The influence of an offshore artificial reef on the abundance of fish in the surrounding pelagic environment

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Abstract. Artificial reefs are a popular fisheries management tool, but the effect of these reefs on the abundance of fish in the surrounding pelagic environment is uncertain. Pelagic baited remote underwater video (PBRUV) was used to observe the fish assemblage surrounding an offshore artificial reef (OAR), near Sydney, Australia. PBRUVs were deployed at three distances (30, 100, 500 m) from the OAR, and compared with a drop camera deployed directly over the OAR. There was a significantly greater abundance of fish on the OAR, but no significant difference in abundance at the 30-, 100- or 500-m distances. Two highly mobile non-resident species (*Seriola lalandi*, *Pseudocaranx dentex*) were significantly more abundant on the OAR, but this association was not detected 30 m away. The lack of a significant difference in total fish abundance, or in assemblage composition, between the 30-, 100- and 500-m distances suggests that any association with the OAR is on a localised scale (<30 m). One exception was the ocean leatherjacket (*Nelusetta ayraudi*), which had an association detected 100 m from the OAR. This predominantly small-scale effect may be influenced by the proximity of this OAR to numerous natural reefs.

Additional keywords: BRUV, *MaxN*, pelagic fish, PERMANOVA, *Trachurus*.

Introduction

Artificial reefs are structures deployed on the sea floor, often with the objective of enhancing fisheries. Artificial reefs can address limiting factors in the environment such as food availability or space for settlement, shelter or reproduction (McCawley and Cowan 2007; Relini *et al.* 2007; Hackradt *et al.* 2011), and can act as habitat for many marine organisms across a range of spatial scales (dos Santos *et al.* 2010). The ecological effects of artificial reefs, in particular the response of fish to reef deployment, is well studied in both estuarine (Burton *et al.* 2002; Duffy-Anderson *et al.* 2003; Folpp *et al.* 2011) and marine environments (Bohnsack and Sutherland 1985; Fabi *et al.* 2002; dos Santos *et al.* 2010). Although these studies focus primarily on reef-resident species, there is a need to understand the spatial scales at which fish interact with artificial reefs, particularly pelagic or non-resident fish. This is especially important for offshore artificial reefs (OARs), which are deployed in the open ocean, typically in deeper water, and are accessible to a variety of both resident and transient pelagic fish species.

Research on the distribution of fish assemblages around OARs and similar offshore structures generally identify localised associations. The highest abundance of fish is typically found directly over the reef (Boswell *et al.* 2010), with abundance declining to 'background levels' 20–50 m away (Stanley and Wilson 2000; Fabi *et al.* 2002; Boswell *et al.* 2010; dos Santos *et al.* 2010), although some studies have found artificial reef residents moving 100 m away (Topping and Szedlmayer 2011), or detected an association 100–300 m away (Lokkeborg *et al.* 2002). The scale of the associated fish assemblage may depend on how an OAR is used by pelagic fish. Pelagic fish are often associated with physical structures including fish aggregating devices, floating logs, seamounts, and reefs (Hobday and Campbell 2009), and several ecological processes may explain these associations. These include geographic orientation, predator avoidance, school formation, feeding, and parasite cleaning (Castro *et al.* 2001; Consoli *et al.* 2013). The mechanisms are likely to be both species- and environment-specific, and research is needed to determine the spatial scales of the associations between the pelagic fish assemblage and OARs.

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 systems of artificial reefs (Jordan *et al.* 2005), including tailoring these distances to benefit target species. This study used pelagic baited remote underwater video (PBRUV) and an unbaited drop camera

to determine the scales at which fish associate with an OAR. The fish assemblage was sampled at <5 m (drop camera) and at 30, 100 and 500 m from the reef. The distribution of fish in the oceanic environment is likely to be influenced by abiotic factors (Bakun and Parrish 1991; Maravelias 1999; Freon *et al.* 2005), so this study assessed whether environmental conditions contribute to the association between the fish assemblage and the OAR.

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 the PBRUV as a sampling tool for these environments.

Materials and methods

Site description

The OAR was located at 33°50.797′S, 151°17.988′E, 1.2 km east of South Head, just outside Sydney Harbour (Supplementary material, Fig. S1), in 36 m of water. The dimensions of the OAR were 15 m by 12 m, and 12 m high (Supplementary material, Fig. S2). The substrate around the OAR was mostly flat and sandy for at least 550 m in all directions, with some areas of mostly low reef and rock platform close to the shoreline. The OAR's exposure to the eddies of the East Australian Current (Everett *et al.* 2012), and proximity to Sydney Harbour, means it could experience large fluctuations in temperature, salinity, nutrient input, and current velocity. The OAR was designed and deployed specifically to enhance local recreational fishing. It was deployed on 7 October 2011.

Underwater video

This study used two types of underwater video to observe the fish assemblage: PBRUV (Heagney *et al.* 2007) and unbaited drop cameras. Underwater video was an appropriate sampling tool for this study because it can be deployed in environments unsuitable for conventional diver-based assessments, can be used to identify fish to species level, provides a permanent visual record of a survey (Cappo *et al.* 2004; Harvey *et al.* 2007), and when used with bait can be used to survey patchy or mobile species (Lowry *et al.* 2012), which was expected for the pelagic assemblage. 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 1 m 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. Pilchards are bait commonly used by anglers in eastern Australia and for baited camera studies (Heagney 2010; Hardinge *et al.* 2013; Taylor *et al.* 2013). The components were combined in a matrix of vegetable meal to create a continuous, fragmenting, bait plume for the duration of the PBRUV deployment.

The PBRUV could not be deployed directly above the OAR because of practical constraints, but the fish assemblage around the top of the OAR was sampled with an unbaited drop camera
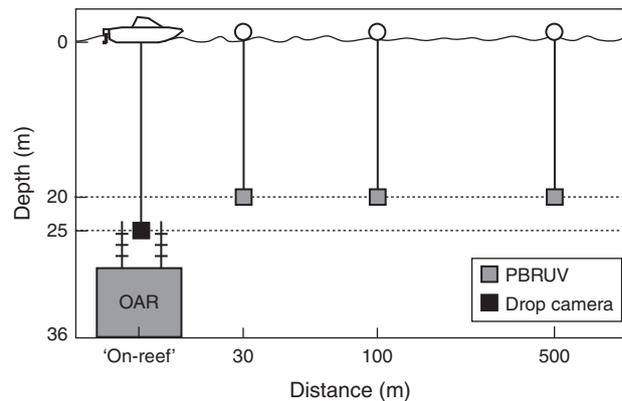

Fig. 1. The sampling design. Pelagic baited remote underwater video (PBRUV) was deployed at three distances from an offshore artificial reef (OAR), suspended from buoys to a depth of 20 m. The 'on-reef' distance was sampled using an unbaited drop camera deployed from a boat to a depth of 25 m. The PBRUVs were deployed shallower than the drop camera to remain >5 m above the shallowest parts of the sampling area.

(Seaviewer Sea-drop 650, FL, USA) with video recorded from a boat positioned above the reef (Fig. 1). This was sufficient to quantify species using the water column directly around the two spires of the OAR (see Supplementary material, Fig. S2), but not the cryptic species inside the reef's base. This suited the scope of the study which was to assess the spatial scale at which an associated fish assemblage could be detected, rather than provide a census of resident fish species. The implications of using two different sampling methods for analysing the distance treatment are discussed in *Data Analysis*.

Experimental design

Four distances were sampled (Fig. 1): 'on reef' (<5 m), 'near' (30 m), 'mid' (100 m), and 'far' (500 m). The three off-reef distances were based on previous studies that sampled similar distances, including 300–500 m away from artificial reefs (Boswell *et al.* 2010; dos Santos *et al.* 2010). The average bottom depth for the off-reef deployments was 37 m, with a maximum depth of 45 m. The on-reef camera was deployed at a depth of 25 m and the PBRUVs at a depth of 20 m (Fig. 1). The 20-m depth was chosen because it was similar to the depth of the top of the OAR (24 m) while remaining at least 5 m above the substrate in the shallowest parts of the sampling area.

Pelagic baited remote underwater video sampling was done over two summer seasons, with 10 days sampled between November 2011 and March 2012, and 5 days sampled between December 2012 and April 2013. The on-reef drop camera sampling was done on 6 days during the same sampling period. There were 33 and 552 days between the OAR's deployment and the first and last sampling days. This time frame was considered sufficient to encompass the initial recruitment and settlement of fish onto the OAR. Sampling days were arbitrarily selected, but were partially dictated by weather conditions. On each sampling day, two transects were selected and PBRUVs deployed at each of the three sampling distances along these transects. Transect bearings were chosen without reference to any feature (such as bottom depth) or oceanography (such as current direction), but

Table 1. The species observed during the study, and their total *MaxN* (the sum of each deployment's *MaxN* for that species)

Total *MaxN* is reported for each distance (on-reef; near, 30 m; mid, 100 m; far, 500 m), with a total of 76 pelagic baited remote underwater video (PBRUV) and six drop-camera deployments. Also reported are the number of deployments in which the taxa were observed (sightings), and the largest single *MaxN* observed for each species during the study

Species	Common name	Total *MaxN* 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 shark	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
Number of deployments		6	28	19	29		
Average total *MaxN* per deployment		257.8	36.7	47.3	27.8		

were selected haphazardly to get a broad spatial coverage while ensuring that every deployment was closer to the artificial reef than the nearest natural reef.

The order of PBRUV deployments in a sampling day was haphazard, and generally two PBRUV units were simultaneously deployed, between 0800 and 1100 hours. A total of four PBRUVs were deployed each sampling day for the first 5 days (the mid distance was not sampled), and six PBRUVs deployed each sampling day on the last 10 days. Some days had fewer deployments because of unfavourable weather, giving a total of 76 PBRUVs deployed during the study. PBRUVs were deployed for 45 min, on the basis of previous research that suggests this duration is sufficient to survey pelagic fish distribution and abundance (Heagney *et al.* 2007). In total, six drop cameras were deployed, each for a duration of 15 min.

All fish observed in the footage were identified to species (Table 1). Oceanographic characteristics were recorded at 20 m depth using a SeabirdCTD (Sea-Bird Electronics, Inc., Bellevue, WA, USA). Parameters analysed were water temperature, salinity, turbidity, and current velocity. Current velocity was recorded using data from the Ocean Reference Station (ORS), managed by the Integrated Marine Observing System (IMOS). The ORS (ORS065) is an oceanographic mooring in 65 m of water, located ~5 km SSE of the OAR, and measures current using an Acoustic Doppler Current Profiler (ADCP). Current velocity at the ORS has since been shown to be highly correlated ($R^2 = 0.8$; 24-h moving average) with velocity measured by an ADCP moored <1 km from the OAR (N. Ribbat, pers. comm.). During the study, the oceanographic parameters varied as follows: water temperature, 15.5–23.8°C; salinity, 34.9–35.5; turbidity, 0.14–0.27 ntu; current velocity, 0.05–0.67 m s^{-1}.

Calculation of MaxN

A metric of fish abundance (*MaxN*) was estimated from the video footage from each PBRUV and drop camera deployment. *MaxN* is the maximum number of a fish species observed in a single frame of footage during one deployment (Willis *et al.* 2000), and is the abundance metric most often used with underwater video (Cappo *et al.* 2004; Gladstone *et al.* 2012; Taylor *et al.* 2013). It is a conservative estimate of abundance, but avoids counting the same individuals more than once (Willis and Babcock 2000). *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 (Løkkeborg 1998); but, assuming this proportion is constant, *MaxN* is sufficient to enable a test of a treatment effect on relative abundance. 'Total *MaxN*' is a metric of total abundance and is calculated by summing the *MaxN* for every species observed in a deployment.

Data analysis

MaxN represented abundance in this study, but needed to be standardised to account for variation in sampling effort (Taylor *et al.* 2013). The ability of the PBRUV to attract and detect fish (i.e. effort) was considered to be influenced by visibility and current velocity. Visibility affects *MaxN* by altering the distance at which fish can be identified in video footage, and it was estimated that the horizontal viewing distance varied between 6 and 12 m during the study period. Varying visibility can be mitigated by counting only those fish that move in front of a reference marker attached near the PBRUV's bait canister (Willis and Babcock 2000), but these might only be a small

fraction of the visible fish. Alternatively, visibility can be standardised amongst deployments (Lowry *et al.* 2012). This was the method used in this study, and turbidity (ntu) was used as a surrogate of visibility. Current 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 *et al.* 2013). Bait plume area can be calculated as:

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

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}). This equation has been adapted from other approaches (Sainte-Marie and Hargrave 1987; Hill and Wassenberg 1999; Heagney *et al.* 2007). 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 *et al.* 2013). This was done because of the considerable uncertainty involved in calculating the area of the bait plume, which is affected by a variety of interacting biotic and abiotic factors (Sainte-Marie and Hargrave 1987), and dependent on fine-scale current velocity data, which are often lacking.

Permutational multivariate ANOVA (PERMANOVA) (Anderson 2001) was used to test the effect of distance on the composition of the fish community, using Bray–Curtis similarity and with a dummy species added to control for deployments with zero observations. A similarity percentage analysis (SIMPER) (Clarke 1993) was done using *MaxN* ntu^{-1} data to identify which species were associated with the significant effects identified using PERMANOVA. PERMANOVA was done on all three datasets (*MaxN*, *MaxN* ntu^{-1}, *MaxN* ntu^{-1} m^{-2}) to demonstrate the consistency of the result. 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. The on-reef distance was included in all analyses (except the *MaxN* ntu^{-1} m^{-2} dataset, which is not applicable for unbaited sampling), and these were compared with analyses of the three distances sampled with PBRUV. This was a robust way of testing the effect of distance from the OAR on fish abundance. 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 (Harvey *et al.* 2007; Hardinge *et al.* 2013) 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.

A distance-based multivariate linear model (DistLM) was used to incorporate oceanographic variation in the analysis of distance on the observed fish assemblage. DistLM evaluates the relationship between a multivariate data cloud and one or more predictor variables using orthogonal matrices (Anderson *et al.*

2008). Oceanographic variables (temperature, salinity, and current velocity) were normalised and a draftsman plot evaluated for skewness in abiotic measurements and correlations between the variables. A DistLM was then calculated using these variables, plus distance as a categorical variable (i.e. distance and water quality variables analysed simultaneously within each depth), against a Bray–Curtis resemblance matrix of fourth-root transformed *MaxN* ntu^{-1} data. Only the PBRUV data were analysed, given the potential for current velocity to mediate a response between the fish assemblage and bait plumes. A dummy species was added to control for deployments with zero species observed.

Species accumulation plots were created to assess the PBRUV methodology and this study's sampling design, by evaluating the number of sampling days and deployment duration. A *post hoc* power analysis was done to examine this study's statistical power. Effect size was calculated for total *MaxN* and for the number of species, given this study's sampling effort (15 days), with power = 0.8, and α = 0.05. The standard deviation for both total *MaxN* and the number of species was calculated as the average standard deviation between distances within days, not including comparisons with zero fish observed.

Results

The distribution of pelagic fish around the offshore artificial reef

A total of 76 PBRUVs were deployed during 15 sampling days, and fish were observed in 42 of these (55%). There was an average 'time-to-first-fish' of 7.4 min. Fish <5 cm in length were unable to be identified to species and were grouped together as 'juvenile fish'. A total of 2768 individuals from 14 taxa were observed in the PBRUV, with six of these observed only once (Table 1). A total of 1547 individuals and seven species were observed in the on-reef deployments, including four species that were also observed in the surrounding pelagic environment. Fish were observed in all six on-reef deployments, with an average *MaxN* up to one order of magnitude greater on the OAR than at the treatment distances (Fig. 2; Table 1). The
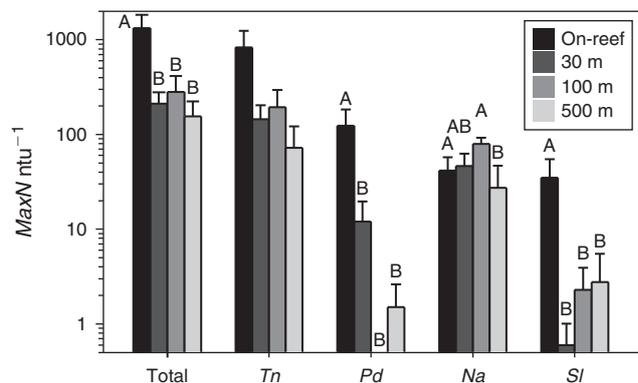

Fig. 2. Average (±s.e.) *MaxN* ntu^{-1} between the four treatment distances (on-reef, 30 m, 100 m, 500 m), for all species observed (Total) and the four species observed both on the reef and in the surrounding pelagic environment: *T. novaezelandiae* (*Tn*), *P. dentex* (*Pd*), *N. ayraudi* (*Na*), 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).

number of species observed at each distance was 7 (on-reef), 9 (near), 8 (mid), and 10 (far). The species *Trachurus novaezelandiae* and *Nelusetta ayraudi* were by far the most abundant species observed in the pelagic zone around the OAR (Table 1).

The PERMANOVA showed a significant effect of distance, when all four distances were analysed, for both *MaxN* and *MaxN* ntu^{-1} (Table 2). 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*, *Atypichthys strigatus*, *Pseudocaranx dentex*, and *Seriola lalandi* explaining an average of 21, 21, 20, 17 and 11% respectively of the dissimilarity between the on-reef assemblage and the surrounding distances.

One-factor ANOVA reported a significant effect of distance on total fish abundance (total *MaxN* ntu^{-1}, $F_{3,81} = 5.61$, $P < 0.01$) (Fig. 2), but there was no difference in abundance between the three off-reef distances (Fig. 2), 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 them (Fig. 2) – *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.

Contribution of oceanographic variables

A distance-based multivariate linear model (DistLM) showed no effect of distance from the OAR on the fish community, given the variation explained by the oceanographic variables, for the three distances sampled with the PBRUV (Table 3). Both sequential tests (each predictor variable is additive to the model) and marginal tests (the contribution of individual variables is measured) showed that temperature and salinity explained a significant amount of the variation in the pelagic assemblage. This sequential model explained 26% of the variation (Table 3).

Assessment of pelagic BRUV

The statistical power of this study and the PBRUV were assessed using a *post hoc* power analysis. The average standard deviation between distances was 28.3 fish (total *MaxN*) and 0.69 species. Thus, with power = 0.8, and $\alpha = 0.05$, this study's sampling effort of 15 days was able to detect an increase between the experimental distances of 44.3 fish (total *MaxN*), and 1.1 species. A large amount of the variance in *MaxN* was due to the schooling pelagic species *T. novazelandiae*, and, if this species is removed, the standard deviation decreases to 13.3 fish, and the detectable effect size of total *MaxN* decreases to 20.8 fish. A species accumulation curve for the average number of species observed during a PBRUV deployment showed that new species were still being observed after 45 min with no obvious asymptote (Fig. 3a). An accumulation curve for the duration of the study showed that new species were still being observed after 15 sampling days (Fig. 3b).

Discussion

Fish were generally associated with the OAR on a small scale (<30 m). This was true for a characteristic reef-resident (*A. strigatus*), which was observed only on the OAR itself, and also for two highly mobile non-residents (*S. lalandi*, *P. dentex*). One exception to this small-scale association was the ocean leatherjacket *N. ayraudi*, which showed a larger-scale association with the OAR (detectable 30–100 m away). Local variation in water temperature and salinity did explain some variation in the fish assemblage, but there was no evidence that this altered the effect of distance to the artificial reef. The total abundance of the fish assemblage at <5 m from the OAR was 5–9 times greater than that found 30 m away. Although this study was non-ideal by comparing abundance data from unbaited (on-reef) and baited cameras (off-reef) (see *Confounding factors*), it showed that the near (30 m), mid (100 m) and far (500 m) fish assemblages were not significantly different, so any reef effect is likely to exist within 30 m of the reef. This finding agrees with other studies that have looked at the 'area of effect' of artificial reefs, and a distance of 20–50 m is often the boundary for a reef effect (Stanley and Wilson 2000; Fabi and Sala 2002; dos Santos *et al.* 2010). This has consequences for the management of artificial reefs, in particular the isolation of artificial reefs and the spacing between reef modules (Walsh 1985; Jordan *et al.* 2005),

Table 2.   Results of PERMANOVA

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**		NA	
	Residual	78	1438			1793.4					
30 m, 100 m, 500 m	Distance	2	1925.3	1.3486	0.223	2406.3	1.3389	0.242	204.69	1.7242	0.121
	Residual	73	1427.6			1797.3			118.72		

Table 3. Results of the distance-based multivariate linear model (DistLM) for *MaxN* ntu^{-1}

The predictor variables were the oceanographic variables (water temperature, salinity, and current velocity), and distance from OAR as a categorical predictor variable. Only the pelagic baited remote underwater video (PBRUV) data were analysed given the potential for current velocity to mediate a response between the fish assemblage and bait plumes. *MaxN* ntu^{-1} was fourth-root transformed. Marginal tests show the proportion of data explained by each variable singularly. Sequential results represent an additive model. Sequential model selection criteria: adjusted R^2 and stepwise. Reported is model significance, as well as the R^2 and cumulative R^2 (Cum. R^2). A $P < 0.05$ indicates a variable has a significant effect (shown in bold)

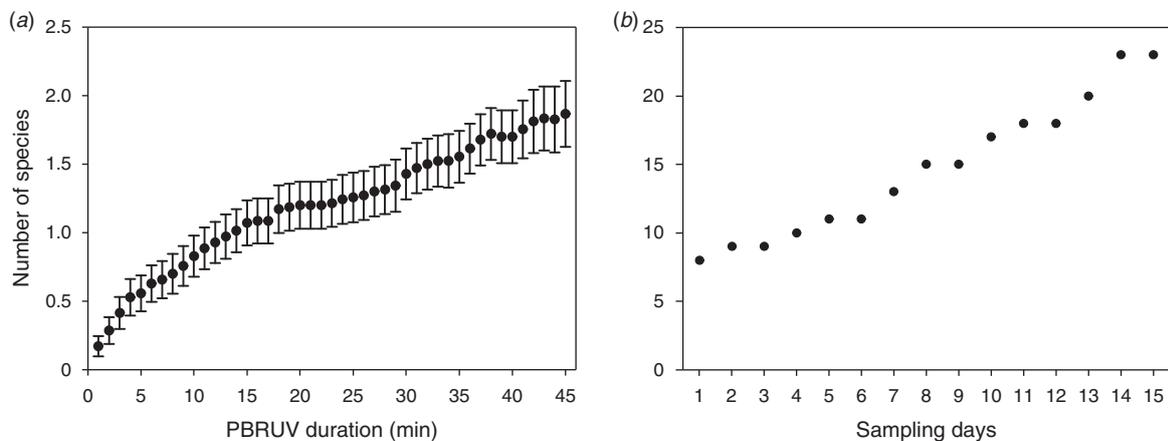| PBRUV only		Sequential				Marginal		
	s.s.	F	P	R^2	Cum. R^2		P	R^2
Salinity	17 549	10.96	**<0.01**	0.13	0.13	Temperature	**<0.01**	0.08
+Temperature	18 224	13.27	**<0.01**	0.13	0.26	Salinity	**<0.01**	0.13
+Distance	3686.5	1.35	0.22	0.03	0.29	Current	0.52	0.01
						Distance	0.23	0.03

Fig. 3. (*a*) Average species accumulation (±s.e.) as a function of pelagic baited remote underwater video (PBRUV) duration, and (*b*) species accumulation as a function of sampling days.

although caution would be used when applying findings from single reef systems to multiple-reef complexes.

The composition and distribution of the fish assemblage around the OAR

The observed fish assemblage around the OAR comprised fish with diverse habitat-use categorisations, from coastal pelagic (e.g. *T. novaezelandiae*, *S. lalandi*) and oceanic pelagic (*I. oxyrinchus*, *M. indica*), to reef-associated (*P. dentex*, *N. ayraudi*), reef-resident (*A. strigatus*), and even soft-bottom benthic (*P. caeruleopunctatus*). Many of the observed species were rare or not observed on the OAR, but four of the most abundant species on the OAR were pelagic or reef-associated species (Fig. 2). This mix is common for an offshore artificial structure (Fabi *et al.* 2002; dos Santos *et al.* 2010; Scarcella *et al.* 2011). It is likely that numerous reef-resident species went unobserved, especially cryptic ones, due to this study focusing on the scale at which a reef effect can be detected and not the identification of all species using the OAR. This meant that the on-reef drop camera focussed on the pelagic species around the top of the OAR (Fig. 1), so would not observe deeper or cryptic species.

Although some studies have found residents moving 100 m from a reef (Topping and Szedlmayer 2011), or increased abundances sometimes 100–300 m away (Lokkeborg *et al.* 2002), most studies find a distance between 20 and 50 m as the boundary for the association between fish and artificial reefs (Fabi and Sala 2002; dos Santos *et al.* 2010), even for large structures such as oil and gas platforms (Stanley and Wilson 1997, 2000; Boswell *et al.* 2010). Beyond this boundary, fish abundance is not significantly different from 'background levels' (Stanley and Wilson 1997). Localised associations may be the norm for reefs, and there is some evidence that this is a balance of predation risk and foraging success, at least for reef-associated fish (Frazer and Lindberg 1994; Biesinger *et al.* 2011; Biesinger *et al.* 2013). An association's spatial scale, if determined in part by swimming speed (Biesinger *et al.* 2011), would explain a small foraging area for prey fish that is largely independent of reef size.

The moderate size of this OAR and its close proximity to the shoreline and natural reefs (the nearest natural reef is 550 m away) may reduce the abundance of the associated fish assemblage. There is a positive relationship between reef isolation and fish abundance (Walsh 1985; Jordan *et al.* 2005; Vega Fernández *et al.* 2008), and it is possible that novel or stronger associations between fish species and the OAR could have been observed were it farther from other reefs. The present OAR was large for a purpose-built unit, but there are much larger artificial structures

deployed elsewhere (Stanley and Wilson 1997; Boswell *et al.* 2010). However, the 'area of effect' of reefs does not scale linearly with reef size (Jordan *et al.* 2005) and generally extends no farther than 50 m, even for large structures. So although a larger OAR or additional OAR units might increase the scale of the detectable association to 30 m, it would be unlikely to extend farther than 50 m. The species composition of the assemblage is potentially as important as reef size for determining the spatial scale of associations. The scale of associations are species-specific, and some species may have larger associations, e.g. 30–100 m for red snapper (*Lutjanus campechanus*) (McDonough 2009; Topping and Szedlmayer 2011) even with moderately sized reefs (Topping and Szedlmayer 2011).

Confounding factors

There was only one artificial reef in this study. Without replicate reefs of similar design and location, it is difficult to validate the observed patterns in fish distribution, as these patterns may be site-specific. Further purpose-built reefs are planned for New South Wales waters, however, including multiple-reef clusters, and future research of these reef deployments will benefit from quantifying the distribution of fish around multiple reef units.

This study may have biased results due to the two different sampling methods used (Fig. 1). Compared with the unbaited drop camera, the baited PBRUVs would be expected to generally have higher total abundance (Hardinge *et al.* 2013), with an increased abundance of predatory and scavenging species, and equal numbers of herbivorous and omnivorous species (Harvey *et al.* 2007). Thus, the abundances of fish at the three off-reef distances were likely to be overestimates compared with the unbaited on-reef estimates. Given that cameras with and without bait have unequal efficiency at sampling specific components of the fish assemblage (such as gregarious, aggressive, or shy species), using both baited and unbaited cameras could enhance future studies measuring relative abundance. The drop cameras were also deeper and sampled for a shorter duration than the PBRUVs because of sampling constraints. Despite the non-ideal comparison of the two methods, the analysis of the off-reef distances was robust (in that the method was consistent across all distances) and provided evidence that the fish assemblage 30 m from the OAR was generally indistinguishable from that 100 or 500 m away. These distances were coarse as the spatial resolution of this study was a simplification of what is a three-dimensional system. An associated fish assemblage will have vertical (Friedlander and Parrish 1998; Anderson and Millar 2004) and horizontal characteristics that are difficult to quantify using cameras, and these were not tested in this study. See Supplementray material for a further evaluation of the BRUV method used in this study.

Sampling transects were selected without reference to potentially contributing factors, such as current direction or tidal state. By doing so, this study was unable to determine complex associations. Creating a factorial design to include these factors, however, was beyond the scope of this study, and would be logistically difficult given oceanographic variability. There is also potential for the duration between deployments, and especially the duration between sampling days, to generate variability in the response between distance from the OAR and fish abundance. A shorter period may reduce some variability, but it

would also make it more difficult to generalise a result. However, it is likely that there are important fine-scale spatial and temporal patterns that were not observed in this study.

Conclusion

Artificial reefs are used worldwide with a goal to enhance fisheries, yet it is unclear at which spatial scales fish use these reefs. This study showed that an associated fish assemblage is unlikely to be detected 30 m away from an OAR, even for associated pelagic species (e.g. *S. lalandi*). This scale is likely to depend on numerous factors, such as the size of the OAR, the species-specific composition of the assemblage, and, perhaps most importantly, the proximity of an OAR to other structures. It is possible that more isolated OARs will have higher species diversity and be used by a larger abundance of pelagic fish (Walsh 1985; Jordan *et al.* 2005; Vega Fernández *et al.* 2008), whereas highly connected OARs will have a higher abundance of reef-resident species (Vega Fernández *et al.* 2008). The proximity between artificial reef units within reef clusters is a key consideration for artificial reef research (Campbell *et al.* 2011), and the low vagility of reef-associated fish inferred from this study suggests reef units as close as 60 m will avoid overlapping distributions of associated fish, while promoting connectivity. Further research examining the role of predator abundance, visibility, and reef connectedness on the scale of associations is needed.

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