BRAZILIAN JOURNAL OF OCEANOGRAPHY, 59(special issue CARAH):55-67, 2011

COLONIZATION AND COMMUNITY DEVELOPMENT OF FISH ASSEMBLAGES ASSOCIATED WITH ESTUARINE ARTIFICIAL REEFS*

*Heath Folpp[1,3]**, Michael Lowry[2], Marcus Gregson[1] and Iain M. Suthers[3]*

[1]Recreational and Indigenous Fishing Division, NSW Department of Primary Industries
(Cronulla Fisheries Research Centre, 202 Nicholson Parade, Cronulla NSW 2230)

[2]Wild Fisheries Research Division, NSW Department of Primary Industries
(Port Stephens Fisheries Institute, Locked Bag 1 Nelson Bay NSW 2315)

[3]School of Biological, Earth and Environmental Sciences (BEES)
(University of New South Wales, Sydney NSW 2052)

**Corresponding author: heath.folpp@industry.nsw.gov.au

ABSTRACT

Despite the long history of the development of artificial structures in NSW estuaries there are no studies that provide any comprehensive scientific evaluation of post-deployment goals. We assessed the effectiveness of estuarine artificial reefs as a fisheries enhancement initiative; described the diversity and abundance of species associated with them, and detailed the patterns of colonization and community development associated with an artificial reef deployment in Lake Macquarie, a large coastal barrier lagoon on the southeast coast of Australia. Six artificial reefs (one artificial reef group), constructed from artificial reef units (Reef Balls®), were deployed in December 2005 and sampled six times per season over two years using baited remote underwater video (BRUV). Colonization of the artificial reef group was relatively rapid with the majority of species identified over the two-year study period observed within the first year post-deployment. Overall, 27 species from 17 families were identified. Key colonising species included *Pelates sexlineatus* (Terapontidae), *Acanthopagrus australis* (Sparidae), *Pagrus auratus* (Sparidae) and *Rhabdosargus sarba* (Sparidae). Species richness showed evidence of potential seasonal fluctuations, being higher in warm water months (Summer/Autumn), and lower in the colder water months (Winter/Spring), while species diversity increased significantly with reef age. Fish assemblage composition remained relatively stable after the first year of sampling, with few discernible patterns in assemblage structure evident after the first year. Distinct separation in reef age groupings was evident during the second year of sampling; a pattern primarily driven by a decrease in abundance of *P. sexlineatus*, a result of the isolated nature of the artificial reefs and the interrelated effects of density dependence and predation.

RESUMO

A despeito da longa história do desenvolvimento de estruturas artificiais nos estuários de NSW, não existem estudos que apresentem uma avaliação global sobre os efeitos obtidos com o estabelecimento dessas estruturas. No presente trabalho abordamos a efetividade dos recifes artificiais estuarinos como iniciativa para aumento da pesca; descrevemos a diversidade e abundância das espécies a eles associadas; descrevemos os padrões de colonização e o desenvolvimento das comunidades associadas a um recife artificial colocado no Lago Macquaire, extensa lagoa de barreira situada na costa sudeste da Australia. Seis recifes artificiais (formando um único grupo), construídos a partir de unidades artificiais (Reef Balls®), foram lançados em Dezembro de 2005 e amostrados seis vezes a cada estação do ano, durante dois anos, utilizando video subaquático remoto (BRUV). A colonização dentro do grupo de recifes ocorreu de maneira relativamente rápida, sendo que a maioria das espécies identificadas nos dois anos de estudo foi observada durante o primeiro ano de amostragem. Um total de 27 espécies pertencentes a 17 famílias foram identificadas. As espécies chave do processo de colonização foram *Pelates sexlineatus* (Teraponidae), *Acanthopagrus australis* (Sparidae), *Pagrus auratus* (Sparidae) and *Rhabdosargus sarba* (Sparidea). A riqueza de espécies mostrou evidência de sazonalidade, enquanto a diversidade aumentou significativamente com o aumento da idade do recife. A composição da assembléia de peixes permaneceu relativamente estável após o primeiro ano de amostragem, com poucos padrões identificáveis relativos à estrutura. Durante o segundo ano tornou-se evidente a formação de grupos por idade, padrão primariamente ocasionado pelo decréscimo na abundância de *P. sexlineatus*; por sua vez este decréscimo mostrou ser resultado da natureza isolada do recife artificial e dos efeitos interdependentes de abundância e predação.

Descriptors: Artificial reef, Estuary, Fish, Colonization, Community development, *Pelates sexlineatus, Acanthopagrus australis.*
Descritores: Recifes artificias, Estuário, Peixes, Colonização, Desenvolvimento da Ictiofauna, *Pelates sexlineatus, Acanthopagrus australis.*

(*) Paper presented at the 9th CARAH – International Conference on Artificial Reefs and Related Aquatic Habitats on 8-13 November, Curitiba, PR, Brazil.

INTRODUCTION

The use of specially designed prefabricated reef structures is now common-place in many countries such as Japan and Korea (KIM, 2001; SUTTON; BUSHNELL, 2007), although the potential ecological benefits of artificial reefs and their ability to enhance production continue to be debated (PICKERING; WHITMARSH, 1997; BORTONE, 1998; LINDBERG, 1997; OSENBERG et al., 2002). The colonization and development of fish communities associated with artificial structures has been described by several previous studies (CUMMINGS, 1994; BOHNSACK; TALBOT, 1980; ALEVIZON; GORHAM, 1989; BURCHMORE et al., 1985; LAUFLE; PAULEY, 1985; MATTHEWS, 1985; PERKOL-FINKEL et al., 2006; PERKOL-FINKEL; BENAYAHU, 2007; DEMARTINI et al., 1989; GORHAM; ALEVIZON, 1989; HAUGHTON; AIKEN, 1989; HUECKEL et al., 1989). However, very few artificial reefs have been constructed in shallow nearshore areas at depths of less than 6m (CUMMINGS, 1994; BOHNSACK; TALBOT, 1980; HAUGHTON; AIKEN, 1989), with even fewer built in estuaries (MARTIN; BORTONE, 1997; BORNTRAGER; FARRELL, 1992; BURTON et al., 2002; MANDERSON, 2003; BORTONE et al., 1994; FOSTER et al., 1994).

As a fisheries enhancement initiative, estuarine artificial reefs provide particular advantages for people inhabiting near shore coastal areas, providing proximal and economical access to fisheries resources when weather or seasonal conditions may not permit access to offshore waters (BORTONE et al., 1994). Recreational anglers are often the most vocal proponents of artificial reef developments due to perceived improvements in fishing quality associated with artificial structures. Increased catch rates linked to the deployment of artificial structures (BORTONE, 1998; GROSSMAN et al., 1997) have contributed to the ongoing debate amongst fisheries biologists and managers as to whether artificial reefs are acting to enhance production of the fishery or attracting fish from the surrounding area, thus making them more vulnerable to sources of fishing mortality (i.e. the attraction production debate) (BOHNSACK, 1989; PICKERING; WHITMARSH, 1997).

While it is clear that developments in the design of artificial reefs have resulted in artificial habitats that are closely aligned with the habitat requirements of the target species, research concerned with the ecological function of artificial reefs lags far behind design and construction (PICKERING; WHITMARSH, 1997; LINDBERG et al., 2006; MILLER, 2002). The majority of published studies on artificial estuarine habitats in Australia have involved assessments of non-design specific structures such as marinas' seawalls and wharves (BURCHMORE et al., 1985; CHAPMAN; BULLERI, 2003; CHAPMAN; CLYNICK, 2006; CLYNICK et al., 2008). The lack of detailed scientific assessment and the relatively poor understanding of 'design specific' artificial reefs located within estuarine systems highlight the need for a better understanding of the interaction between artificial reefs and their estuarine fish communities (CLYNICK et al., 2008).

The development of artificial reef use in Australia has been previously documented and reviewed in detail (BRANDEN et al., 1994; KERR, 1992; COUTIN, 2001; POLLARD, 1989; POLLARD; MATTHEWS, 1985). While these reviews have been mostly descriptive assessments of materials and developments through time, they have collectively highlighted the lack of detailed, post-deployment scientific assessment of artificial reef programs. The trajectory of artificial reef development within the Australian state of New South Wales (NSW) has followed a similar pattern to the evolution of artificial reef projects worldwide, with initial deployments using 'materials of opportunity', with little or no monitoring program to enable assessment of pre-deployment objectives. Our project represents the first use of design specific materials for the construction of estuarine artificial reefs in Australia and has provided an opportunity to: (1) assess the effectiveness of 'design specific' estuarine artificial reefs for the enhancement of recreational fisheries; and (2) describe diversity and abundance; and the patterns of colonization and community development associated with such structures.

MATERIALS AND METHODS

Study Area and Deployment of Estuarine Artificial Reef Structures

The study was carried out in Lake Macquarie, a coastal saltwater barrier lagoon located in southeast Australia (33°05'S, 151°36'E) (Fig. 1). Lake Macquarie is the largest of at least seven marine-dominated lagoons along the 1900 km NSW coast (HANNAN; WILLIAMS, 1998). The lake is 24 km long with a catchment area of approximately 700 km^2 and a waterway area of 120 km^2. Water temperatures within the lake range from around 13°C during winter to 28°C during summer (EYRE; FERGUSON, 2002). The permanently open entrance channel (Swansea Channel) that connects the lake to the ocean is approximately 5 km long and varies in width from 100 to 400 m and in depth from two to five metres (TRNSKI, 2002). Tidal flushing is estimated to exchange only 1% of the lake's volume during each tidal cycle with lake tidal ranges of less than ten cm (SPENCER, 1959). Circulation within the lake is primarily driven by wind, while average and maximum depths in the lake are 6.7 m and 11 m, respectively (ROY et al., 2001; KING, 1986; SPENCER, 1959). The catchment supports a wide range of land uses from high density urban development, standard residential to agricultural, industrial, mining and conservation areas. The lake is a declared Recreational Fishing Haven with commercial fishing effort prohibited as from 2002, and recreational fishing methods the only extractive fishing methods allowed.

Between the 1st and 9th December 2005, a total of 180 artificial reef units (Mini-Bay Reef Ball® modules) were deployed by barge to create 6 separate artificial reefs along the 5 m depth contour in the southeast portion of the lake (Fig. 1). The site was selected due to its lack of seagrass or natural reef habitat and the presence of coarse sandy sediments; the latter of which provides a stable base for the artificial reef units. Each reef was located approximately 180m from the next with approximately 900 m between AR1 and AR6. Large artificial reefs AR1, AR3 and AR5 were constructed from 50 artificial reef units, while small artificial reefs AR2, AR4 and AR6 were constructed from ten units. The large and small reefs occupied a 'footprint' of approximately 22 m^2 and 4 m^2, respectively, and total reef volume (i.e. all reefs combined) was approximately 36 m^3.

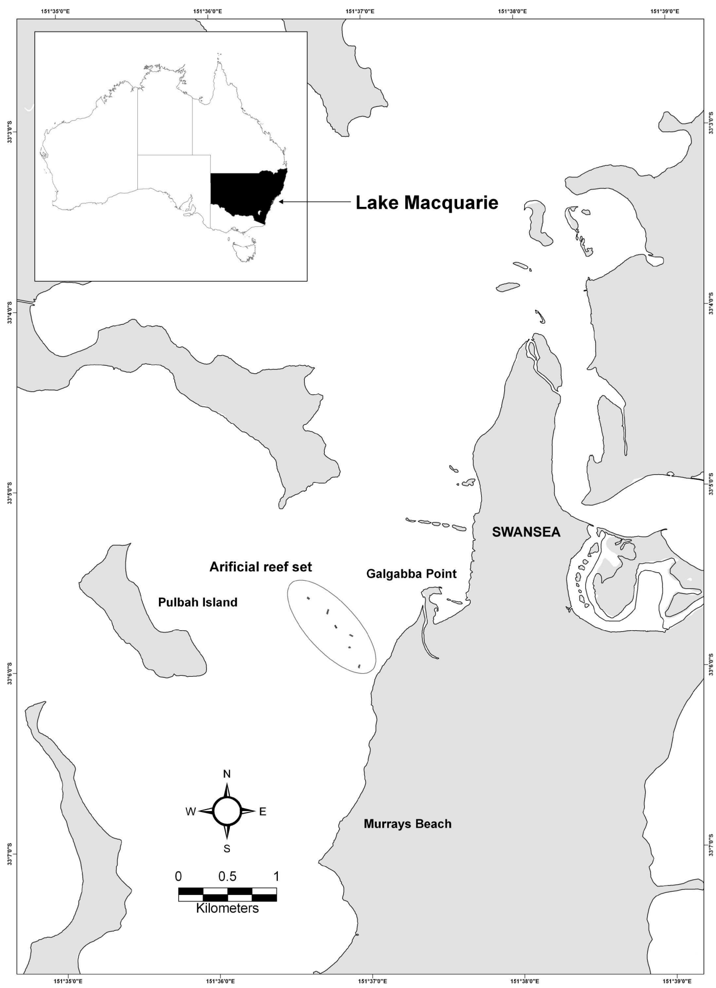

Fig. 1. Map indicating location of study site.

Data Collection - Baited Remote Underwater Video (BRUV)

Data collection using baited remote underwater video (BRUV) was conducted from January 2006 (mid-summer) to November 2007 (early-summer), spanning a total sampling period of almost 2 years (699 days) following reef deployment. The reefs were sampled in random order by single BRUV units, deployed for a 30 minute period between 8am and 4pm two sampling days per month.

Three identical BRUV units were constructed based on the design of Cappo et al. (2004). The BRUV unit consisted of a simple stainless steel frame constructed as a mount for the camera and underwater housing. A bait arm (20 mm plastic conduit) extending 1 m from the face of the camera housing supported a plastic bait container, containing standardized bait (ground chickpea, water and tuna oil), which was replenished prior to every deployment. Initial trials indicated that the standardized mixture provided a constant dispersal over the 30 min deployment period under a variety of conditions. Each camera was set on 'short play' (SP) mode and the focus set to 'manual infinity'.

Analysis of video recordings was carried out using the BRUVS tape reading interface 2.1 (Australian Institute of Marine Science, 2006). For each species seen, the time of first sighting, and the maximum number seen together, known as the 'Max N', and the time of the Max N event was recorded (CAPPO et al., 2004).

Data Analysis

Colonization

Data were stratified by: (1) reef age (0-6, 7-12, 13-18 and 19-24 months), and (2) sighting frequency groupings, where sighting frequency is defined as the proportion (i.e. %) of the total days on which each species was identified. Sighting frequency was then categorized into four reef residency groups: permanent species (>75%), frequent species (74.9-30%), scarce (29.9-10%) and rare species (<9.9%) (TESSIER et al., 2005).

Species richness was calculated as the total number of species observed during each sample day. Cumulative species richness was calculated as the total accumulation of species numbers over the total 699 day sampling period. Diversity was calculated using the Shannon-Wiener index for each sampling day over the sampling period.

Community development

Non-metric multivariate analysis was used to investigate the effect of reef age on fish community assemblages and was done using PRIMER V.6. (CLARKE; WARWICK, 2001). To standardize data among sampling days, the count for each species within a sampling day was transformed to the proportion relative to the total count (all species combined) for that day. A similarity matrix was then constructed using the Bray-Curtis similarity measure and patterns in fish assemblages among reef-age groupings then visually explored using non-parametric multidimensional scaling (nMDS) ordination plots (CLARKE; WARWICK, 2001).

One-way analysis-of-similarities (ANOSIM) was used to test for significant differences in fish assemblages among reef-age groupings, with the main species contributing to similarities among groupings determined using similarity-percentages (SIMPER) analyses. The ratio of the average similarity among reef-age groupings (Sim) and the associated standard deviation (SD) is a measure of how consistent the contribution of a given species is to the characterization of similarities within reef age groupings. Species displaying a high Sim/SD ratio (i.e. > 1) and similarity contribution percentage can be considered good key discriminating species (CLARKE; WARWICK, 1994). Relative abundance indices for each of the key discriminating species were displayed by superimposing bubble plots on the nMDS ordination to indicate the relative contribution of those species to any discernible patterns in the nMDS plot.

One-way ANOVA (Kruskal-Wallis test) was used to test for significant differences among reef-age groupings with respect to species richness, species diversity (Shannon-Wiener index), and the relative abundance estimates for each of the key discriminating species.

RESULT

Colonization

A total of 27 species belonging to 17 families was identified in BRUV samples during the 699 day post-deployment period. *Pelates sexlineatus* (Terapontidae), *Acanthopagrus australis* (Sparidae), *Pagrus auratus* (Sparidae) and *Rhabdosargus sarba* (Sparidae) were found to be either permanent or frequent reef residents, with sighting frequencies of 76.4%, 82.8%, 81.7% and 70.8%, respectively (Table 1). All four species were recorded from the first day of sampling (i.e. reef age of 26 days).

P. sexlineatus was by far the most abundant species identified, making up 64% of total (i.e. all species combined) abundance over the sampling period. In contrast, *A. Australis* made up 11%, *P. auratus* 7% and *R. sarba* 7%. Of the remaining 23 species identified, 5 were found to be scarce, with 18 species only found rarely in the reef group, with sighting frequencies ranging from 0.4% to 25.5% (Table 1). The combined abundance of these 23 species only accounted for 6% of the total abundance recorded in the artificial reef group.

Colonization to the artificial reef group was relatively rapid during the first year of sampling, with cumulative species richness increasing to 13 species within the first three months and 21 species by the end of the first year, which accounted for over 77% of all species indentified during the 699 day sampling period (Fig. 2a). Colonization by new species after the first year remained low and sporadic (Fig. 2b). Among sampling days, species richness ranged between 4 and 15 species, with diversity indices ranging between 0.52 and 1.99.

Community Development

Results from the one-way ANOSIM showed significant differences between artificial reef age groupings (R = 0.206) (Table 2), while results from the SIMPER analysis showed that similarity between reef age groupings was primarily driven by interactions among the permanent and frequent artificial reef residents *P. sexlineatus* and *P. auratus* (Table 3). Both species had a consistently high Sim/SD of >1.0 within each of the four reef age groupings, ranging between 5.3 and 1.17, and 4.64 and 1.22 respectively. The nMDS ordination plot illustrates some separation between reef age groupings (Fig. 3a). The key discriminating species showed differences in their relative abundance among reef-age groupings (Fig. 3b-e). *P. sexlineatus* had higher relative abundances during the first 12 months post reef deployment with larger circles representing higher abundances (Fig 3b), while in contrast, *A. australis* and *R. sarba* has higher abundances overlaying the older reef age groupings (Fig. 3c,e). The results for *P. auratus* were less distinct, with a relatively consistent relative abundance across the reef age groupings (Fig. 3d).

There were no significant differences in species richness among reef age groupings (ANOVA, P>0.05; Fig. 4a). In contrast, there was a significant increase in species diversity with increasing reef age (ANOVA, P>0.05; Fig. 4b). There were significantly fewer *P. sexlineatus* with increasing reef age (ANOVA, P<0.05; Fig. 5a), while significantly more *A. australis* with increasing reef age (ANOVA, P<0.001; Fig. 5b). No significant variation in the mean relative abundance of *P. auratus* and *R. sarba* was detected with reef age (ANOVA, P>0.05; Fig. 5c-d).

Table 1. All species identified by BRUV from January 2006 to November 2008 including family, total counts, mean counts and standard error (SE). Each species is classified according to category of occurrence (permanent, frequent, scarce and rare).

Family	Species	Total Count	Mean (Max N)	Standard Error	Frequency (%)	Residency
Sparidae	*Acanthopagrus australis*	975	4.41176	0.238151	82.772	permanent
Apogonidae	*Apogon fasciatus*	151	5.03333	0.971411	11.236	scarce
Scorpididae	*Atypichthys strigatus*	1	1	0	0.375	rare
Diodontidae	*Dicotylichthys punctulatus*	12	1	0	4.494	rare
Enolopsidae	*Enoplosus armatus*	1	1	1	0.375	rare
Gerreidae	*Gerres subfasciatus*	66	3.47368	0.859649	7.116	rare
Muraenidae	*Gymnothorax prasinus*	1	1	0	0.375	rare
Mugilidae	*Liza argentea*	1	1	0	0.375	rare
Monacanthidae	*Meuschenia freycineti*	30	1.30435	0.116517	8.614	rare
Monacanthidae	*Meuschenia trachylepis*	115	1.69118	0.120828	25.468	scarce
Scorpididae	*Microcanthus strigatus*	71	1.775	0.194434	14.981	scarce
Monacanthidae	*Monacanthus chinensis*	47	1.27027	0.092095	13.858	scarce
Monodactylidae	*Monodactylus argenteus*	4	1	0	1.498	rare
Mugilidae	*Mugil cephalus*	2	1	0	0.749	rare
Ophichthidae	*Ophisurus serpens*	1	1	0	0.375	rare
Sparidae	*Pagrus auratus*	615	2.8211	0.139609	81.648	permanent
Teraponidae	*Pelates sexlineatus*	5659	27.7402	1.652202	76.404	permanent
Platycephalidae	*Platycephalus fuscus*	1	1	0	0.375	rare
Carangidae	*Pseudocaranx dentex*	15	1.66667	0.372678	3.371	rare
Sparidae	*Rhabdosargus sarba*	599	3.16931	0.173742	70.787	frequent
Carangidae	*Seriola dumerili*	15	1.66667	0.235702	3.371	rare
Carangidae	*Seriola lalandi*	8	2	0.707107	1.498	rare
Siganidae	*Siganus fuscescens*	4	4	0	0.375	rare
Sillanginidae	*Sillago ciliata*	7	1.75	0.478714	1.498	rare
Carangidae	*Trachurus novaezelandiae*	398	5.60563	0.691564	26.592	scarce
Belonidae	*Tylosurus gavialoides*	3	1	0	1.124	rare
Mullidae	*Upeneus tragula*	2	1	0	0.749	rare

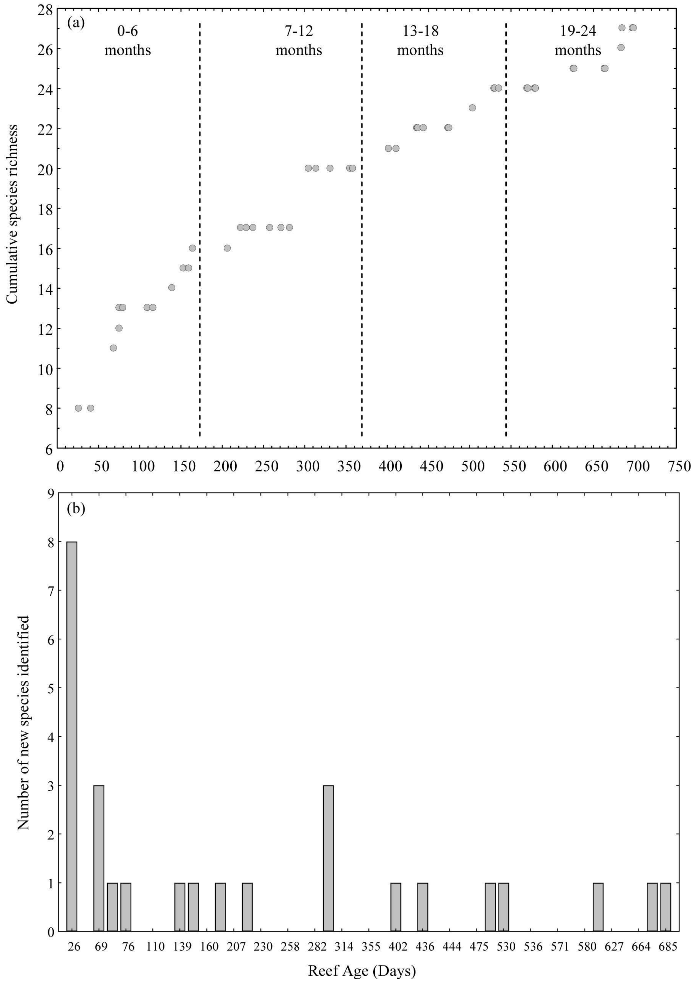

Fig. 2. (a) Species accumulation curve (broken vertical lines denote the reef age groupings); and (b) bar chart depicting new species identification on the artificial reef set by BRUV over the 699 day sample period. Day 0 is 05 December 2005 and day 700 is 10 November 2007.

Table 2. Results of one-way ANOSIM (R values and significance levels) for relative total abundance of species sampled from BRUV on the artificial reef set and Pairwise tests between reef age groupings. * indicates significant result (P=<0.05), ** indicates highly significant result (P=<0.001).

Global R = 0.206 Significance: <0.001*

Pairwise Tests

Reef Age Groupings	R - stat.	Significance
0-6 months, 7-12 months	0.028	n/s
0-6 months, 13-18 months	0.27	P=<0.001*
0-6 months, 19-24 months	0.319	P=<0.05*
7-12 months, 13-18 months	0.117	P=<0.05*
7-12 months, 19-24 months	0.277	P=<0.05*
13-18 months, 19-24 months	0.179	P=<0.05*

Number of permutations: 999

Table 3. Results of SIMPER analysis for relative total abundance of species sampled from BRUV on the artificial reef set. Similarity ratio (Sim/SD) indicate the consistency with which each species contributes towards similarity within reef age groupings, with larger values (>1.0) indicating greater consistency as a discriminating species.

Reef Age Grouping / Species	Av. Rel. Abund.	Sim/SD	Contrib%
0-6 months	Average similarity	80.26	
Pelates sexlineatus	72.05	5.3	81.58
Pagrus auratus	6.17	4.64	5.99
Rhabdosargus sarba	5.78	2.36	4.95
7-12 months	Average similarity	74.88	
Pelates sexlineatus	66.46	3.92	76.12
Pagrus auratus	10.81	2.08	9.48
Acanthopagrus australis	8.99	1.81	7.47
13-18 months	Average similarity	70.58	
Pelates sexlineatus	54.38	1.77	61.81
Acanthopagrus australis	20.21	2.93	19.28
Pagrus auratus	5.48	3.83	6.5
Rhabdosargus sarba	6.85	1.92	5.76
19-24 months	Average similarity	56.44	
Pelates sexlineatus	38.75	1.17	38.36
Acanthopagrus australis	23.65	1.31	26.57
Rhabdosargus sarba	14.88	1.5	17.16
Pagrus auratus	10.39	1.22	11.18

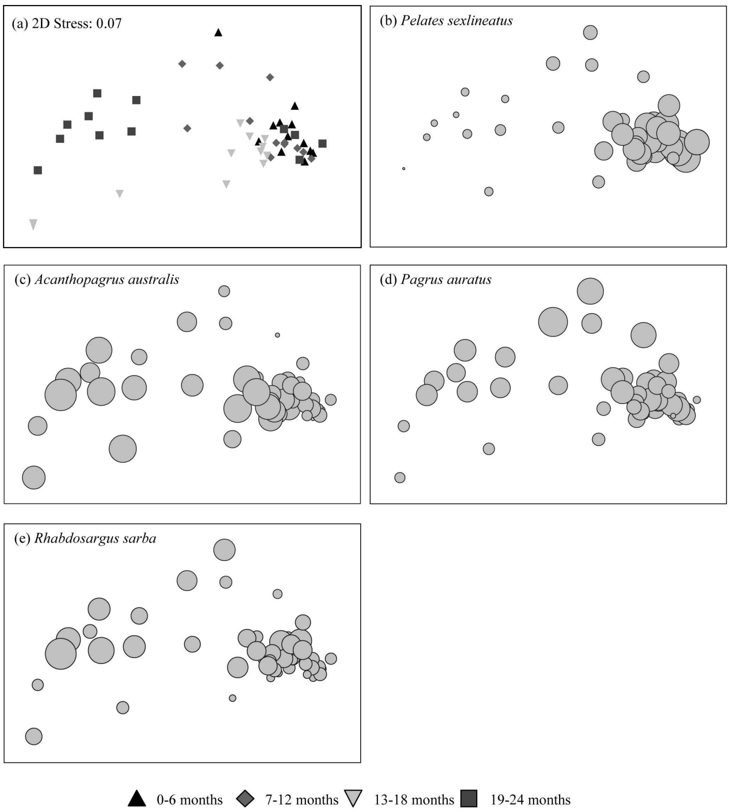

Fig. 3. (a) Non-metric multidimensional scale (nMDS) ordination plot of total relative abundance (standardised) and reef age groupings (month) from the artificial ref set sampled by BRUV for all species. Fig. b, c, d & e are species identified as key colonising species (permanent and frequent and are ranked by % contribution to similarity between reef age groups and represented by superimposed “bubbles”. Bubbles of increasing size represent increasing abundance.

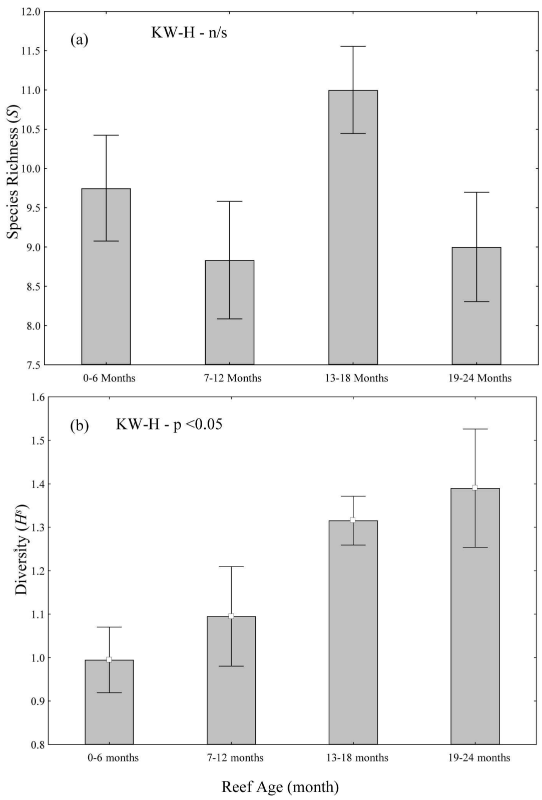

Fig. 4. Mean (a) species richness (*S*) and (b) diversity (H^s) by reef age (month) (+/-SEM) for relative abundance estimates recorded by BRUV. Results of Kruskal-Wallis tests (at P = 0.05) for significant interactions of reef age (month) are written above graphs (n/s = no significance).

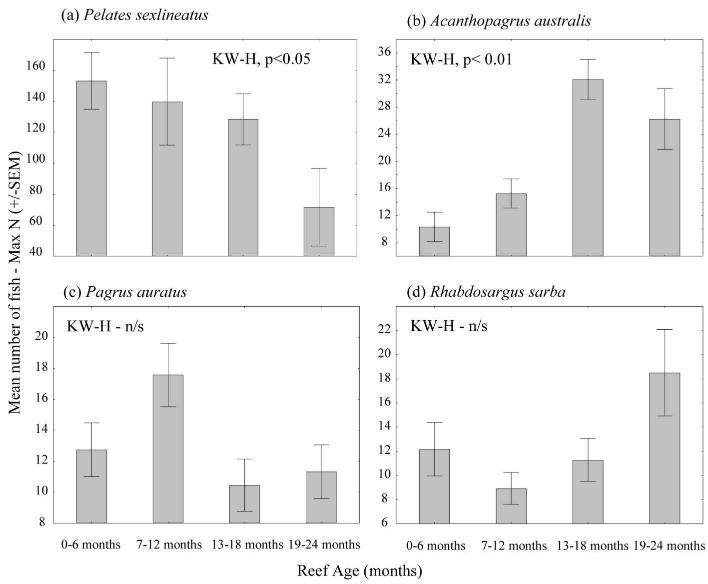

Fig. 5. Mean number of fish (Max N) by reef age (month) for key colonising species. Results of Kruskal-Wallis tests (at P = 0.05) for significant interactions reef age groupings (month) are shown above graphs (n/s = no significance).

DISCUSSION

The recruitment of a diverse range of estuarine species (27 species in total) involving an initial rapid increase recruitment during the first year post-deployment followed by a relatively moderate increase during the following year is a pattern consistent with other similar, non-estuarine studies (CUMMINGS, 1994; GOLANI; DIAMANT, 1999, MANDERSON; ABLE, 2003, BOHNSACK; TALBOT, 1980; MATTHEWS, 1985; MARKEVICH, 2005; WALSH, 1985; HAUGHTON; AIKEN, 1989). For example, two separate artificial reefs, one located in northern tropical Australian waters (Great Barrier Reef) and the other in Florida, showed that colonization reached saturation within 1-2 months of deployment, with a total of 88 species identified over 32 months in Australia, and 89 species identified over 39 months off Florida (BOHNSACK; TALBOT 1980). A similar study of near-shore boulder artificial reefs (CUMMINGS, 1994) recorded the first colonizers within hours of its construction, with 31 species recorded within the first two months post reef-deployment. In the Red Sea, 94 species colonized a newly deployed artificial reef over a 728 day sampling period, with maximum species richness reached within the first seven months of the reef's construction (GOLANI; DIAMANT 1999). What is consistent among all these studies regardless of the location, size, material or time of deployment, is that colonization of the artificial reef was initially rapid, followed by a leveling off, and in some cases a reduction, in the number of species being recorded.

The colonization and community development associated with artificial reefs has been defined by several theories, the most widely accepted being the concept of island biogeography (MACARTHUR; WILSON, 1967). Classic island biogeographical principles predict that colonization will be a result of the rate of movement of the colonizing species, the distance from the source of new recruits and the size of the area being colonized (WALSH, 1985). Previous studies have shown that on patch reef habitats, fish richness declined with increasing distance from larger 'source' reefs and that species richness increased with increasing patch reef area (MOLLES JR, 1978). At a much larger spatial scale, reef species richness has been shown to decrease with increasing distance from sources of historic biodiversity (STEHLI; WELLS, 1971; MORA et al., 2003). Studies designed to test the principles of island biogeography at more local scales determined that there was no consistent pattern in the response of the community to disturbance and concluded that the species composition of such a small assemblage was almost entirely a matter of chance (SALE; DYBDAHL, 1975; SALE; DYBDAHL, 1978). While models based on island biogeography may provide some of the uniform guiding concepts associated with the colonization and community development, understanding patterns of recruitment and succession of fish communities associated with an estuarine artificial system requires assessment of physical and biological aspects of the environment and the associated fish assemblage.

The rapid colonization of artificial reefs is often hypothesized to be due to a 'draw-down' effect of post-settlement individuals being attracted to the newly constructed reef structure from nearby natural habitats (BOHNSACK; SUTHERLAND, 1985; MATTHEWS, 1985; ALEVIZON; GORHAM, 1989, HUECKEL et al., 1989; GOLANI; DIAMANT, 1999). The number and type of species attracted to the newly deployed structure will be mediated by factors such as the time of deployment (seasonality) and the proximity of the structure in relation to other sources of recruitment (SALE, 1980). Variation in the ontogenetic, physical and behavioral aspects of species (PIZZOLON et al., 2008), the proximity and degree of connectivity among suitable habitat patches in relation to the artificial reef, as well as the resilience of the developing community to respond to post settlement processes such as predation (FERNANDEZ et al., 2007; WALSH, 1985; HERRERA et al., 2002b) have also been demonstrated to influence the composition of fish communities associated with artificial structures.

The rate of colonization and community development of the Lake Macquarie artificial reefs is directly related to the position of the artificial reef in relation to existing habitats. The four key colonizing species identified here - P. sexlineatus, and the sparids A. australis, P. auratus and R. sarba - remained frequently or permanently associated with the reef structure and were probably all recruited from areas of adjacent natural habitat, which they are known to inhabit as post-settlement juveniles, sub-adults or adults (MISKIEWICZ, 1987; HANNAN; WILLIAMS, 1998). This rapid colonization of these species to the artificial reefs is thought to be a result of the ability of individuals (or schools) to move relatively large distances over sand habitats that may present a barrier for other less mobile reef associated species. Some species are capable of moving over bare sand for feeding (AMBROSE; ANDERSON, 1990), while others are reluctant to cross it (CHAPMAN; KRAMER, 2000; COLL et al., 1998; FERNANDEZ et al., 2007) with extensive sand patches perceived as barriers of variable permeability in relation to the size and vagility of each species (COLL et al., 1998; BELL; WESTOBY, 1986; STAMPS et al., 1987). Medium-sized mobile fish are least influenced by reef isolation or low habitat connectivity (AULT;

JOHNSON, 1998; MCCLANAHAN; MANGI, 2000; FERNANDEZ et al., 2007). Sparids have been found to be key colonizing species on isolated patch artificial reefs due to their ability to cross relatively large expanses of sand where little protection from predation is found (FERNANDEZ et al., 2007).

While the potential of species to colonize the reef structure may be mediated by behavior and habitat connectivity, the development of the reef fish assemblage over time is influenced by a suite of 'post-settlement' processes. Predation has been identified as one of the most significant processes in structuring natural reef communities with inverse relationships identified between local abundances of prey species and resident piscivores in natural rocky and coral reef areas (SHULMAN, 1985; SHULMAN et al., 1983; HIXON, 1991; HIXON; BEETS, 1993, OVERHOLTZER-MCLEOD, 2006; JOHNSON, 2006). The effect of predation in structuring artificial reef communities remains poorly understood and more recent studies indicate significantly higher visitation rates and a greater diversity of larger predators have been identified as factors responsible for higher rates of mortality of prey species on artificial reefs than comparable natural reefs (OVERHOLTZER;KAREN, 2004) and that artificial reefs may contribute to increases in the natural mortality of juvenile species by facilitating predator prey interactions (LEITÃO et al., 2008).

The observation that predation is strongly influenced by prey abundance has been identified by a variety of studies. Predatory fish species are also more likely to respond to larger aggregations of prey species (CONNELL, 2002; STEWART; JONES, 2001), which may result in an increase in the proportional mortality of aggregated prey, since the predators may feed at a greater rate (CONNELL, 2002; CONNELL, 2000). Numbers of *P. sexlineatus,* a highly abundant schooling species were found to decrease significantly with increasing reef age. *P. sexlineatus* were initially identified around the artificial reef in large numbers, with mean relative abundance estimates of greater than 150 individuals which decreased to less than 70 individuals over the sampling period. Scars and injuries (e.g. bites and partial removal of fins) were regularly observed on individuals of this species, during analysis of BRUV data. Two important transient predators - S*eriola dumerili* and *Seriola lalandi* were not observed on the naturally occurring reefs within the lake (unpublished data), but regularly identified by diver observations and on BRUVS tapes in the vicinity of the artificial reef complex. These species (particularly *S. dumerili*) are more commonly associated with inshore and coastal shelf waters and are not usually encountered in the upper reaches of coastal estuaries (KAILOLA et al., 1993). Specific predator species have also been observed to be chiefly responsible for controlling the structure of artificial reef fish assemblages (HERRERA et al., 2002a). Given the lack of resident piscivores, it is likely that these transient predators could have a significant role in the reduction in the relative abundance of *P. sexlineatus*, as they are known to be opportunistic feeders consuming a range of small fish, crustaceans and squid (KAILOLA et al., 1993).

The reduction in relative abundance of *P. sexlineatus* after the first year of sampling may have potentially presented an opportunity for new species to recruit to the reef or for existing reef species such as *A. australis* to recruit in significantly higher numbers. However, when interpreting these results, we were careful to note that although a significant decrease in numbers of *P. sexlineatus* was detected, any corresponding increase in other species (e.g. *A. australis*) may potentially be an artefact of inherent biases associated with the use of BRUV systems. The density saturation effect of species, with a low time to first feeding and high relative abundance (Max N) value, and the feeding behavior of some species being simply outcompeted to the bait by other more abundant or aggressive species, has been found to be a possible cause of bias for BRUV systems resulting in conservative relative abundance estimates of more mobile species such as *A. australis* (LOWRY, in press).

The number of species identified on a developing estuarine artificial reef is expected to increase with reef age (BORTONE et al., 1994). This increase reflects the different colonization patterns of different fish species (PIZZOLON et al., 2008). Some studies have found that an equilibrium in species diversity can be reached within months of an artificial reef's deployment (CUMMINGS, 1994), while other studies have found that species diversity continually increases during the first two years post reef-deployment (PIZZOLON et al., 2008; HAUGHTON; AIKEN, 1989) - a pattern similar to that found here. The mean number of species observed during this study increased rapidly during the first six months post reef deployment but slowed and almost leveled off as the reef aged, with no significant increase in species richness found. This suggests that although the development of the reef was not complete, the colonization process was potentially reaching a point of climax where no new species from surrounding natural habitats were likely to colonize the reef in large numbers. Species diversity significantly increased with reef age and no clear equilibrium in species diversity was found during this study. Consistent with other studies (GOLANI; DIAMANT, 1999), it is thought that this finding may be the result of a decrease in one or more highly abundant key colonizing species.

ACKNOWLEDGEMENTS

We would like to thank the NSW Recreational Fishing Trust for funding the construction of the artificial reef and also funding this research project. Special thanks to the many field assistants including A. Brohier, N. Hannerford and B. van der Walt for their valuable time and assistance. Thanks to W. Macbeth and N. Otway for their review of this manuscript. And a special mention of Lake Macquarie City Council as the consenting authority for approving the deployment of artificial reefs in Lake Macquarie.

REFERENCES

ALEVIZON, W. S.; GORHAM, J. C. Effects of artificial reef deployment on nearby resident fishes. **Bull. mar. Sci.,** v., 44**,** p. 646-661, 1989.

AMBROSE, R. F.; ANDERSON, T. W. Influence of an artificial reef on the surrounding infaunal community. **Mar. Biol.**, v. 107**,** p. 41-52, 1990.

AULT, T. R.; JOHNSON, C. R. SPATIALLY AND TEMPORALLY PREDICTABLE FISH COMMUNITIES ON CORAL REEFS. **Ecol. Monogr.**, v. 68**,** p. 25-50, 1998.

BELL, J. D.; WESTOBY, M. Variation in seagrass height and density over a wide spatial scale: effects on common fish and decapods. **J. expl mar. Biol. Ecol.** v**.** 104**,** p. 275-295, 1986.

BOHNSACK, J. A. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? **Bull. mar. Sci.**,v. 44**,** p. 631-645, 1989.

BOHNSACK, J. A.; SUTHERLAND, D. L. Artificial reef research: a review with recommendations for future priorities. **Bull. mar. Sci.**,v. 37**,** p. 11-39, 1985.

BOHNSACK, J. A.; TALBOT, F. H. Species-packing by reef fishes on Australian and Caribbean reefs: an experimental approach. **Bull. mar. Sci.**, v. 30**,** p. 710-723, 1980.

BORNTRAGER, J. F.; FARRELL, T. M. The effect of artificial reef size on species richness and diversity in a Florida estuary. **Fla. Scient.**, v. 55**,** p. 229-235, 1992.

BORTONE, S. A. Resolving the attraction-production dilemma in artificial reef research: some yeas and nays. **Fisheries**, v. 23**,** p. 6-10, 1998.

BORTONE, S. A.; MARTIN, T.; BUNDRICK, C. M. Factors affecting fish assemblage development on a modular artificial reef in a northern Gulf of Mexico estuary. **Bull. mar.Sci.,** v. 55, n. 2, p. 319-332, 1994.

BRANDEN, K. L.; POLLARD, D. A.; REIMERS, H. A. A review of recent artificial reef developments in Australia. **Bull. mar. Sci.**, v. 55**,** p. 982-994, 1994.

BURCHMORE, J. J.; POLLARD, D. A.; BELL, J. D.; MIDDLETON, M. J.; PEASE, B. C.; MATTHEWS, J. An ecological comparison of artificial and natural rocky reef fish communities in Botany Bay, New South Wales, Australia. **Bull. mar. Sci.**, v. 37**,** p. 70-85, 1985.

BURTON, W. H.; FARRAR, J. S.; STEIMLE, F.; CONLIN, B. Short communication Assessment of out-of-kind mitigation success of an artificial reef deployed in Delaware Bay, USA. **ICES J. mar. Sci.** ,v. 59**,** p.S106-S110, 2002.

CAPPO, M.; SPEARE, P.; DEATH, G. Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. **J. expl mar. Biol. Ecol.**, v. 302**,** p. 123-152, 2004.

CHAPMAN, M. G.; BULLERI, F. Intertidal seawalls-new features of landscape in intertidal environments. **Lands. urban Plan.**, v. 62**,** p. 159-172, 2003.

CHAPMAN, M. G.; CLYNICK, B. G. Experiments testing the use of waste material in estuaries as habitat for subtidal organisms. **J. expl. mar. Biol. Ecol.**, v. 338**,** p. 164-178, 2006.

CHAPMAN, M. R.; KRAMER, D. L. Movements of fishes within and among fringing coral reefs in Barbados. **Environ. Biol. Fishes**, v. 57**,** p. 11-24, 2000.

CLARKE, K. R.; WARWICK, R. M. **Changes in marine communities:** An approach to statistical analysis and interpretation. Marine Laboratory, Plymouth, 1994.

CLARKE, K. R.; WARWICK, R. M. **Changes in marine communities:** an approach to statistical analysis and interpretation. Marine Laboratory, Plymouth, 2001.

CLYNICK, B. G.; CHAPMAN, M. G.; UNDERWOOD, A. J. Fish assemblages associated with urban structures and natural reefs in Sydney, Australia. **Austral Ecol.**, v. 33**,** p. 140-150, 2008.

COLL, J.; MORANTA, J.; RENONES, O.; GARCIA-RUBIES, A.; MORENO, I. Influence of substrate and deployment time on fish assemblages on an artificial reef at Formentera Island (Balearic Islands, western Mediterranean). **Hydrobiologia**, v. 385**,** p. 139-152, 1998.

CONNELL, S. D. Is there safety-in-numbers for prey? **Oikos,** p. 527-532, 2000.

CONNELL, S. D. Effects of a predator and prey on a foraging reef fish: implications for understanding density-dependent growth. **J. Fish Biol.**, v. 60**,** p. 1551-1561, 2002.

COUTIN, P. **Artificial reefs: applications in Victoria from a literature review**. Melbourne: Marine and Freshwater Resources Institute, 2001.

CUMMINGS, S. L. Colonization of a nearshore artificial reef at Boca Raton (Palm Beach County), Florida. **Bull. mar. Sci.**., v. 55**,** p. 1193 - 1215, 1994.

DEMARTINI, E. E.; ROBERTS, D. A.; ANDERSON, T. W. Contrasting patterns of fish density and abundance at an artificial rock reef and a cobble-bottom kelp forest. **Bull. mar. Sci.**, v. 44**,** p. 881-892, 1989.

EYRE, B.; FERGUSON, A. **Lake Macquarie Water Quality Review.** N.S.W**.,** Australia: Centre for Coastal Biogeochemistry, Southern Cross University, 2002. (Unpublished report).

FERNANDEZ, T. V.; DANNA, G.; BADALAMENTI, F.; PÉREZ-RUZAFA, A. Habitat connectivity as a factor affecting fish assemblages in temperate reefs. **Aquat. Biol.**, v. 1**,** p. 239-248, 2007.

FOSTER, K. L.; STEIMLE, F. W.; MUIR, W. C.; KROPP, R. K.; CONLIN, B. E. Mitigation Potential of Habitat Replacement: Concrete Artificial Reef in Delaware Bay Preliminary Results. **Bull. mar. Sci.,** v. 55, n. 2**,** p. 783-795, 1994.

GOLANI, D.; DIAMANT, A. Fish colonization of an artificial reef in the Gulf of Elat, northern Red Sea. **Environ. Biol. Fishes**, v. 54**,** p. 275-282, 1999.

GORHAM, J. C.; ALEVIZON, W. S. Habitat complexity and the abundance of juvenile fishes residing on small scale artificial reefs. **Bull. mar. Sci.**,v. 44**,** p. 662-665, 1989.

GROSSMAN, G. D.; JONES, G. P.; SEAMAN, W. J. Do artificial Rrefs increase regional fish production? A review of existing data. **Fisheries**, v. 22**,** p.17-23, 1997.

HANNAN, J. C.; WILLIAMS, R. J. Recruitment of juvenile marine fishes to seagrass habitat in a temperature Australian estuary. **Estuaries Coasts**, v. 21**,** 29-51, 1998.

HAUGHTON, M. O.; AIKEN, K. A. Biological notes on artificial reefs in Jamaican waters. **Bull. mar. Sci.,** v. 44**,** p. 1033-1037, 1989.

HERRERA, R.; ESPINO, F.; GARRIDO, M.; HAROUN, R. J. Observations on fish colonization and predation on two artificial reefs in the Canary Islands. **ICES J. mar. Sci.**, v. 59**,** p. S69, 2002a.

HERRERA, R.; ESPINO, F.; GARRIDO, M.; HAROUN, R. J. Short communication Observations on fish colonization and predation on two artificial reefs in the Canary Islands. **ICES J. mar. Sci.**, v. 59**,** p.S69-S73, 2002b.

HIXON, M. A. Predation as a process structuring coral reef fish communities. 1991.

HIXON, M. A.; BEETS, J. P. Predation, prey refuges, and the structure of coral-reef fish assemblages. **Ecol.Monogr., v.** 77, p. c101, 1993.

HUECKEL, G. J.; BUCKLEY, R. M.; BENSON, B. L. Mitigating rocky habitat loss using artificial reefs. **Bull. mar. Sci.**, v. 44**, p.** 913-922, 1989.

JOHNSON, D. W. Predation, habitat complexity, and variation in density-dependent mortality of temperate reef fishes. **Ecology**, v. 87**,** 1179-1188, 2006.

KAILOLA, P. J.; WILLIAMS, M. J.; STEWART, P. C.; REICHELT, R. E.; MCNEE, A.; GRIEVE, C. **Australian fisheries resources**. Brisbane, AU: Bureau of Resource Sciences and the Fisheries Research and Development Corporation,1993.

KERR, S. **Artificial reefs in Australia**. Their construction, location and function. Canberra, AU: Bureau of Rural Resources, 1992.

KIM, C. G. Artificial reefs in Korea. **Fisheries**, 26**,** 15-18, 2001.

KING, R. J. **Aquatic angiosperms in coastal saline lagoons of New South Wales: 1. The vegetation of Lake Macquarie**. 1986.

LAUFLE, J. C.; PAULEY, G. B. Fish colonization and materials comparisons on a Puget Sound artificial reef. **Bull. mar. Sci.**, v. 37**,** p. 227-243, 1985.

LEITÃO, F.; SANTOS, M.; ERZINI, K.; MONTEIRO, C. The effect of predation on artificial reef juvenile demersal fish species. **Mar. Biol.**, v. 153**,** p. 1233-1244, 2008.

LINDBERG, W. J. Can science resolve the attraction-production issue? **Fisheries**, v. 22**,** 10-13, 1997.

LINDBERG, W. J.; FRAZER, T. K.; PORTIER, K. M.; VOSE, F.; LOFTIN, J.; MURIE, D. J.; MASON, D. M.; NAGY, B.; HART, M. K. Density-dependent habitat selection and performance by a large mobile reef fish. **Ecol. Applicat.** , v. 16**,** p. 731-746, 2006.

LOWRY, M. F., H. GREGSON, M. MCKENZIE, R. A **Comparison of methods for estimating fish assemblages associated with estuarine artificial reefs**. (In press).

MACARTHUR, R. H.; WILSON, E. O. Island biogeography. The theory of island biogeography. Princeton, N.J.: Princeton University Press, 1967. 203 p. .

MANDERSON, J.; ABLE, K. W. A characterization of juvenile fish assemblages around man-made structures in the New York-New Jersey harbor estuary, USA. **Bull. mar. Sci.**, v. 72**,** p. 877-889, 2003.

MARKEVICH, A. I. Dynamics of Fish Colonization of an Experimental Artificial Reef in Peter the Great Bay, Sea of Japan. **Russian J. mar. Biol.,** v. 31**,** p. 221-224, 2005.

MARTIN, T. R.; BORTONE, S. A. Development of an epifaunal assemblage on an estuarine artificial reef. **Gulf Mexico Sci.**, v. 15**,** p. 55-70, 1997.

MATTHEWS, K. R. Species similarity and movement of fishes on natural and artificial reefs in Monterey Bay, California. **Bull. mar. Sci.**, v. 37, p. 252-270, 1985.

MCCLANAHAN, T. R.; MANGI, S. Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. **Ecol. Applicat.**, v. 10**,** p. 1792-1805, 2000.

MILLER, M. W. Using ecological processes to advance artificial reef goals. **ICES J. mar. Sci.**, v. 59**,** p. S27-31-S27-31, 2002.

MISKIEWICZ, A. G. **Taxonomy and ecology of fish larvae in Lake Macquarie and New South Wales coastal waters.** Sydney: University of New South Wales, 1987.

MOLLES JR, M. C. Fish species diversity on model and natural reef patches: experimental insular biogeography. **Ecol. Monogr.**, v. 48**,** p. 289-305, 1978.

MORA, C.; CHITTARO, P. M.; SALE, P. F.; KRITZER, J. P.; LUDSIN, S. A. Patterns and processes in reef fish diversity. **Nature**, v. 421**,** p. 933-936, 2003.

OSENBERG, C. W.; ST. MARY, C. M.; WILSON, J. A.; LINDBERG, W. J. A quantitative framework to evaluate the attraction-production controversy. **ICES J. mar. Science**, v. 59**,** p. S214, 2002.

OVERHOLTZER-MCLEOD, K. L. Consequences of patch reef spacing for density-dependent mortality of coral-reef fishes. **Ecology**, v. 87**,** p. 1017-1026, 2006.

OVERHOLTZER, M.; KAREN, L. **Variance in reef spatial structure masks density dependence in coral-reef fish populations on natural versus artificial reefs**, Oldendorf: Inter-Research, 2004.

PERKOL-FINKEL, S.; BENAYAHU, Y. Differential recruitment of benthic communities on neighboring artificial and natural reefs. **J. expl mar. Biol. Ecol.**,v. 340**,** p. 25-39, 2007.

PERKOL-FINKEL, S.; SHASHAR, N.; BENAYAHU, Y. Can artificial reefs mimic natural reef communities? The roles of structural features and age. **Mar. environ. Res.**, v. 61**,** p. 121-135, 2006.

PICKERING, H.; WHITMARSH, D. Artificial reefs and fisheries exploitation: a review of the "attraction versus production" debate, the influence of design and its significance for policy. **Fish. Res.**, v. 31**,** p. 39-59, 1997.

PIZZOLON, M.; CENCI, E.; MAZZOLDI, C. The onset of fish colonization in a coastal defence structure (Chioggia, Northern Adriatic Sea). **Estuar. Coast. Shelf Sci.,** v. 78**,** p. 166-178, 2008.

POLLARD, D. A. Artificial habitats for fisheries enhancement in the Australian region. **Mar. Fish. Rev.**, v. 5**,** p. 11-26, 1989.

POLLARD, D. A.; MATTHEWS, J. Experience in the construction and siting of artificial reefs and fish aggregation devices in Australian waters, with notes on and a bibliography of Australian studies. **Bull. mar. Sci.,**, v. 37**,** p. 299-304, 1985.

ROY, P. S.; WILLIAMS, R. J.; JONES, A. R.; YASSINI, I.; GIBBS, P. J.; COATES, B.; WEST, R. J.; SCANES, P. R.; HUDSON, J. P.; NICHOL, S. Structure and function of south-east Australian estuaries. **Estuar. coast. Shelf Sci. v.**, 53**,** 351-384, 2001.

SALE, P. F. Assemblages of fish on patch reefs-predictable or unpredictable? **Environ. Biol. Fishes**, v. 5**,** p. 243-249, 1980.

SALE, P. F.; DYBDAHL, R. Determinants of community structure for coral reef fishes in an experimental habitat. **Ecology**, v. 56**,** p. 1343-1355, 1975.

SALE, P. F.; DYBDAHL, R. Determinants of community structure for coral reef fishes in isolated coral heads at lagoonal and reef slope sites. **Oecologia**, v. 34**,** 57-74, 1978.

SHULMAN, M. J. Recruitment of coral reef fishes: effects of distribution of predators and shelter. **Ecology**, v. 66**,** p. 1056-1066, 1985.

SHULMAN, M. J.; OGDEN, J. C.; EBERSOLE, J. P.; MCFARLAND, W. N.; MILLER, S. L.; WOLF, N. G. Priority effects in the recruitment of juvenile coral reef fishes. **Ecology**, v. 64**, p.** 1508-1513, 1983.

SPENCER, R. S. Some aspects of the ecology of Lake Macquarie, NSW, with regard to an alleged depletion of fish. II. Hydrology. **Aust. J. mar. Freshwat. Res.**, v. 10**,** p. 279-296, 1959.

STAMPS, J. A.; BUECHNER, M.; KRISHNAN, V. V. The effects of edge permeability and habitat geometry on emigration from patches of habitat. **Am. Naturalist**, v. 129**,** p. 533-552, 1987.

STEHLI, F. G.; WELLS, J. W. Diversity and age patterns in hermatypic corals. **Syst. Biol.**,v. 20**,** p. 115, 1971.

STEWART, B. D.; JONES, G. P. Associations between the abundance of piscivorous fishes and their prey on coral reefs: implications for prey-fish mortality. **Mar. Biol.**, v. 138**,** p. 383-397, 2001.

SUTTON, S. G.; BUSHNELL, S. L. Socio-economic aspects of artificial reefs: Considerations for the Great Barrier Reef Marine Park. **Ocean coast. Mgmt**, v. 50**,** p. 829-846, 2007.

TESSIER, E.; CHABANET, P.; POTHIN, K.; SORIA, M.; LASSERRE, G. Visual censuses of tropical fish aggregations on artificial reefs: slate versus video recording techniques. **J. Expl mar. Biol. Ecol.**,v. 315**,** p. 17-30, 2005.

TRNSKI, T. Behaviour of settlement-stage larvae of fishes with an estuarine juvenile phase: in situ observations in a warm-temperate estuary. **Mar. Ecol. Progr. Ser.,** v. 242**,** p. 205-214, 2002.

WALSH, W. J. Reef fish community dynamics on small artificial reefs: the influence of isolation, habitat structure, and biogeography. **Bull. mar. Sci.,** v. 36**,** p. 357-376, 1985.

(Manuscript received 14 May 2010; revised 17 March 2011; accepted 11 April 2011)