et al.*, 2005b
Tweed River	Mar 1994-Feb 1995	All estuarine habitats	-	28°14.38'S	153°32.42'E	20.25	Steffe *et al.*, 1996
Richmond River	Mar 1994-Feb 1995	All estuarine habitats	-	28°52.24'S	153°32.7'E	25.85	Steffe *et al.*, 1996
Clarence River	Mar 1994-Feb 1995	All estuarine habitats	-	29°27.35'S	153° 9.39'E	101.37	Steffe *et al.*, 1996
Brunswick River	Mar 1994-Feb 1995	All estuarine habitats	-	28°31.95'S	153°32.0'E	1.58	Steffe *et al.*, 1996
Sandon River	Mar 1994-Feb 1995	All estuarine habitats	-	29°41.05'S	153°18.17'E	1.49	Steffe *et al.*, 1996
Wooli River	Mar 1994-Feb 1995	All estuarine habitats	-	29°57.31'S	153°09.09'E	2.17	Steffe *et al.*, 1996
Mooball Creek	Mar 1994-Feb 1995	All estuarine habitats	-	28°25.75'S	153°33.33'E	0.40	Steffe *et al.*, 1996
Tuross estuary	Mar 1999-Feb 2000; Dec 2003-Nov 2004	All estuarine habitats	-	36°03.80'S	150° 6.08'E	14.47	Steffe *et al.*, 2005a
Hawkesbury estuary	Mar 2007-Feb 2008; Mar 2008-Feb 2009	All estuarine habitats	-	33°33.0'S	151°20.15'E	120.81	Steffe and Murphy, 2011
Port Hacking estuary	Mar 2007-Feb 2008; Mar 2008-Feb 2009	All estuarine habitats	-	34°04.31'S	151°09.30'E	11.51	Steffe and Murphy, 2011
Manning River	Mar 2007-Feb 2008; Mar 2008-Feb 2009	All estuarine habitats	-	31°53'14'S	152°39'13'E	25.35	Bucher, 2006

Comparative coastal fishing effort data from the greater Sydney region

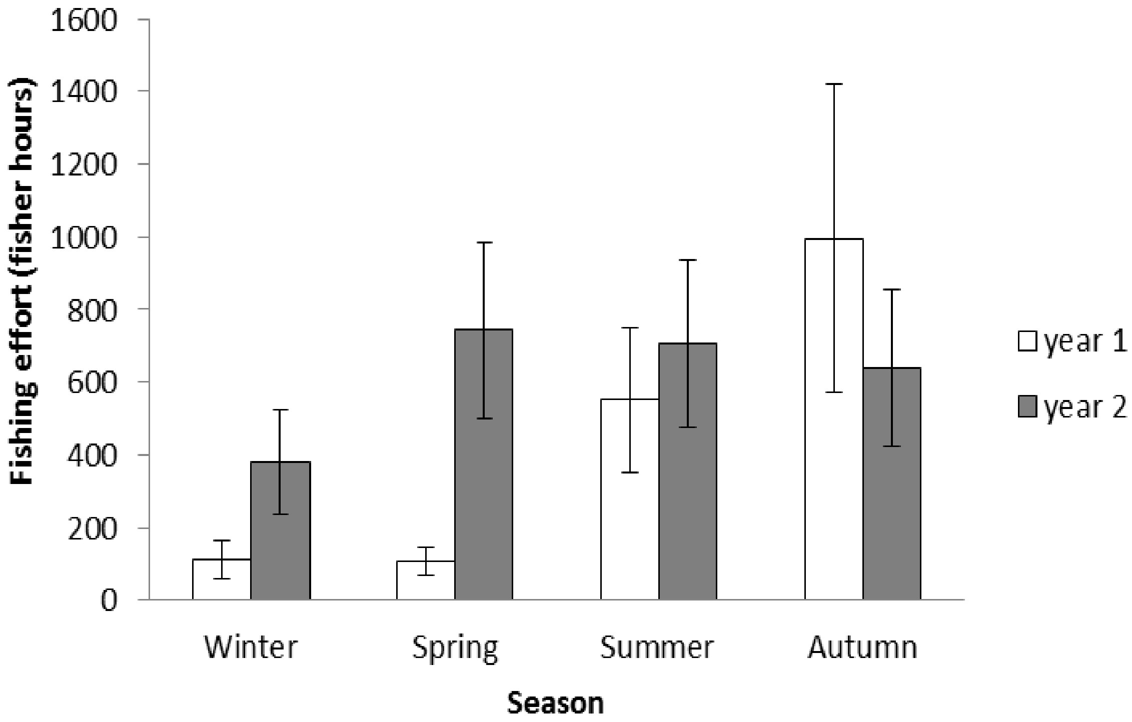
Comparisons in the seasonal fishing effort estimates from the OAR with the coastal fishing effort from the Sydney region were also made. Total coastal fishing effort (angling trips ± S.E.) was calculated using unpublished data from a survey of coastal marine fishing originating from the Hawksbury River, Port Hacking, Botany Bay and Sydney Harbour conducted between March 2007 and February 2009. Counts of boats returning from sea were made by observers from headland vantage points to each of these sites. All counts started one hour after sunrise and ended at sunset. Each survey year was stratified into seasons and day types within seasons (weekday and weekend day). Days were the primary sampling unit, with sampling done on 9 weekdays and 9 weekend days within each season at each of the four sites.

Results

The validation study provided evidence that the data derived from the camera images was significantly biased (see Figure 23), therefore fishing effort estimates derived from camera images were adjusted to compensate for the underestimation caused by weather-related changes in detectability of vessels. This visibility bias resulted in an underestimate of roughly 7.5% in the levels of fishing effort. Given this, all effort estimates and measures of precision were adjusted to correct this bias.

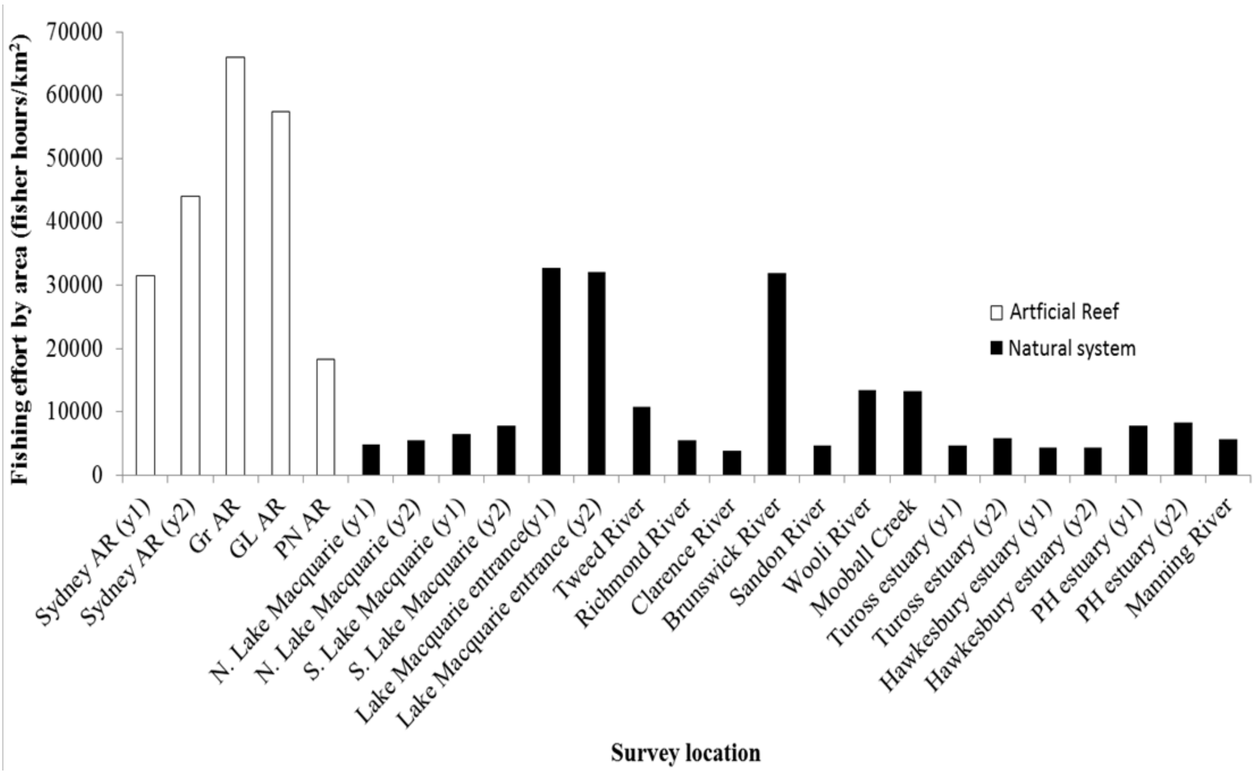
We estimated 1765 fisher hours during June 2012 – May 2013 (year 1) and 2460 hours during June 2013 – May 2014 (year 2 - Figure 24). These annual estimates of effort did not differ between years. However there were some seasonal differences in fishing effort. For example, fishing effort in spring of year 2 was greater than during spring of year 1. All other pairwise comparisons of seasons between years showed no differences in fishing effort.

Figure 24 **Seasonal fishing effort (fisher hours ± S.E.) at the Sydney OAR for each survey year (year 1 = June 2012 – May 2013, year 2 = June 2013 – May 2014).**



In the first year of the survey little fishing was observed during winter and spring, however fishing effort significantly increased during summer and autumn (Figure 24). In the second year of the survey there were no differences among each of the seasons.

Figure 25 **Comparison of annual fishing effort by area (fisher hours/km²) between study locations. Sydney AR (this study); South Australia ARs (Gr AR = Grange artificial reef, GL AR = Glenelg artificial reef, PN AR = Port Noarlunga artificial reef (McGlennon and Branden 1994); N. Lake Macquarie = Northern Lake Macquarie, S. Lake Macquarie = Southern Lake Macquarie, Lake Macquarie entrance = Swansea channel (Steffe *et al.* 2005b); Tweed River, Richmond River, Clarence River, Brunswick River, Sandon River, Wooli River and Mooball Creek (Steffe *et al.* 1996); Tuross estuary (Steffe *et al.* 2005a); Hawkesbury estuary (Steffe and Murphy 2011); PH estuary = Port Hacking estuary (Steffe and Murphy 2011); Manning River (Bucher 2006). For details of survey periods see Table 11.**



The OAR received 31,525 and 44,116 fisher hours per square kilometre during the two survey years respectively (Figure 25). Effort intensity comparisons between the three South Australian artificial reefs (McGlennon and Branden 1994) and the Sydney OAR showed that effort intensity was 2.1 time higher at the Grange than the Sydney OAR in year 1 of the survey and 1.5 times greater than the second survey year. However, effort intensity was higher at the Sydney OAR than at the Port Noarlunga artificial reef in both survey years.

Annual effort intensity was higher at the Sydney OAR compared to most of the estuarine fisheries from NSW included in this study (Figure 25). Effort intensity during both survey years at the OAR was between 5.8 and 9.1 times greater than effort intensity at northern Lake Macquarie, and between 4.1 and 6.8 times greater than at southern Lake Macquarie. For the Tweed, Richmond, Clarence, Sandon, Wooli and Mooball, Tuross, Hawkesbury and Port Hacking estuaries, effort intensity was between 2.3 and 11.7 times greater at the OAR (Figure 25). In contrast, effort intensity was similar between the first year of the OAR survey and Lake Macquarie entrance (Swansea channel) and Brunswick River. However during the second year of the survey, fishing intensity at the OAR was greater than at both years Swansea channel was surveyed and Brunswick River.

Discussion

This study demonstrates that shore-based camera systems are effective for monitoring changes in recreational fishing effort at near-shore artificial reefs. An important finding from this work was the importance of validating data derived from camera images due to changes in weather conditions (e.g. fog, rainfall, wind) which all can affect the ability to view vessels in digital photos, ultimately leading biased data. Similar findings have been observed in other studies where fishing effort was underestimated due to visibility bias (Smallwood, Pollock *et al.* 2012). By validating our digital images with trained, on site observers, we found that our fishing effort estimates were 7.5% lower than the true value. Future studies that rely on camera technologies to capture effort information for recreational fisheries should routinely include a validation component with the minimal additional cost required to undertake such an evaluation far outweighing the potential risk of basing management decisions on less accurate data.

The utility of camera systems for monitoring recreational fishing effort can be expected to increase as technological advances occur. The cost of high quality lenses are likely to decrease in the future allowing for enhanced digital image clarity and resolution. The ability of lenses to both optically and digitally zoom in a compact unit is also likely to increase, this is important as many artificial reefs are located at greater distances from the shore than the Sydney OAR (e.g. Shoalhaven artificial reef is ~ 9 km from the nearest vantage point). Also, improvements in the low light capacity of digital sensors will allow images from dawn, dusk and even in the dark for future studies.

The seasonal pattern of fishing effort observed during the two year survey period was influenced by both the length of time since the OAR was deployed, and more general seasonal patterns in fishing activity which is observed by fishers in the general region of NSW.

Monitoring of fishing effort at the OAR commenced 8 months after its October 2011 deployment. It is known that fish colonization on artificial reefs can occur rapidly following the initial deployment and the community can remain in a state of flux up to 10 years (Relini, Relini *et al.* 2002). Results from Chapter 2 indicate that the resident fish assemblage at the OAR was not fully established during the two years of survey conducted for this study period. This may have either discouraged recreational fishers initially, or they were unaware of the OAR, with this potential explaining the low levels of fishing effort recorded during the first winter and spring seasons (11% of the maximum). The pattern of seasonal fishing effort recorded after this initial period at the OAR closely resembled the known patterns of coastal fishing effort within this region, where generally effort decreases during the cooler winter and spring months.

Effort intensity recorded at the OAR was 31,525 and 44,116 fisher hours per square kilometre for years 1 and 2 respectively. This level of usage was up to 12 times greater than that recorded form many estuarine fisheries in NSW, but similar to the recreational fisheries in the Brunswick River and Swansea channel in Lake Macquarie. In comparison to other Australian artificial reefs, effort intensity at the Sydney OAR was greater than that reported at the Port Noarlunga OAR (18,310 fisher hours per km^2) but less than the artificial reefs at Grange (66,046 fisher hours per km^2) and Glenelg (57,505 fisher hours per km^2). This indicates that artificial reefs may concentrate fishing effort into small areas. Fish density can be higher on artificial compared to natural reefs (Ambrose and Swarbrick 1989; Bohnsack and Sutherland 1985), so these artificial reefs can be expected to concentrate fishing effort in the vicinity as anglers target fish assemblages associated with these structures.

The high levels of recreational fishing effort per square kilometre during both years of this study indicate that recreational fishing opportunities are likely to have been enhanced by the introduction of the OAR. This may be due to factors such as an increase in biomass as a consequence of additional food being provided by the artificial reef substrate, fish attraction and/or fish movements from adjacent habitats. Similarly, Santos and Monteiro (2007) found local fishing yields were higher at artificial reefs and that fish biomass was enhanced, particularly at protection reefs which provided shelter for fish.

The deployment of artificial reefs both in NSW and Australia more generally, are expected to increase in the future, with a large offshore artificial reef field just south of Sydney to be deployed by September 2017. Furthermore, a reef at Merimbula is in an advanced stage of planning, and an additional 4 reefs allocated funding for deployment. Monitoring of recreational fishing effort at future artificial reefs will form a central aspect of any evaluation process and camera based technologies provide a cost effective solution to monitor these fisheries.

Chapter 4 – Zooplanktivory as a pathway for fish production on the Sydney OAR – Curtis Champion, Iain Suthers, James Smith

Introduction

It is generally agreed that artificial reefs provide both habitat, space and food resources for fishes (Peterson *et al.* 2003; Powers *et al.* 2003), with this provision of refuge and food likely to drive the production of fish biomass on these reefs (Charnonnel *et al.* 2002; Powers, Grabowski *et al.* 2003). Thus, quantitative research is needed to understand how artificial reefs support fish production by supplying these factors.

Zooplanktivores can be extremely abundant on artificial reefs (Scott, Smith *et al.* 2015) and their capacity to continuously access zooplankton supplied by prevailing currents highlights that the provision artificial reef habitat may allow for increased production of zooplanktivorous fishes. These fishes are in turn preyed upon by larger piscivorous species (Young *et al.* 2010), and require the refuge provided by artificial reefs to forage the surrounding zooplankton. It is therefore likely that production on artificial reefs may largely be dependent on the direct and underappreciated trophic link between zooplankton and reef-resident zooplanktivorous fishes.

There is evidence to suggest that a balance of predation risk and foraging success influences the association of fish with the maximum distance from refuge habitat that prey fish will forage (Biesinger, Bolker *et al.* 2011). The distance that reef resident zooplanktivorous fish will forage from refuge determines the total volume of water surrounding a reef that is available to be foraged, and it is likely this distance is independent of reef size (Scott, Smith *et al.* 2015). Thus reef size would not scale linearly with the surrounding foraging volume, which has interesting implications for food availability and reef-resident zooplanktivore density.

The goal of this study was to explore the contribution of zooplanktivory to the production of fish biomass on artificial reefs and the management implications of this ecological process. Specifically, this study aimed to (1) describe the diet and habitat use of an abundant zooplanktivorous fish on a designed coastal artificial reef, (2) estimate the depletion of zooplankton due to predation by zooplanktivorous fish on this reef, and (3) illustrate the influence of artificial reef size on foraging volume and food availability for resident zooplanktivores.

Methods

Dietary analysis for *Atypichthys strigatus*

The fish species chosen for this study was Mado (*Atypichthys strigatus*), this is one of the most abundant species found on the OAR (see Chapter 2), is zooplanktivorous and common to reefs in temperate south-eastern Australia (Glasby and Kingsford 1994). The diet of *A. strigatus* was determined from 55 individuals which were captured at the OAR using hook and line techniques, with the total prey biomass and diet composition by mass quantified for each fish. Stomach contents were sorted into broad taxonomic groups and the wet weight values used to calculate the proportion of *A. strigatus* diet comprised of zooplankton.

Density and foraging volume of *A. strigatus*

The density of *A. strigatus* at the OAR needed to be calculated in order to estimate the total consumption of zooplankton. This was conducted using underwater video surveys with ‘drop cameras’ lowered to the reef from boat, these consisted of two outward facing GoPro™ cameras housed in a metal frame. Five replicate drop camera deployments lasting 10 min in duration were made on separate days to estimate the abundance of *A. strigatus*. Drop cameras could not be lowered inside the OAR, therefore two surveys using a remotely operated vehicle (ROV; Seamor Marine) were done, each lasting 30 min in duration. The average abundance of *A. strigatus* was estimated from 80, randomly selected still frames, selected from suitable drop camera and ROV footage.

To refine the spatial distribution of *A. strigatus* around the reef, the density of fish was partitioned into 0.5 m distance bins (e.g. 0.5 – 1 m from the reef). This was done by estimating the distance of each observed *A. strigatus* from the OAR using known distances between the reefs structural features. The total volume of water in each snapshot was partitioned into distance bins to estimate bin specific fish densities. The reef volume and surrounding foraging volume had to be determined to calculate total *A. strigatus* abundance from the previously estimated density estimates. The volume of water within the reef was calculated from engineering diagrams (hereafter 'reef volume'), and the surrounding volume as distance from the reef increased (hereafter 'foraging volume'). The volume of water held within 0.5 m foraging bins extending from the structure were combined with bin-specific density estimates to calculate the abundance of *A. strigatus* within each bin.

Figure 26 ***Atypichthys strigatus*** **(Mado) is one of the most common fish observed at the Sydney OAR, the diet of this species principally consists of zooplankton (Photo: James Smith).**

Consumption by *A. strigatus*

Food consumption as a function of biomass was estimated using an empirical formula derived from published literature and included data on water temperature, caudal fin aspect ratio and the main feeding mode of the species (i.e. carnivore, detritivore etc.). For more detail see Palomares and Pauly (1998). Data on caudal fin aspect ratio was calculated from 53 captured *A. strigatus* across its size range, and water temperature was measured on each sampling day using a SBE 19-plus V2 SeaCAT Profiler CTD (Sea-Bird Electronics Incorporated, Washington, USA) at depths relevant to the reef (24-38 m).

Zooplankton supply and availability

The supply of zooplankton was measured using 50 – 200 m plankton tows up-current of the OAR to ensure sampled zooplankton had not been exposed to consumption by resident fish on the structure. A 40 cm diameter, 100 µm mesh plankton net was towed horizontally from a boat for 4 min per tow, 15−20 m from the surface. The distance of each tow was calculated using a GPS, and each tow sampled ~25 m3 of seawater. Three replicate tows were done per sampling day (n = 27). Plankton samples were preserved with 5% formaldehyde immediately after collection. A laser optical plankton counter was used to sort the samples and estimate the number and biomass (mg m^{-3}) per size class (Suthers *et al.* 2004). Mean current velocities were used to estimate the supply of zooplankton per unit time to the foraging volumes of *A. strigatus*.

These velocities were obtained using a mechanical flow meter attached to the reef, which measured average velocity at 10 min intervals for June – August 2013 and from December 2013 – February 2014. The zooplankton size selection by *A. strigatus* was determined from the stomach contents of the 55 individuals that were dissected. Zooplankton were placed into 200 µm size class bins, the proportions of zooplankton size classes found in fish stomachs were compared to the plankton found in the tows to refine the biomass of zooplankton available for consumption by *A. strigatus*.

Model 1: Zooplankton depletion

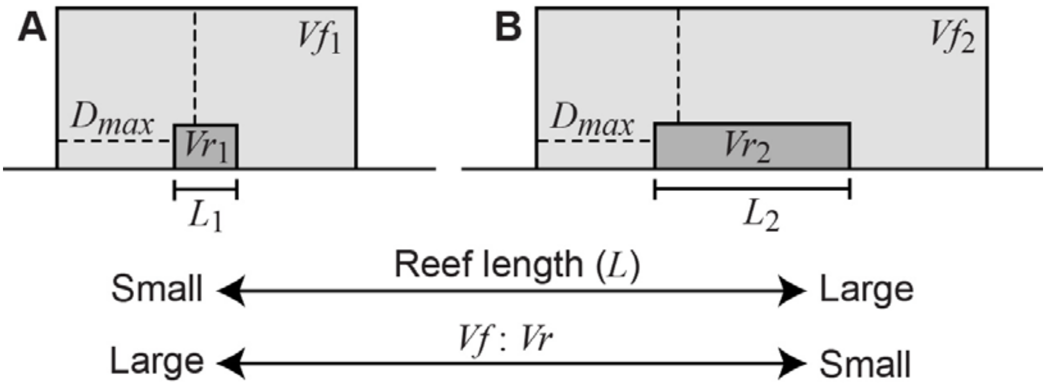
A numerical model was developed to estimate the depletion of zooplankton caused by *A. strigatus* predation at the OAR. This model predicted the depletion of each size class of zooplankton by incorporating zooplankton biomass, current velocity, and the consumption rate, density, and foraging volumes of *A. strigatus*. For more detail on the modelling of zooplankton depletion see Champion *et al.* (2015).

Model 2: Reef size

A general model was developed to identify how the size of the OAR influences the availability of food relative to the availability of habitat for reef-resident zooplanktivorous fish such as *A. strigatus* (Figure 27). Specifically, this model was used to identify how artificial reef size influences the ratio of the total zooplankton supply (food supplied) to the required consumption of a reef's maximum zooplanktivore population (food required). This model focused on the relationship between reef size and foraging volume, which is non-linear because the maximum foraging distance (D_{max}) for reef residents using the reef as a refuge is a constant that is generally independent of reef size (Biesinger, Bolker *et al.* 2011). Model 2 was also designed to explore management implications associated with artificial reef design, given that reef size is relative to reef construction cost.

This general model can apply to any reef-resident zooplanktivorous fishes, i.e. those that use the reef structure for refuge and forage the surrounding pelagic environment, although the model depends on species-specific traits such as average body size and swim speed. The zooplankton biomass and current velocity input values were selected to reflect the harshest ecological conditions observed during the survey period (i.e. lowest density of available zooplankton, and slowest observed daily current velocity) in order to conservatively estimate output values. To create the most general model possible, it was assumed that the internal reef volume was not part of the foraging volume (whereas fish could forage for zooplankton inside the reef in Model 1). This is true for concrete artificial reefs such as those deployed at Shoalhaven, Port Macquarie and soon to be deployed south of Sydney, which, unlike the Sydney OAR, are not designed to encourage water to flow through the structure. Like Model 1, this model used a Monte Carlo simulation to include parameter variation in the model. For details on these Monte Carlo simulations and parameters used in Model 2 see Champion *et al*. (2015).

Figure 27 Conceptual schematic of the changing relationship between food availability and refuge availability with a changing reef size. Food availability depends on the foraging volume (V_f; light grey), and refuge availability depends on the reef volume (V_r; dark grey); and the ratio between these (V_f:V_r) declines as reef size increases. Per-capita food availability is highest when reef volume is small and foraging volume relative to reef volume (the V_f:V_r ratio) is large (A). Refuge is most abundant when reef volume is large (B), but foraging volume relative to reef volume is small and food limitation will, at some point, limit the density of fish. This declining Vf:Vr ratio occurs because the foraging distance D_{max} – the distance fish will generally forage from shelter based on predation risk (Biesinger, Bolker *et al.* 2011) – is independent of reef size. A hypothetical example is two square reefs of height 1 m, with L_1 = 1 m and L_2 = 3 m, and D_{max} = 4 m. This gives Vf_1:Vr_1 = 404 for the smaller reef, which declines to Vf_2:Vr_2 = 66.2 for the larger reef.



Monte Carlo simulation and Sensitivity Analysis

Both Model 1 and Model 2 contained parameters with uncertain or variable values, so a Monte Carlo simulation was used to incorporate this parameter variation into model outputs. Each model was iterated 5000 times, and the mean and variance calculated from these iterations. Each parameter's sampling distribution was either a normal, lognormal, or beta distribution. The lognormal was used when there was evidence the data were skewed, and the beta distribution was used for proportion data. Standard deviations were generated from raw data wherever possible, and standard errors were used when the goal was to generate error in estimating the population mean, rather than variation at the individual level.

Results

Diet of A. strigatus

The diet of *A. strigatus* mostly consisted of zooplankton (approximately 93% of the diet; Table 12), when this was broken down, copepods were found to exceed all other identifiable prey groups combined (Table 12). Non-zooplanktonic items formed only a minor component of diets consisting of < 17% of the total occurrence. While much of the food was too digested for positive identification, other dietary studies of *A. strigatus* support the dominance of zooplankton in their diet (Glasby and Kingsford 1994). The parameters were included in the model to account for this uncertainty (see Champion, Suthers *et al.* 2015) and took values between 60 – 100% during simulations. The total mean wet biomass of *A. strigatus* stomach contents was 466.4 mg (± 39.3 SE), or 0.013 mg food per mg *A. strigatus* (± 0.001 SE).

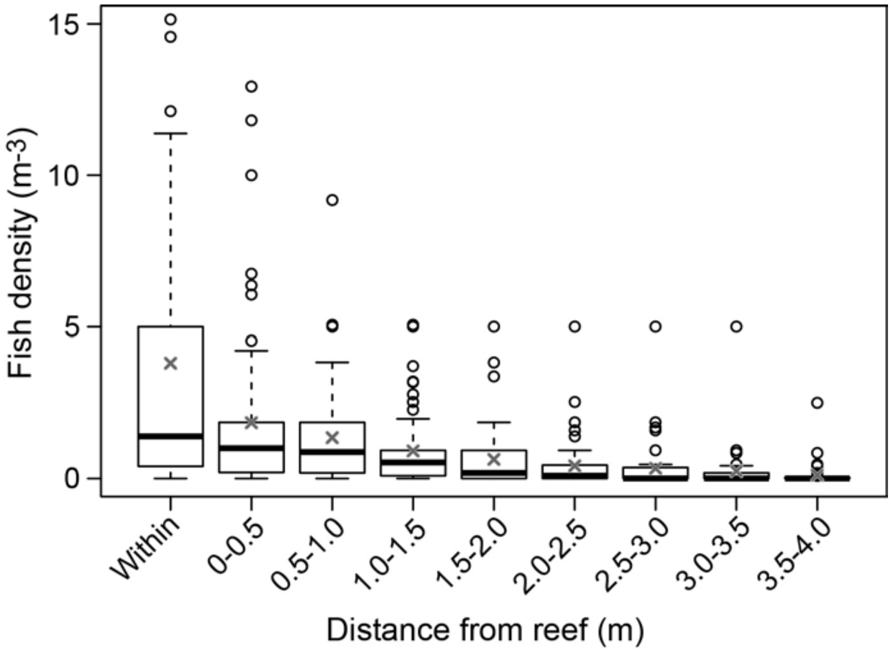
Table 12 Mean proportion (± S.E.) by wet mass of prey items in the diet of *A. strigatus* (*n* = 55) * Excludes unidentifiable material.

Prey category	Proportion of *A. strigatus* diet	Occurrence (%)
Copepod	0.160 (0.012)	100
Shrimp	0.032 (0.006)	65.5
Ostracod	0.010 (0.002)	52.7
Amphipod	0.002 (0.001)	10.9
Zoea	0.014 (0.004)	41.8
Nauplius	0.001 (0.001)	5.5
Gastropod	0.009 (0.005)	10.9
Mollusc	0.013 (0.009)	5.5
Plant material	0.010 (0.004)	16.4
Unidentifiable Crustacea	0.167 (0.011)	100
Unidentifiable	0.583 (0.017)	100
Total zooplankton	**0.929* (0.022)**	-

Foraging volume and density of *A. strigatus*

The density of *A. strigatus* declined as distance from the Sydney OAR increased (Figure 28) and were never observed more than 4 m from the structure. The density observed within the reef (3.81 *A. strigatus* m^{-3} ± 0.67 S.E.) greatly exceeded the mean density of all foraging bins surrounding the reef (0.73 *A. strigatus* m^{-3}). The foraging volume available to *A. strigatus* increased non-linearly with increasing distance from the reef. Reef volume was calculated as 663.6 m^3 and the total foraging volume (within reef plus surrounding volume to 4 m from reef) was 2879 m^3.

Figure 28 Quantile boxplots of densities of *A. strigatus* (fish m^{-3}) at the artificial reef within specific foraging distance bins. Each value represents a single observation from video footage (n ≈ 80 for each binned foraging distance). Grey crosses denote mean values.



Consumption by *A. strigatus*

At the OAR, *A. strigatus* were estimated to consume 8.88 times their biomass on average annually (Q/B), which equated to an average of 0.77 g (0.10 S.E.) *A. strigatus*$^{-1}$ day^{-1}. As caudal fin aspect ratio was not significantly correlated with fish size (r = 0.21, p = 0.14, n = 53), the mean aspect ratio of *A. strigatus* (2.29 ± 0.47 S.D.) was applied to the multiple regression model. Mean individual body mass of *A. strigatus* was calculated as 33.9 g (1.6 S.E.; n = 55).

Zooplankton supply and availability

Although the smallest size of zooplankton (< 200 and 200 – 400 µm) being the most abundant, *A. strigatus* selectively preyed upon food items between 601 – 800 µm. The abundance of zooplankton prey items in size classes larger than 601 – 800 µm declined in approximate proportion to the environmental availability.

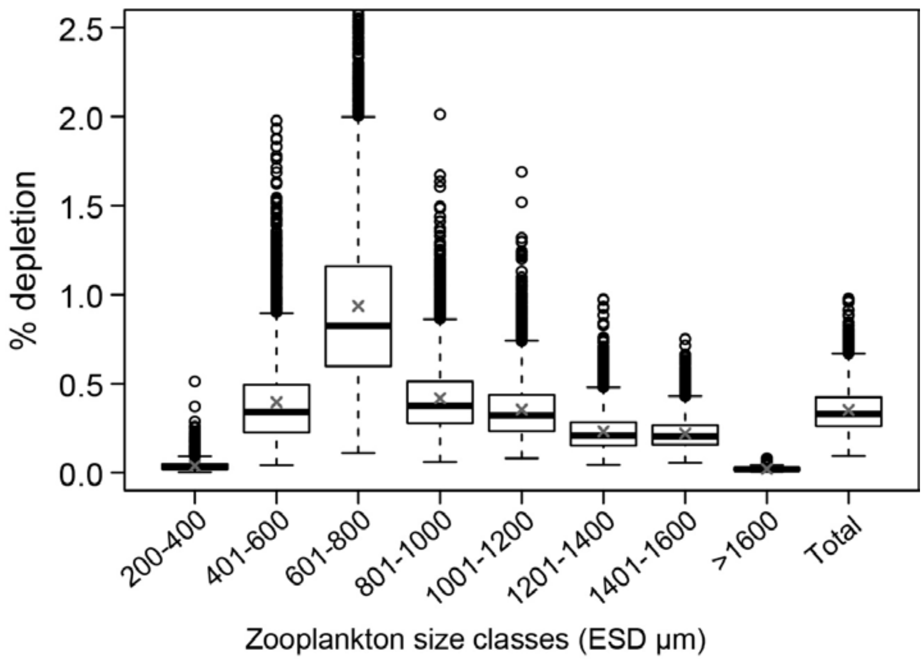
The observed mean total biomass of zooplankton up-current of the artificial reef was 871 mg m^{-3} (± 168 S.E). The observed size-selection of *A. strigatus* for zooplankton < 200, 200-400 and 401-600 µm was determined to equal zero, 0.02 and 0.46, respectively, while selection for size classes ≥ 601-800 µm were assumed all equal to 1. Thus, the mean biomass of zooplankton available to *A. strigatus* at this artificial reef under the observed pattern of prey size-selection was 637 mg m^{-3} (± 109 S.E.), which represented 73 % of the total zooplankton biomass.

Mean daily current velocity at the artificial reef between June-August 2013 and December-February 2013/2014 was 0.091 m s^{-1} (± 0.04 S.D.). Maximum and minimum daily current velocities observed during this period were 0.178 and 0.042 m s^{-1} respectively.

Model 1: Zooplankton depletion

The *A. strigatus* population at this study's coastal artificial reef was estimated to consume 2906 g (± 425 S.D.) of zooplankton per day, which is approximately 1.0 g per m^3 of reef habitat (reef and foraging volumes combined) per day. This equates to an average depletion of 0.35 % (± 0.13 S.D.) of the total zooplankton biomass delivered to the artificial reef (Figure 29), or 0.49 % (± 0.17 S.D.) depletion of the available zooplankton biomass. The size-specific analysis of zooplankton depletion revealed that size class 601-800 µm were most depleted by *A. strigatus* predation; equal to 0.94 % (1521 g per day) of the total biomass of that size class (Figure 29). Zooplankton within size classes 200-400 and > 1600 µm were least depleted by *A. strigatus* predation (excluding size class < 200 µm, which was not consumed by *A. strigatus*), equal to 0.04 % of the total binned biomass or 30 g per day, and 0.02 % of the total binned biomass or 20 g per day, respectively. There was considerable variation in these depletion estimates (Figure 29), due to the variation and uncertainty in the foraging volume, the supply of zooplankton, and Mado density and consumption. Sensitivity analysis showed that all these model facets are equally influential in the model.

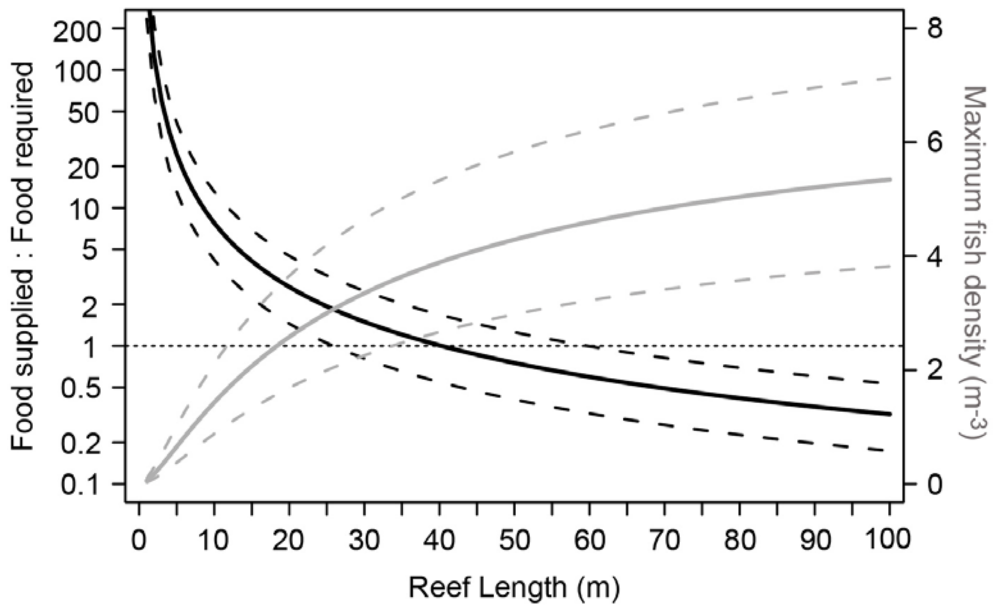
Figure 29 **The percentage of total zooplankton biomass depleted per size class by *A. strigatus*; grey crosses denote mean values. The spread of these quantile boxplots illustrates the variation in zooplankton depletion across the 5000 iterations in the Monte Carlo simulation of model 1.**



Model 2: Reef size

As the modelled reef became larger, foraging volume per unit reef volume decreased, which resulted in: (1) higher possible densities of reef-resident zooplanktivorous fish in the surrounding water due to the increase refuge; and (2) a corresponding decline in zooplankton availability (Figure 30). Given minimum observed values for current velocity and zooplankton biomass, the zooplankton supplied to reefs larger than approximately 40 m in length (between 25 - 55 m) would not support the required consumption rate of the maximum density of the resident zooplanktivore *A. strigatus* (Figure 30). The biomass of zooplankton available for consumption on reefs smaller than 40 m in length increased exponentially with decreasing reef size. The maximum density of the resident zooplanktivore in the foraging volume increased asymptotically with reef size (Figure 30). These results are for square reefs 4 m in height, and increasing this height would decrease the reef length at which food limitation begins. Changing reef shape would also change these relationships. As in Model 1, there was considerable uncertainty around the estimates due to variation and uncertainty in model parameters. The sensitivity analysis likewise showed that all parameters are equally influential for determining the reef size at food limitation.

Figure 30 **An illustration of the influence of reef size on food (zooplankton) availability and maximum density of resident zooplanktivorous fish. Food availability is the ratio of food supplied to food required (black lines; mean and 95% CI; axis is log_{10} scaled; C_{ratio}, equation 14), which declines as reefs get larger. When this ratio falls below 1 (dotted line), there is not enough food to support the maximum density of fish. The maximum density of zooplanktivorous fish in a reef's foraging volume (Den_L, equation 8) increases with reef size (grey lines; mean and 95% CI). This suggests that there is an intermediate range of reef sizes that offer large abundance of resident zooplanktivores while avoiding food limitation.**



Discussion

This study has shown how the Sydney OAR supports a large biomass of zooplanktivorous fish, and an important link between zooplankton and fish biomass on artificial reefs. Few *A. strigatus* foraged more than a couple of meters from the artificial reef, highlighting their close association with the structure and that, without the refuge it provides, would unlikely be able to exploit the zooplankton resources which are supplied by prevailing currents. Despite an estimated 3800 *A. strigatus* with a biomass of 130 kg populating the Sydney OAR, their total consumption depleted less than 0.5% of the prevailing supply of zooplankton. Given this, it might seem logical that increasing the size of the OAR would support a greater biomass of zooplanktivores, ultimately leading to greater benefits for recreational fishers. However, this study shows that doing so would provide more refuge volume for zooplanktivores, but relatively less foraging volume to support the increased consumption by a larger population, ultimately limiting its size.

Zooplankton consumption by reef fish

The large proportion of zooplankton in the *A. strigatus* diet, and their high abundance and resident behaviour, suggests that the supply of food from the pelagic environment is a very important driver in the function of the Sydney OAR. The zooplanktivore *A. strigatus* population was estimated to consume a large amount of zooplankton at this artificial reef (2.9 kg day^{-1} ± 0.5 S.D.), yet this was only a tiny proportion of the average total zooplankton biomass supplied to the reef (0.35 %).

A number of zooplanktivorous fish are common at the Sydney OAR, in particular Yellowtail Scad (*T. novaezelandiae*) is also found in high abundances (Becker *et al.* 2017; Scott, Smith *et al.* 2015). It is therefore likely the zooplankton consumption estimated in this study only represents part of the energy that is transferred from the pelagic environment to the OAR via zooplanktivory. Animals other than fish, such as the sessile invertebrates which attach to the structure of the OAR, these invertebrates can also consume large quantities of zooplankton (Ayukai 1995; Glynn 1973), the combined consumption of zooplankton by reef-associated fishes and invertebrates forms a ‘pelagic pathway’ of energy to the artificial reef (Cresson *et al.* 2014; Kingsford and MacDiarmid 1988). This synthesis of energy and its transfer to higher trophic levels is a key process driving biomass production on the OAR and artificial reefs more generally (Leitão 2013; Lindberg 1997).

Influence of reef size on zooplanktivorous fish

Reef associated planktivorous fish will seek refuge within the complex structures of reefs and forage in the immediately surrounding water volume, thereby limiting their exposure to predation (Hamner *et al.* 1988; Motro *et al.* 2005). A maximum foraging distance (D_{max}) defines this foraging volume surrounding reef habitats. An interesting outcome of a fixed D_{max} is a non-linear relationship between reef size and foraging volume. As a result, when reef size increases, the foraging volume surrounding a reef declines relative to the reef refuge volume. This non-linearity was found to have an important influence on the dynamics of reef-resident zooplanktivorous fish. The reef system modelled in this study suggests reefs greater than ~ 40 m in length have insufficient foraging volume to feed the maximum density of zooplanktivorous fish. Essentially, large reefs have refuge for a high abundance of fish, but lack the foraging volume in which to feed them. This means that the trophic relationship between zooplankton and reef-resident zooplanktivorous fishes becomes increasingly inefficient with increasing reef size, and the density of zooplanktivores should decline due to food limitation. Studies have observed higher fish densities on smaller reefs than on larger reefs (Bohnsack, Harper *et al.* 1994; Jordan, Gilliam *et al.* 2005), providing support for this relationship.

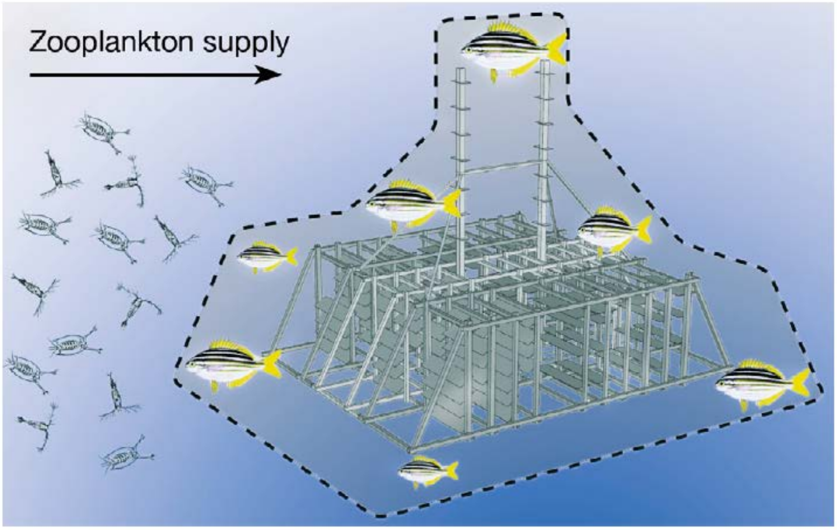
Management Implications

This study has shown the trophic link between zooplankton and zooplanktivorous fishes is an important avenue of energy for reef assemblages and probably contributes much to the fish production on coastal artificial reefs. However, the efficiency of this link can be influence by the size of the artificial reef. Both refuge volume and foraging volume are drivers of fish abundance, but the non-linear relationship between them means that small reefs can have an abundance of food but little refuge, while large reefs can have lots of refuge but insufficient foraging volume to support all possible residents. This suggests that an optimum reef size exists that can successfully trade-off between food and refuge. Understanding the limitations of artificial reefs that are either too small or overly large is essential for designing reefs that effectively facilitate the important trophic link between zooplankton and reef-resident fishes. This is important as larger reefs cost more to construct, yet may not optimise the transfer of energy from zooplankton to reef-resident zooplanktivorous fishes. A way to overcome this problem is to create reef fields that consist of clusters of multiple smaller reefs which collectively have the same refuge volume as a single large reef, yet have a much larger foraging volume (Brandt and Jackson 2013). This strategy would be beneficial as it is likely to result in a similar abundance of individual reef-resident zooplanktivorous fishes, but a larger per-capita food supply. The orientation of these reef clusters could be positioned so they are perpendicular to prevailing currents and maximising the supply of zooplankton for zooplanktivorous fishes.

Management objectives of artificial reefs are often to enhance fishing opportunities, optimising the production of zooplanktivores may promote such opportunities by increasing piscivorous fish which are generally targeted by anglers. The transfer of energy from zooplankton to local fish production on artificial reefs across multiple trophic levels remains to be quantified (Grossman *et al.* 1997), but it is likely that zooplanktivory by resident fish is a dominant process contributing to a larger food web.

Ways to facilitate the consumption of zooplankton by zooplanktivorous fishes should be considered when planning future reefs, including the manipulation of reef size and shape, as this trophic link may have the greatest potential for enhancing the production of fish biomass from artificial reefs.

Figure 31 **Schematic diagram of the association of zooplanktivores with the Sydney artificial reef.**



General Discussion

The artificial reef project has been based on the success of evidence based testing of objectives scaled from deployments of relatively small estuarine based structures, to now large scale offshore reef fields. The Sydney OAR was the first purpose built, offshore reef in NSW, and established a clearly defined project blueprint for future reef proposals that importantly incorporated close research links to academic institutions, provided the opportunity for NSW DPI to comprehensively investigate the response of fish to the placement of the structure, as well as the levels of secondary productivity resulting from the deployment. This revealed a succession of fish species on the OAR over time, differences between the OAR and natural reefs, the close spatial association fish have with the structure, and movements of fish between the OAR and other reefs in region. Additionally, the usage by the public was demonstrated through angler surveys, and the increase in local productivity due to the deployment of the OAR was modelled through zooplanktivory. Together this represents the most detailed and thorough examination of an artificial reef in Australia to date.

The broader artificial reef program was initially undertaken in large estuaries and consisted of deploying Reef Balls® beginning in 2005. This provided a number of important lessons, including the importance of a thorough pre-deployment program to develop goals, select site locations as well as the post deployment monitoring required to evaluate the artificial reef (Lowry, Folpp *et al.* 2010). These lessons were subsequently taken into consideration from the very beginning of the offshore artificial reef program. Essentially, the process remained the same, however due to the proposed size of the structure and the need to position it in offshore waters, a more complex planning and post-deployment monitoring program was required.

The planning process of the OAR was extensive, but proved to be valuable in guiding the post deployment research program. Probably the most important step in the planning process of the offshore reef was to identify the need, which was then used to develop clear and well defined goals for which the OAR needed meet over time. The location of the OAR was to be one which would benefit recreational anglers in trailer boats, providing additional fishing opportunities and locations. Selecting the final location took time, but the constraint mapping exercise and steps developed in the planning process for the Sydney OAR have now been incorporated into the planning procedures for subsequent reefs deployed by NSW DPI (e.g. Shoalhaven, Port Macquarie and South Sydney). The establishment of goals was critical in evaluating the success of the program through the research and monitoring programs outlined in this report. In this way, the goals themselves fundamentally directed much of the research undertaken, such as the BRUV monitoring program, evaluation of the pelagic fish assemblage and the estimation of angler participation rates. The suggested methods outlined in the EMMP were largely followed in each of the research programs. There were some exceptions; for example, there was no sampling of the area prior to the deployment of the structure. While this would have been ideal, the EMMP did recognise that all the sampling requirements identified were unlikely to be undertaken. That being said, all biological and socio-economic priority 1 monitoring objectives were conducted and are outlined in this report, and generally followed the suggested sampling frequency and duration. Future reef deployment would benefit from a similarly detailed planning and evaluation process.

Outcomes of the research presented in Chapters 2 – 4 point to the success of the OAR in meeting the original goals of the deployment. Multiple lines of evidence support this conclusion, for example, fish rapidly colonised the structure, and as expected, the assemblage underwent changes over time which were due to successional processes. Importantly, key recreational species were continually observed over time in both benthic and pelagic BRUV footage, with length data showing many were greater than the minimum legal length, so could be harvested by anglers. Participation rates by recreational anglers suggest the OAR is a popular fishing location for people based in Sydney. Although not identified as an original goal for the deployment of the OAR, modelling identified a trophic pathway in which fish biomass can be produced through zooplanktivory. While these fish themselves are generally not targeted by recreational anglers, they are prey items for larger pelagic piscivores such as Yellowtail Kingfish. More importantly, this research also showed how the design, specifically the size of the structure, may influence the biomass of zooplanktivores through the provision of the optimum ratio of refuge and foraging space. Together this body of research not only demonstrates the OAR met the goals for which it was designed and deployed, but also provides valuable information in the design of future reefs and the monitoring programs which would likely be required to evaluate these deployments. Already NSW DPI is in the final planning stages for multiple new artificial reefs stretching the entire NSW coast in coming years.

By far, the bulk of the research undertaken on the Sydney OAR was biological in nature, with the exception of the angler usage surveys detailed in Chapter 3. A review of the literature reveals this is a common pattern worldwide, following the deployment of artificial reefs (Becker, unpublished research). An area which was not investigated, or identified in the planning stages, was a thorough economic evaluation of the OAR. An understanding of commonly used economic measures in recreational fisheries, such as willingness to pay, would be an interesting and valuable addition to the existing research. As an increasing number of reefs are deployed along the coast of NSW, how these effects patterns in fishing activity, both at artificial reef sites and existing popular locations, would provide valuable information which would assist with future planning for deployments.

Conclusions and recommendations

The Sydney reef has led the way for the NSW offshore artificial reef program, with reefs now deployed in Shoalhaven, Port Macquarie and soon at southern Sydney, another reef is in the advanced planning stage offshore of Merimbula and due for deployment in the second half of 2018. Multiple reefs are also planned with funding approval up until 2020. The expansion of the artificial reef program is possible due to the ability of the Sydney OAR project to demonstrate it met its pre-deployment goals and provided tangible benefits to the recreational anglers of NSW. Recommendations from the work conducted at the Sydney OAR include:

1) A clear statement of a need and associated development of clear goals for the deployment of an artificial reef is a vital first step. Failing to develop any goals will result in a lost opportunity for a robust evaluation of the reef, and the ability to demonstrate any benefit derived from its deployment.

2) Ideally, long-term monitoring is required to properly evaluate the response of reef fish assemblages to the deployment of offshore artificial reefs. Successional processes at the Sydney OAR led to inter-annual changes in the fish assemblage which were still apparent four years after the OAR was lowered into the water. Monitoring of reefs in some capacity (e.g. BRUVS, rapid drop camera, sonar surveys, acoustic receivers), even at low temporal resolution should be undertaken for at least 3 years as successional processes will result in a rapidly changing assemblage.

3) Pelagic camera recordings revealed fish generally have a close spatial relationship with the structure of the Sydney OAR. Although the Sydney OAR consisted of only a single large structure, future artificial reefs will most likely consist of a ‘field’ of smaller concrete and/or steel structures. Findings from this study suggest reef units as close as 60 m will avoid overlapping distributions of associated fish, while promoting connectivity. Future work further evaluating the distribution of fish around structures over longer monitoring periods and assessing other variables such as biomass would be insightful.

4) Shore based camera systems are effective for monitoring changes in recreational fishing effort for artificial reefs positioned closed to the coast. Improvements in camera technology in the future will allow for long-range cameras to observe OARs located further offshore. The ability to monitor recreational fishing effort is an important step in demonstrating the success of artificial reefs and should be incorporated where practical. Our understanding of recreational fishers behaviour to the deployment of artificial reefs is an area where future research needs to be focused. The location of many artificial reefs makes this difficult but the advancement in camera technology should open new opportunities.

5) The size of the structure needs to be carefully considered and provide a mix of refuge and foraging habitats, particularly for zooplanktivorous species. Future OARs could maximise both these habitat types by incorporating a design consisting of multiple structures creating reef fields.

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Appendices

Appendix 1 – Peer reviewed publications arising from work outlined in this report

Becker, A., Taylor, M.D., Lowry, M.B. (2017) 'Monitoring of reef associated and pelagic fish communities on Australia's first purpose built offshore artificial reef' *ICES Journal of Marine Science* 74: 277-285.

Scott, M.E., Smith, J.A., Lowry, M.B., Taylor, M.D., Suthers, I.M. (2015) 'The influence of an offshore artificial reef on the abundance of fish in the surrounding pelagic environment' *Marine and Freshwater Research*, 66: 427-437.

Keller, K., Smith, J.A. Lowry, M.B., Taylor, M.D., Suthers, I.M. (2017) 'Multispecies presence and connectivity around a designed artificial reef' *Marine and Freshwater Research*, 68: 1489-1500.

Keller, K., Steffe, A.S., Lowry, M.B., Murphy, J.J., Suthers, I.M. (2016) 'Monitoring boat-based recreational fishing effort at a nearshore artificial reef with a shore-based camera' *Fisheries Research*, 181: 84-92.

Champion, C, Suthers, I.M., Smith, J.A. (2015) 'Zooplanktivory is a key process for fish production on a coastal artificial reef' *Marine Ecology Progress Series*, 541: 1- 14.

Other titles in this series

No. 1	Andrew, N.L., Graham, K.J., Hodgson, K.E. and Gordon, G.N.G., 1998. Changes after 20 years in relative abundance and size composition of commercial fishes caught during fishery independent surveys on SEF trawl grounds. Final Report to Fisheries Research and Development Corporation. Project no. 96/139.
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No. 5	Kennelly, S.J. and Broadhurst, M.K., 1998. Development of by-catch reducing prawn-trawls and fishing practices in NSW's prawn-trawl fisheries (and incorporating an assessment of the effect of increasing mesh size in fish trawl gear). 18pp + appendices.
No. 6	Allan, G.L. and Rowland, S.J., 1998. Fish meal replacement in aquaculture feeds for silver perch. 237pp + appendices.
No. 7	Allan, G.L., 1998. Fish meal replacement in aquaculture feeds: subprogram administration. 54pp + appendices.
No. 8	Heasman, M.P., O'Connor, W.A. and O'Connor, S.J., 1998. Enhancement and farming of scallops in NSW using hatchery produced seedstock. 146pp.
No. 9	Nell, J.A., McMahon, G.A. and Hand, R.E., 1998. Tetraploidy induction in Sydney rock oysters. 25pp.
No. 10	Nell, J.A. and Maguire, G.B., 1998. Commercialisation of triploid Sydney rock and Pacific oysters. Part 1: Sydney rock oysters. 122pp.
No. 11	Watford, F.A. and Williams, R.J., 1998. Inventory of estuarine vegetation in Botany Bay, with special reference to changes in the distribution of seagrass. 51pp.
No. 12	Andrew, N.L., Worthington D.G., Brett, P.A. and Bentley N., 1998. Interactions between the abalone fishery and sea urchins in New South Wales.
No. 13	Jackson, K.L. and Ogburn, D.M., 1999. Review of depuration and its role in shellfish quality assurance. 77pp.
No. 14	Fielder, D.S., Bardsley, W.J. and Allan, G.L., 1999. Enhancement of Mulloway (*Argyrosomus japonicus*) in intermittently opening lagoons. 50pp + appendices.
No. 15	Otway, N.M. and Macbeth, W.G., 1999. The physical effects of hauling on seagrass beds. 86pp.
No. 16	Gibbs, P., McVea, T. and Louden, B., 1999. Utilisation of restored wetlands by fish and invertebrates. 142pp.
No. 17	Ogburn, D. and Ruello, N., 1999. Waterproof labelling and identification systems suitable for shellfish and other seafood and aquaculture products. Whose oyster is that? 50pp.
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No. 19	Otway, N.M. and Parker, P.C., 2000. The biology, ecology, distribution, abundance and identification of marine protected areas for the conservation of threatened Grey Nurse Sharks in south east Australian waters. 101pp.
No. 20	Allan, G.L. and Rowland, S.J., 2000. Consumer sensory evaluation of silver perch cultured in ponds on meat meal based diets. 21pp + appendices.
No. 21	Kennelly, S.J. and Scandol, J. P., 2000. Relative abundances of spanner crabs and the development of a population model for managing the NSW spanner crab fishery. 43pp + appendices.
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