Assessment of artificial reefs in Lake Macquarie NSW

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NON-TECHNICAL SUMMARY

In 2004, I&I NSW (formerly NSW DPI) were successful in gaining funding from the Recreational Fishing Trust for deployment and monitoring of artificial reefs in a number of NSW estuaries. Initial constraint mapping identified potential areas that would be suitable for the deployment of a series of artificial reefs. Following an extensive period of consultation with relevant local government and community groups Lake Macquarie was chosen as the site for the construction of a number of small multi-component artificial reefs.

A combination of Baited Remote Underwater Video (BRUV) and dive surveys was used to monitor the recruitment and succession of the fish and epibenthic community associated with the Lake Macquarie artificial reef complex. Complementary surveys of naturally occurring reef systems were used to compare changes in the structure of fish assemblages between the artificial structures and natural reefs in the immediate area. A photographic survey documented the recruitment and succession of the algal and invertebrate community associated with the artificial reefs. Independent angler surveys were also carried out to provide a comparison of catch and effort between artificial and naturally occurring reefs.

The results of this study have provided a clearer understanding of the type of fish and epibenthic assemblages that result from the deployment of an estuarine artificial reef. It is the first study to demonstrate that structures specifically designed as artificial reefs can be effective at extending the habitats of a variety of fish species in temperate estuarine systems in southeast Australia. We found that the Lake Macquarie artificial reefs were rapidly colonised by a diverse fish community with fifty-one species observed at artificial reef sites over the two year study period with the majority of the species observed within the first year post-deployment. Resident or 'permanent' species, identified on over 75% of observations on artificial reefs include popular recreational species such as yellowfin bream, snapper and tarwhine. Recruitment of epibenthic species was also relatively rapid, however, unlike the fish community was characterised by low diversity with only three species groups (filamentous turfing algae, polychaetes and echinoderms) recorded.

Comparisons between artificial and naturally occurring reefs indicated differences in both species diversity and relative abundance. Artificial reefs recorded a higher number of species (28) compared to natural reefs (21). Of the nineteen species observed on both artificial and natural reefs fourteen recorded greater mean relative abundance on artificial reefs. Species contributing most to the differences between groups were striped trumpeter, yellowfin bream and snapper. Striped trumpeter and snapper were, on average, three times more abundant on artificial reefs than on natural reefs. Yellowfin bream however, were approximately twice as abundant on natural reefs. Other species making significant contributions were tarwhine and yellowtail scad, both with higher average abundances on the artificial reefs.

An independent angling survey demonstrated the utility of artificial reefs as a possible means of enhancing recreational fisheries. Tarwhine, snapper and sand whiting were the species landed in the greatest numbers at both artificial and natural reefs accounting for between 70 – 75% of the total catch at each location. Total mean catch rates of 5.5 fish per hour on the artificial reefs compared with 3.6 – 4.3 fish per hour at the natural reef sites. Analysis of fish length and fish weight data indicated a variable response between locations and species. Artificial reefs did however record the highest mean weight and length for tarwhine and sand whiting.

Although similarities between artificial and naturally occurring reefs were evident, some differences in relative abundances and species diversity of the fish communities was observed. It is unclear whether these differences were a factor of the structural differences of the reef themselves or a result of species succession as the artificial reefs evolve to stable 'climax community'.

A review of similar studies indicates that fish and epibenthic communities may continue to change over an extended period (5 – 10 years) and recommend long term monitoring to obtain a comprehensive understanding of the dynamics of fish and epibenthic communities associated with artificial structures.

1.   INTRODUCTION

1.1.   Development of artificial reef technology

Sutton and Bushnell (2007) define artificial reefs as "any material purposely placed in the marine environment to influence physical, biological, or socio-economic processes related to living marine organisms". The European Artificial Reef Research Network (EARRN) however, defines artificial reefs as "any deliberately placed structure on the seabed that mimics some characteristics of natural reefs," (Baine 2001, Boaventura *et al.* 2006). These definitions can be extended to include materials that are not intentionally placed on the sea floor yet perform similar ecological roles as deliberately placed materials (Chapman & Clynick 2006).

Artificial structures have been deployed worldwide for many purposes including coastal protection, trawling deterrents, habitat loss mitigation, fishery resource enhancement and recreational diving opportunities (White *et al.* 1990, Svane & Petersen 2001). Artificial reefs constructed to provide additional habitat as part of a fisheries enhancement initiative are designed to mimic the structural habitat requirements for specific species and encourage colonisation of primary producers that eventually support new reef communities in otherwise barren areas (White *et al.* 1990, Fowler *et al.* 1999, Lukens *et al.* 2004, Bruno *et al.* 2005, Boaventura *et al.* 2006).

Figure 1.   'Materials of opportunity' such as tyres have historically been the material of choice in artificial reef construction.

Traditionally, artificial reefs were constructed from a wide variety of materials including car tyres (Figure 1), scuttled ships, obsolete oil rig platforms, car bodies, pulverised ash blocks, fibreglass, bamboo and general waste material (D'Itri 1986, Reggio 1987, White *et al.* 1990, Pickering 1996, Svane & Petersen 2001, Walker *et al.* 2002, Jan *et al.* 2003, Brickhill *et al.* 2005, Chapman &

Clynick 2006, Einbinder *et al.* 2006, Krohling *et al.* 2006). These 'materials of opportunity' were often cheap, readily available and considered a novel method of utilising waste materials for environmental benefit (Sherman & Gilliam 1999, Chapman & Clynick 2006). The dubious environmental quality of early artificial reef construction materials, a lack of pre-established objectives and ineffective monitoring frameworks resulted in a series of artificial reef deployments with no clear purpose and undefined outcomes. As a result, many of the earlier artificial reef deployments did not met expectations due to poor design and lack of understanding of the factors responsible for the development of successful artificial habitats (Seaman 2000, Svane & Petersen 2001).

In the last 50 years however, concerted scientific efforts have been made to better understand how the construction and design of artificial reefs influences recruitment and succession of the fish and epibiotic communities associated with them and the impact that these structures have on the ecological processes of their surrounding environments (Butler & Connolly 1996, Carr & Hixon 1997, Pitcher & Seaman 2000). The evidence based approach to the construction and design of artificial reef materials coincided with more stringent legislative requirements resulting in a transition away from materials of opportunity to materials designed and fabricated specifically for use in the construction of artificial reefs (Sherman & Gilliam 1999, Sutton & Bushnell 2007, Su *et al.* 2008). These 'design specific' structures have been demonstrated to provide benefits in terms of biological effectiveness, long-term cost effectiveness, and general performance of the reef system (Sherman *et al.* 2002). Concrete Reef Ball® modules (Figure 2) are an example of design specific materials and have become a popular artificial reef construction method in recent years, deployed in approximately 3,500 projects in over 70 countries worldwide (Sayer *et al.* 2005).
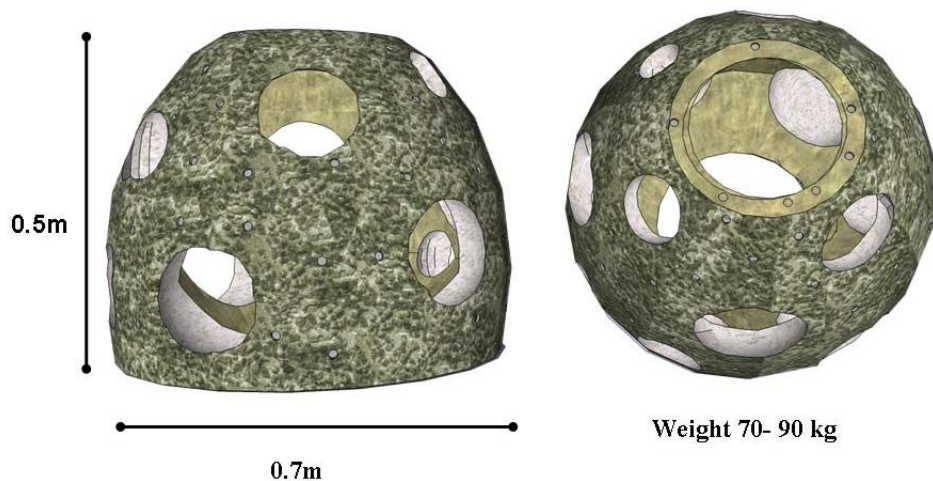
Figure 2.   Side view and top down view of 'Mini Bay' Reef Ball.

Japan and Korea are leaders in the research and development of purpose built, large-scale artificial reefs for fisheries enhancement. In 2001, Korea planned to invest over $2 billion (US) in coastal fisheries enhancement projects over a 6 year period (2002 – 2008). Many of the world's largest reefs have been deployed in Japan, as part of the national fisheries program for enhancement of commercial fish stocks. They include both shallow water reefs targeting shellfish and deeper water reefs for finfish (Figure 3).

Figure 3. Examples of 'design specific' artificial reef units (a) individual concrete cubes for deployment in deeper water (20 – 40m) (b) steel reefs designed for abalone production at shallower depths (10 – 20m).

1.2.   Artificial reefs in Australia

There have been at least four detailed reviews of artificial reefs in Australia (Pollard & Matthews 1985, Kerr 1992, Branden *et al.* 1994, Coutin 2001). Collectively these reviews detail the development of Australian artificial reef design, construction, deployment and monitoring from 1965 to 2001. Historically, the trajectory of artificial reef development in Australia is similar, albeit on a smaller scale to reef deployment world-wide with initial deployment of reefs constructed of waste material with little thought given to pre-deployment objectives or post-deployment assessment of the reefs performance.

Figure 4.   Initial deployment of artificial reefs in Lake Macquarie constructed from tyres.

In NSW, the use of artificial reefs began in 1966, with the deployment of a series of artificial reefs in Lake Macquarie using car bodies and tyres (Figure 4) (Pollard 1989). Augmentation of these reefs continued in the 1970's with additional reefs constructed in Batemans Bay, Port Stephens and Port Hacking using car tyres (Coutin 2001). In addition to these tyre and car body reefs, up to twelve vessels were scuttled beginning in the mid 1970's to create single-component artificial reefs in NSW coastal waters (Pollard & Matthews 1985, Pollard 1989). More recently a number of ex-naval vessels have also been scuttled in Australian waters for the purpose of creating artificial reefs for recreational SCUBA diving, including the former-HMAS Swan and -HMAS Perth (WA), the former-HMAS Hobart (South Australia) and the former-HMAS Brisbane (Queensland). In NSW, there are plans to scuttle the former-HMAS Adelaide offshore from Terrigal, (Central Coast, NSW).

In January 2000, the Premier of NSW and the Minister for Fisheries released the 'Sustaining our Fisheries' consultation paper, which suggested that funds generated from the sale of the general recreational fishing licence could be used to, among other things, construct artificial reefs in NSW. Specifically, the paper stated, "artificial reefs could be constructed in estuarine, inshore and

offshore waters to enhance recreational fishing. The effectiveness of these structures will be evaluated through scientific research programs".

In May 2002, 30 areas along the NSW coast were designated as fishing havens resulting in the closure of 24% of the States estuarine waters to commercial fishing. In 2004, I&I NSW (formerly NSW DPI) were successful in gaining funding for the deployment and monitoring of design specific Reef Ball artificial reefs within selected fishing havens. Initial constraint mapping identified potential areas that would be suitable for the deployment. Following an extensive period of consultation with relevant local government and community groups an artificial reef system was constructed in the Galgabba Point area of Lake Macquarie.

1.3.   Need and objectives

Lake Macquarie is a Recreational Fishing Haven. Recreational fishing is an important leisure activity for approximately 17 % of the NSW population (approximately 1 million people) and provides significant social and economic benefits (Henry & Lyle 2003). The central objective of the Artificial Reef Program is to enhance existing recreational fisheries and is consistent with statutory objectives and Departmental strategy "to promote quality recreational fishing opportunities", and "to provide quality recreational fishing….." The purpose of the research program is to assess the response of fish communities to the introduction of artificial structures, consistent with the Departmental Strategies to apply scientific knowledge to ensure fisheries are sustainably harvested by commercial and recreational fishers. Project objectives that relate specifically to the Lake Macquarie deployment are as follows;

- Determine differences in the number, size and species composition of fish assemblages among large and small artificial reefs, and naturally occurring reefs.

- Relate differences in the variables measured (number, size and species composition of fish) to changes in physical variables such as temperature, current speed/direction, turbidity and salinity.

- Evaluate the effectiveness of Baited Remote Underwater Video (BRUV) techniques for assessing fish populations around artificial reefs within estuaries.

- Transfer knowledge regarding construction, deployment and monitoring of artificial reefs in other suitable locations.

2.   MATERIALS AND METHODS

2.1.   Site selection

An initial constraint mapping exercise was used to identify areas within Lake Macquarie where artificial reefs would either be logistically impractical to deploy due to physical constrains (depth, sediment type, seagrass beds, protected areas) or interference with existing activities or planned developments (navigation channels, anchorages, marinas). Verification of potential sites identified by constraint mapping was followed up with sediment sampling and in-situ inspections to ensure correlation with existing information regarding the distribution of sensitive and protected benthic communities. The site-selection process was combined with extensive consultation involving community groups and relevant local government authorities. An area in southern Lake Macquarie adjacent to Galgabba Point (Figure 5) was selected for the initial deployment of six artificial reefs.

2.2.   Reef construction and design

Reefs were constructed using artificial reef modules (Mini-Bay Reef Balls®) individually cast from pH balanced micro-silica concrete, within fibreglass moulds and are treated to create a rough surface texture, in order to promote settling by marine organisms. The Reef Ball units are open at the top and have a large central void space created by a central bladder that is inflated for casting then deflated and removed after the concrete has hardened (see Figure 2). Reefs units were deployed individually using a commercial barge and hydraulic crane. All reefs were located on a uniform sandy bottom and the position of each reef was determined using GPS.

2.3.   Study area

A total of 180 artificial reef modules (Mini-Bay Reef Balls®) were deployed in six locations (AR1 – AR6) at Galgabba Point along the 5m depth contour in December 2005 (Figure 6). Each of the reef groups were located approximately 180m from the next with approximately 900m between AR1 and AR6. Reefs AR1, AR3 and AR5 were constructed from 50 reef balls and reefs AR2, AR4 and AR6 were constructed from 10 reef balls. All reefs were located on a uniform sandy bottom and larger and small reefs occupied a "footprint" of approximately $22m^2$ and $4m^2$ respectively. Areas of naturally occurring reef "reef control" locations" (RC1 – RC3) were also surveyed to provide a comparison of fish assemblages between artificial and naturally occurring reef systems.
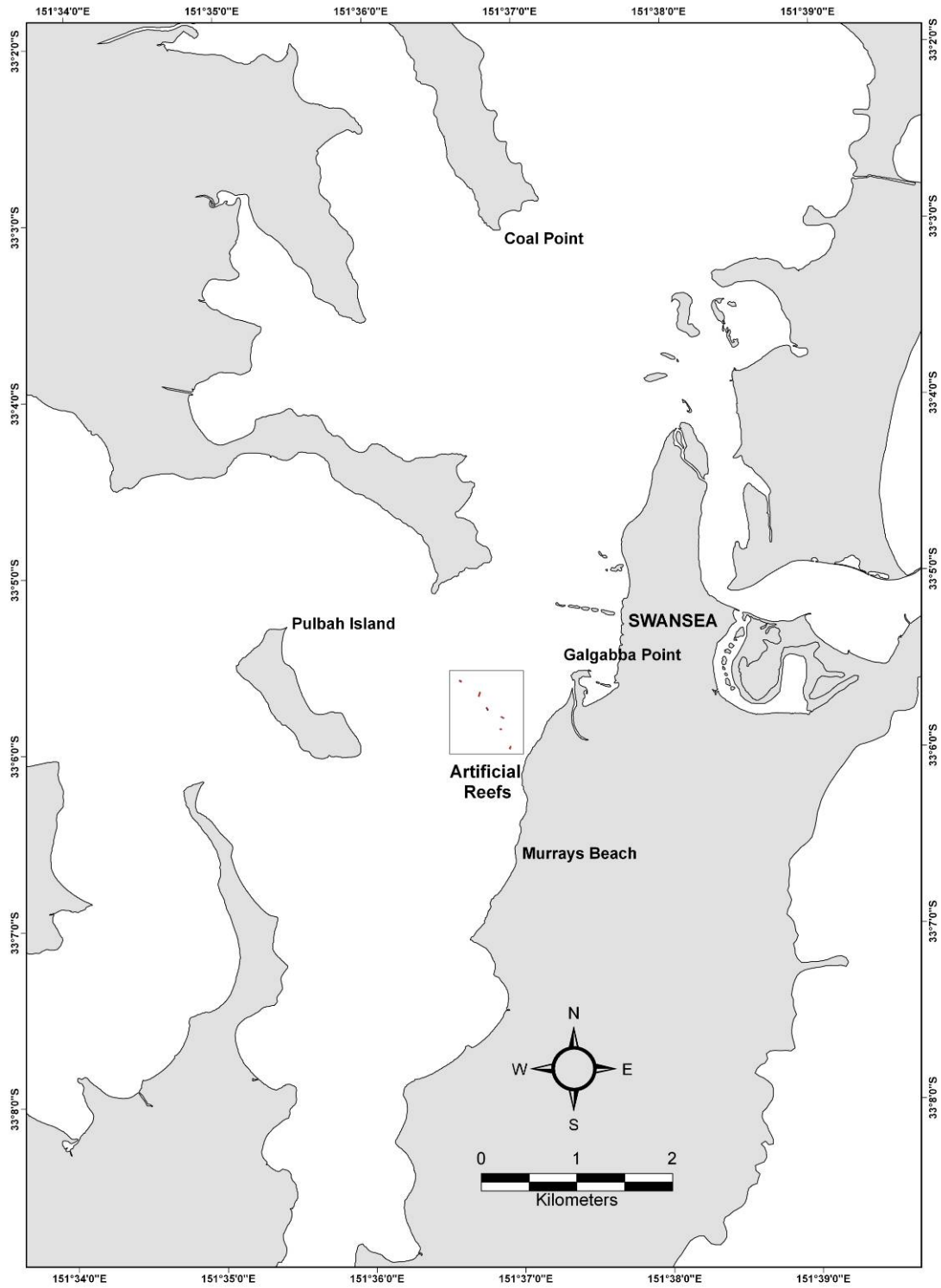
Figure 5. Map of southern Lake Macquarie indicating general location of the artificial reef complex.
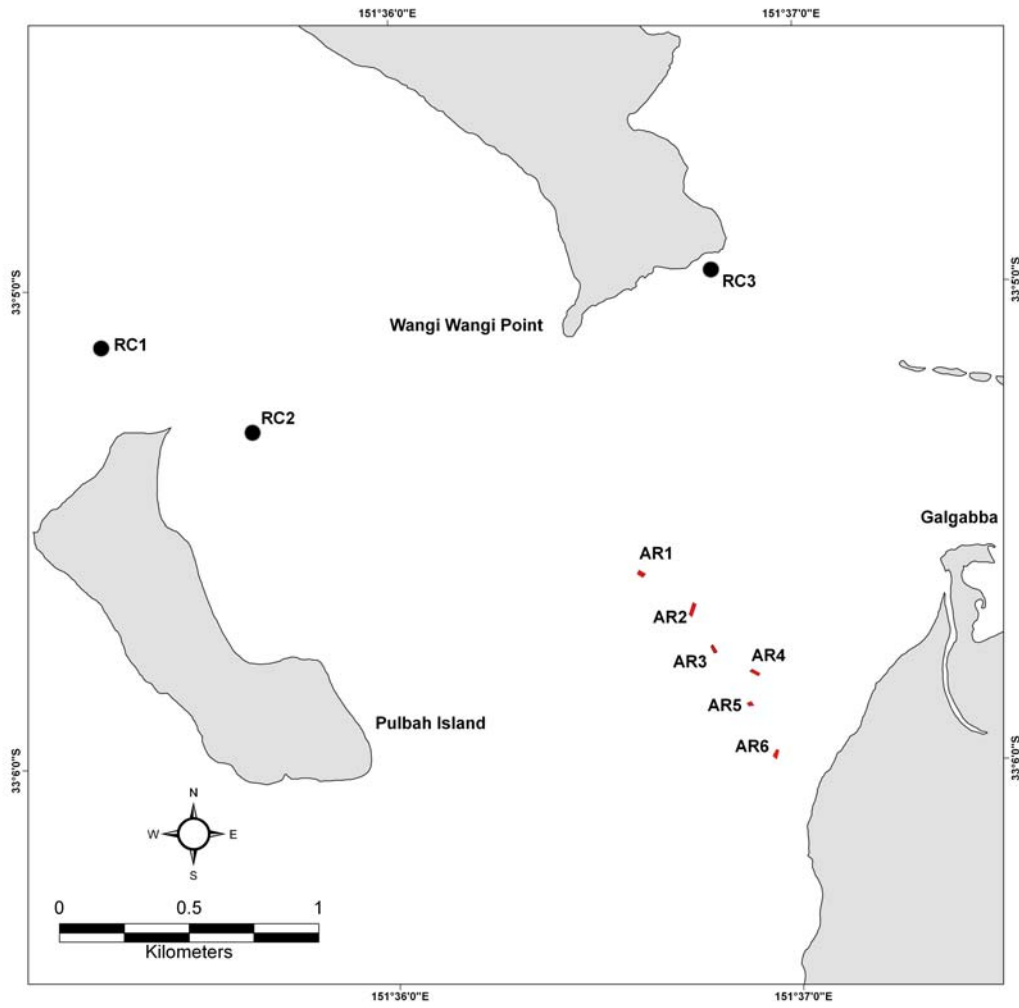
Figure 6. Map indicating the location of the six artificial reef units (AR1 – AR6) which define the Lake Macquarie artificial reef complex and natural or 'control reef' sites (RC1 – RC3).

2.4. Finfish community

Artificial reefs were surveyed using a combination of Baited Remote Underwater Video (BRUV) and direct (diver based) Underwater Visual Census (UVC) over a period of two years post deployment. Surveys were carried out on six randomly selected days per season. BRUV units were deployed for a period of 30 minutes at each site (artificial reefs and control locations) on each sample day. UVC observations were carried out every second season on artificial reefs only. The information obtained from the direct UVC observations was used as a means of evaluating the effectiveness of baited underwater video techniques for assessing fish populations around artificial reefs within estuaries.

2.4.1. Baited Remote Underwater Video (BRUV)

Three BRUV systems were built based on the design of Cappo *et al.* (2004) (Figure 7). A stainless steel frame was constructed to provide a platform for the camera housing. Video cameras each mounted inside a submersible housing were secured to the frame. A bait arm extending a distance of 1m from the face of the camera housing supported a plastic bait container. Units were baited using standardised bait that was replenished prior to every deployment. BRUV systems were deployed randomly at artificial reef (AR1 – AR6) and control locations (RC1 – RC3) for a period of 30 minutes. Analysis of BRUV tapes was carried out using the BRUVS tape reading interface 2.1. Information recorded from the tapes included the time to first sighting and, Max N (the maximum number of individuals of each species observed in one frame over the sampling period) and time of Max N for all species observed. See Cappo *et al.* (2006) for a detailed review of Max N as an estimator of relative abundance.

Figure 7.   Baited Remote Underwater Video (BRUV) used to estimate relative abundance (Max N) between artificial reef and control locations.

2.4.2. Underwater Visual Census (UVC)

Diver census was carried out by means of two counts, a five minute stationary count to identify all immediately visible species and a five minute roaming count to record the heavily reef-associated and cryptic species residing within the reef structure itself. When monospecific groups of more than fifty fish were observed, sets of individuals were used by the diver to estimate the total population. For schools of mixed species, the number was estimated for the entire school and an approximation made of the proportion of each species comprising the school.

2.4.3.   *Independent angler survey*

Independent angler surveys were conducted to detect differences in catch rate and species diversity between artificial reefs (AR1 – AR6) and natural occurring reefs (C1 – C4). Surveys were carried out over two sampling periods (25/8/2006 – 7/10/2006) and (8/1/2007 – 23/2/2007) with 7 survey days randomly allocated to each sampling period. Sampling was carried out using standardised recreational methods (hook and line) each site was randomly selected and sampled for 1 hour with all sites sampled on each sample day. Information collected included location of capture, species identification, total length (TL), fork length (FL) and weight (g). Catch per unit effort (CPUE) rate was interpreted as the number of fish caught per hour at each location. All fish were handled in a manner designed to minimise stress and reduce mortality (Broadhurst *et al.* 2005) and returned at the point of capture as soon as possible.

2.5.   Epibenthic community

Succession of the epibenthic community associated with the artificial reefs was monitored as part of the UVC. Six random replicate photo-quadrats representing approximately 0.5m^2 of the reef's exterior surface were taken at each artificial reef site (AR1 – AR6) by SCUBA divers using a digital camera fitted with a waterproof housing (Figure 8). Monitoring began approximately nine months (267 days) post-deployment with each of the reefs photographed 24 times over the 14 month sampling period. Material was collected and preserved directly from the reef or images were enhanced and forwarded to experts for positive identification when necessary. Samples were identified to the lowest taxonomic level when possible and were also categorised as functional groups based on morphology (Fowler-Walker & Connell 2002). Sampling and analysis of the epibenthic community was only done for artificial reef sites.

Figure 8.   Photographic surveys were used to document the growth of the epibenthic community associated with the artificial reefs.

2.6.   Data analysis

2.6.1.   Fish community

Each of the species was classified on the basis of their vertical distribution inside the water column and their position relative to the reef, collectively defined as ecological groups (Nakamura 1985) (Figure 9). The first group (Type A) included species that tend to have direct contact with the reef structure itself, and often occupy crevices, holes or gaps within the reef. The second group (Type B) included species found in the immediate vicinity, but not coming into direct contact with the reef. The third group (Type C) included more transient species that did not demonstrate any site associated attachment to the reef but were observed to move through the reef area.

Percent sighting frequency was defined as the percentage of all survey days in which the particular species or family was recorded by each method (UVC and BRUV). The percentage occurrence of each species was categorised into four groups; permanent species (>75%), frequent species (75 – 30%), scarce species (30 – 10%) and rare species (<10%) (Tessier *et al.* 2005). To identify trends in the development of the fish assemblage, time post deployment was categorised into four periods (0 – 6, 7 – 12, 13 – 18 and 19 – 24 months). Multivariate analysis (MDS) was used to identify relationships between UVC counts and relative abundance estimates generated by BRUV on artificial reefs. A combination of multivariate (MDS) and univariate (ANOVA) techniques were used to analyse differences in estimates of relative abundance (Max N) between artificial and control locations and changes in species composition associated with artificial reefs over time. The ratio of the average similarity and standard deviation (Sim/SD) (SIMPER) for artificial reef age groupings is given as a measure of how consistent a given species contributes to the characterisation of species within each reef age group. Species displaying a high average similarity/standard deviation ratio (>1.0) were regarded as 'key species' associated with each reef age grouping.

2.6.2.   Epibenthic community

Photo-quadrats were analysed in the laboratory using the image analysis computer program Coral Point Count (Kohler & Gill 2006). One hundred random points were superimposed over each photo-quadrat image and each point was assigned to the taxa located beneath it (Bohnsack 1979, Clynick *et al.* 2007, Walker *et al.* 2007). Point count data provided the basis for the calculation of percentage cover of each taxa identified on each artificial reef (AR1 – AR6).

Multivariate analysis (MDS) was used to identify differences in assemblage structure between reefs. The relationship between time (days post-deployment) and mean abundance (% cover) of taxa indentified was further investigated using CHAID analysis (Chi squared automatic interaction detector) (Kass 1980). CHAID is an exploratory method used to study the relationship between a dependent variable, in this case days post-deployment and a series of predictor variables (percentage cover of each taxon).

2.6.3.   Angler survey data

Catch per unit effort (CPUE) were expressed as the number of each species captured per angler hour. Kruskal Wallace tests were used to identify significant differences in CPUE and the weight (g) and fork lengths (cm) of the primary species captured at each location.
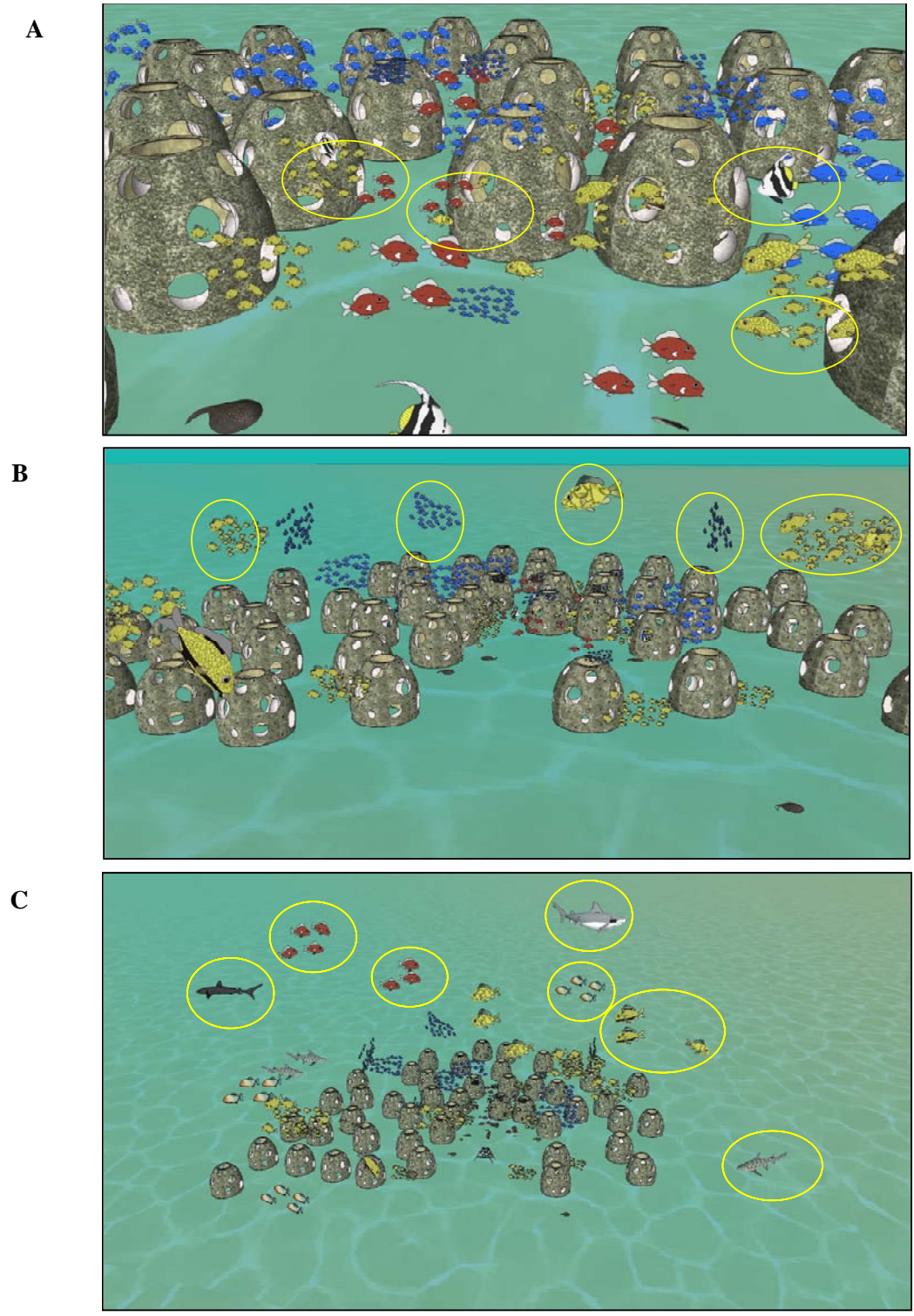
Figure 9. Species identified were classified as type A, B or C depending on the level of association with the reef structure.

3.   RESULTS

3.1.   Fish community (artificial reefs)

Fifty-one species belonging to 27 families were observed during analysis of BRUV information or directly during diver surveys associated with artificial reefs. Categorisation by ecological grouping identified the majority of species as either closely associated within the reef structure (Type A: 41%) or species found in the immediate vicinity of the reef (Type B: 49%) with a minority (10%) of transient or Type C species observed. Five species (yellowfin bream, snapper, tarwhine, striped trumpeter and cardinal fish) were classified as permanent (>75% of observations), however the majority of species (86%) identified were classified as scarce or rare (see Appendix 1).

Recruitment to the artificial reefs was relatively rapid with 77% of all species recorded identified within the first year (Figure 11). Key colonising species included striped trumpeter, snapper and tarwhine; all species with the exception of striped trumpeter are targeted by the recreational angling community. Following the initial colonisation of the artificial reef group, recruitment of new species remained relatively rapid and within 6 months cumulative species richness had doubled to sixteen species.

Sighting frequency values for species identified by BRUV and UVC indexed by ecological groupings indicated a relationship between observation method and ecological group. UVC identified a greater proportion of rare Type A species which represented 35% of all species identified by this method, compared to 12% identified by BRUV (Figure 10).
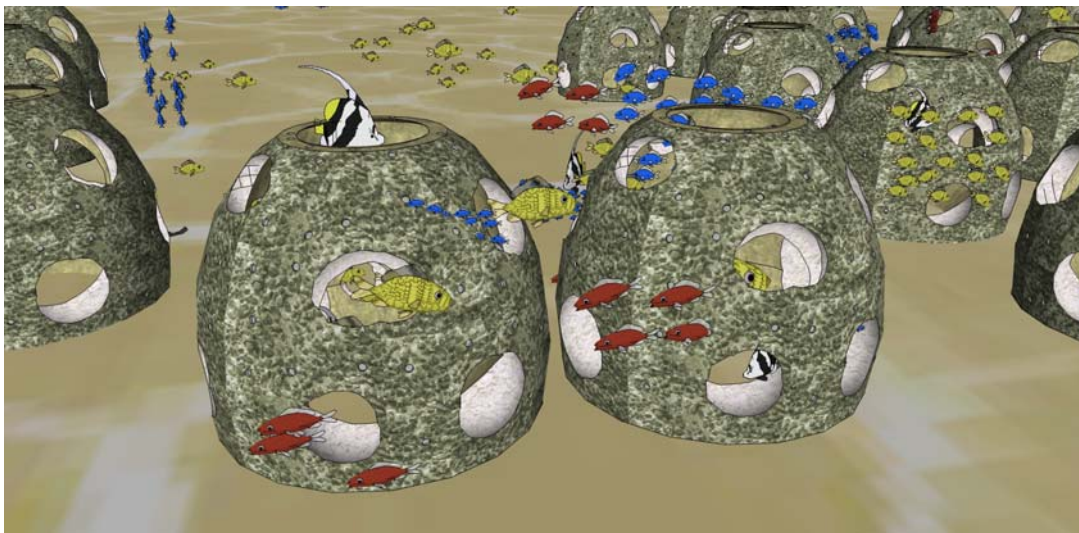
Figure 10.   Underwater Visual Census (UVC) was particularly important for the detection of the more cryptic 'Type A' species most often found within the structure of the reef.
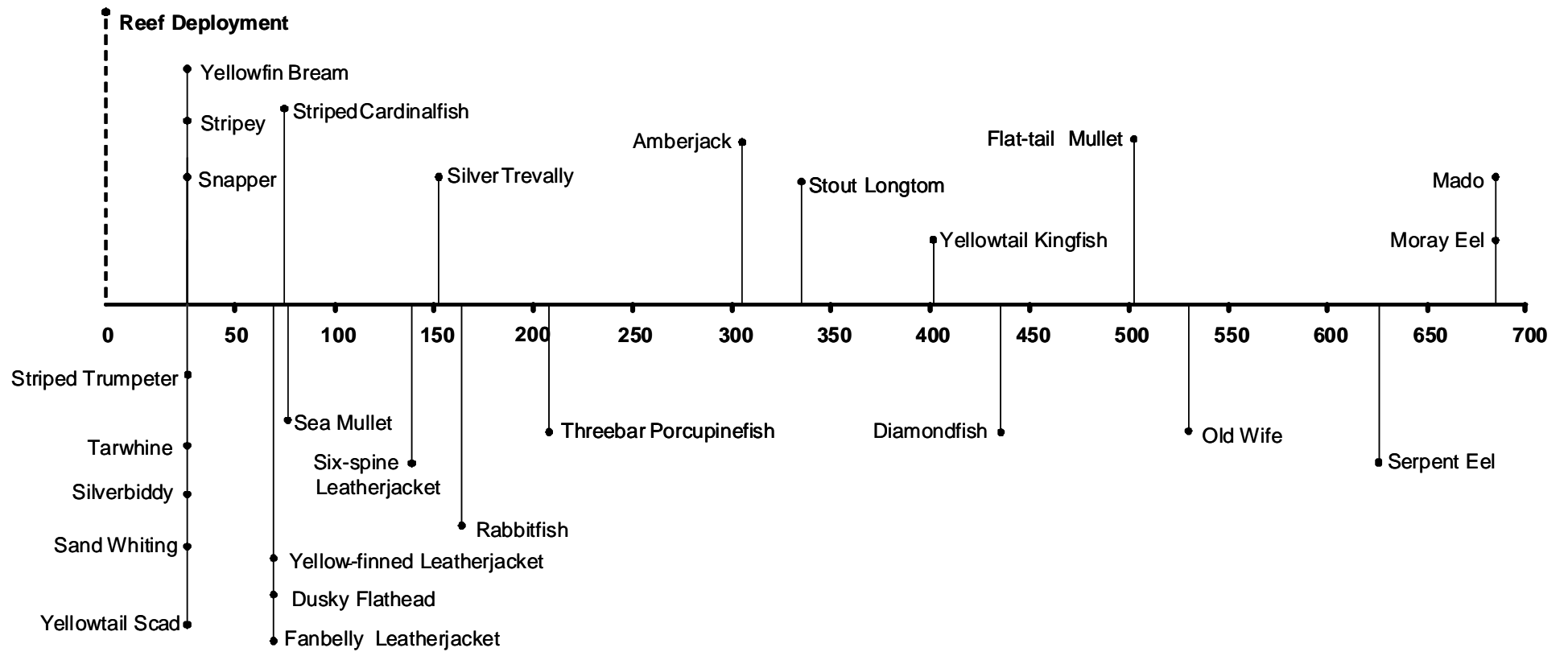
Figure 11. Timeline indicating the appearance of each species against time (number of days) post deployment.

Significant differences in the structure of the fish assemblage with the time post-deployment (reef age) was observed. Relative abundance of striped trumpeter was significantly greater during the initial twelve month period post-deployment. In contrast, significantly higher abundance estimates of yellowfin bream and tarwhine were associated with the older reef age groupings. The results for snapper were not as well defined, with varying estimates of relative abundance across the reef age groupings (Figure 12).
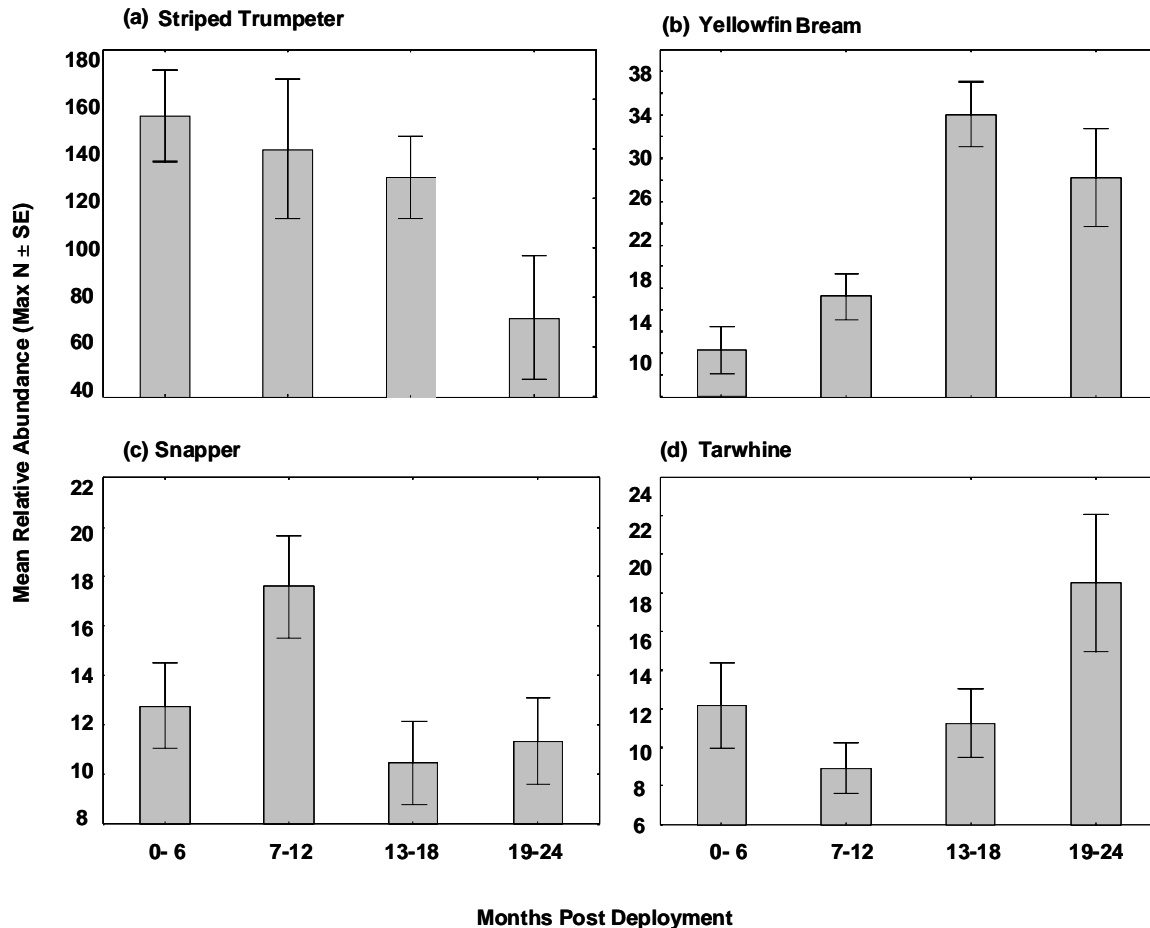
Figure 12. Mean relative abundance (Max N ± SE) by reef age categories for key colonising species.

3.2. Comparison between artificial and natural reefs

BRUV sampling identified a total of twenty-seven species on artificial reefs compared to twenty-one species on naturally occurring reefs (see Appendix 2). Nine species (yellowtail kingfish, diamondfish, stout longtom, bartail goatfish, mado, serpent eel, old wife, dusky flathead and moray eel) were identified on artificial reefs only. Two species, the large-toothed flounder and the striped scat were observed on natural reefs only. Of the nineteen species observed on both artificial and natural reefs, fourteen species recorded greater mean abundance (Max N) on artificial reefs than naturally occurring reef locations.

Analysis of mean abundance estimates identified significant differences in species assemblages between reef types. Species contributing most to the differences between groups were striped trumpeter, yellowfin bream, snapper and tarwhine. Striped trumpeter and snapper were on average, three times more abundant on artificial reefs than on natural reefs. Yellowfin bream however, were approximately twice as abundant on natural reefs. Other species making significant contributions were tarwhine and yellowtail scad both with higher average abundances on the artificial reefs (Figure 13).
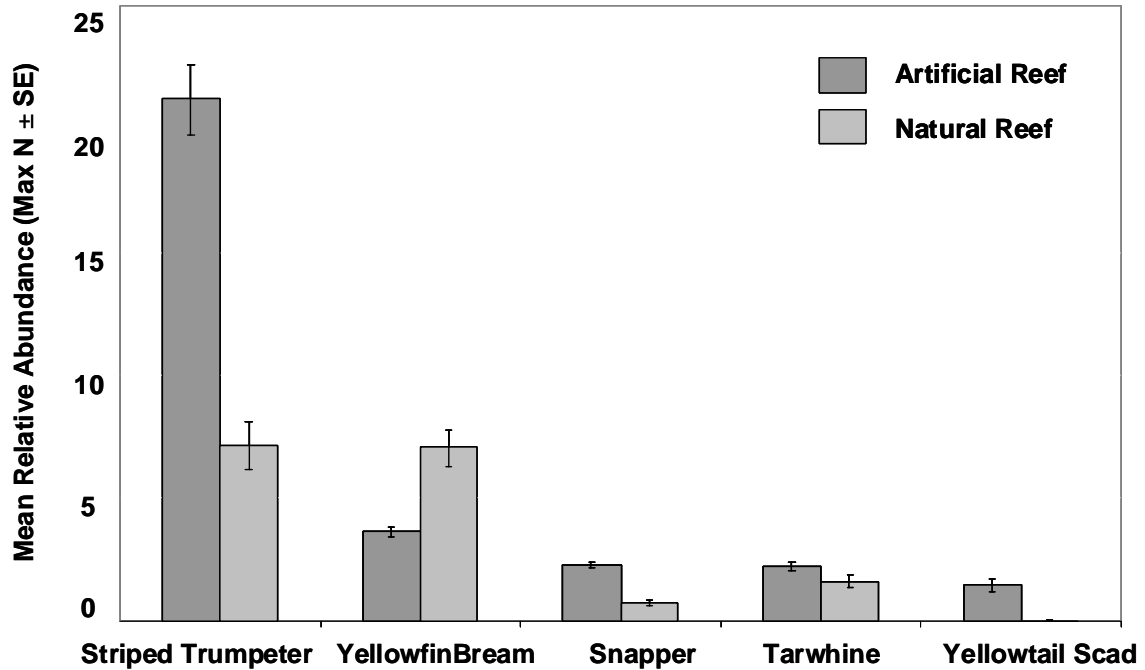
Figure 13.   Species identified to contribute most to differences in mean relative abundance (Max N ± SE) between artificial and natural reefs.

3.3.   Angler survey

A total of 295 fish were captured, resulting in the identification of thirteen species. The number of species identified varied from a high of ten at the artificial reef to a low of eight at two of the control locations (C1 and C3). Tarwhine, snapper and sand whiting accounted for between 70% and 75% of the total catch on both artificial and natural reefs. Two species, the three-barred porcupine fish and the silver biddy were captured on artificial reefs but not captured at any of the control locations. A further four species (rough leatherjacket, fanbelly leatherjacket, tailor and yellowtail scad) were identified on natural reef but not captured on artificial reefs.

Total catch rates varied between locations from a high of 5.5 (±0.9) fish per hour at the artificial reef with remaining control locations ranging between 4.3 (±0.6) at C1 to 3.6 (±0.4) at C4 (Figure 14), however, there were no significant differences in total catch rates between artificial reefs and control locations. No significant differences in catch rates between artificial reefs and natural reefs for individual species were identified; however, catch rates for both snapper and tarwhine were on average 35% and 28% greater respectively on artificial reefs, whereas catch rates of sand whiting were marginally greater on natural reefs (Figure 15).
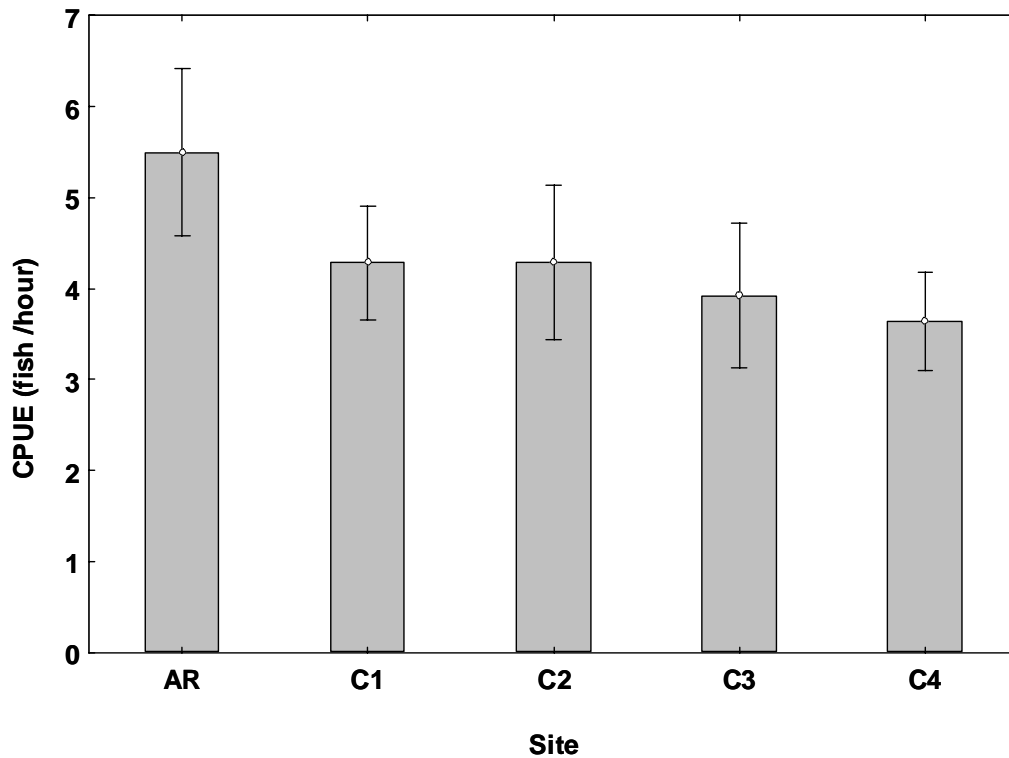
Figure 14. Mean catch per unit effort (CPUE) (±SE) for artificial reef (AR) and control locations (C1 – C4).
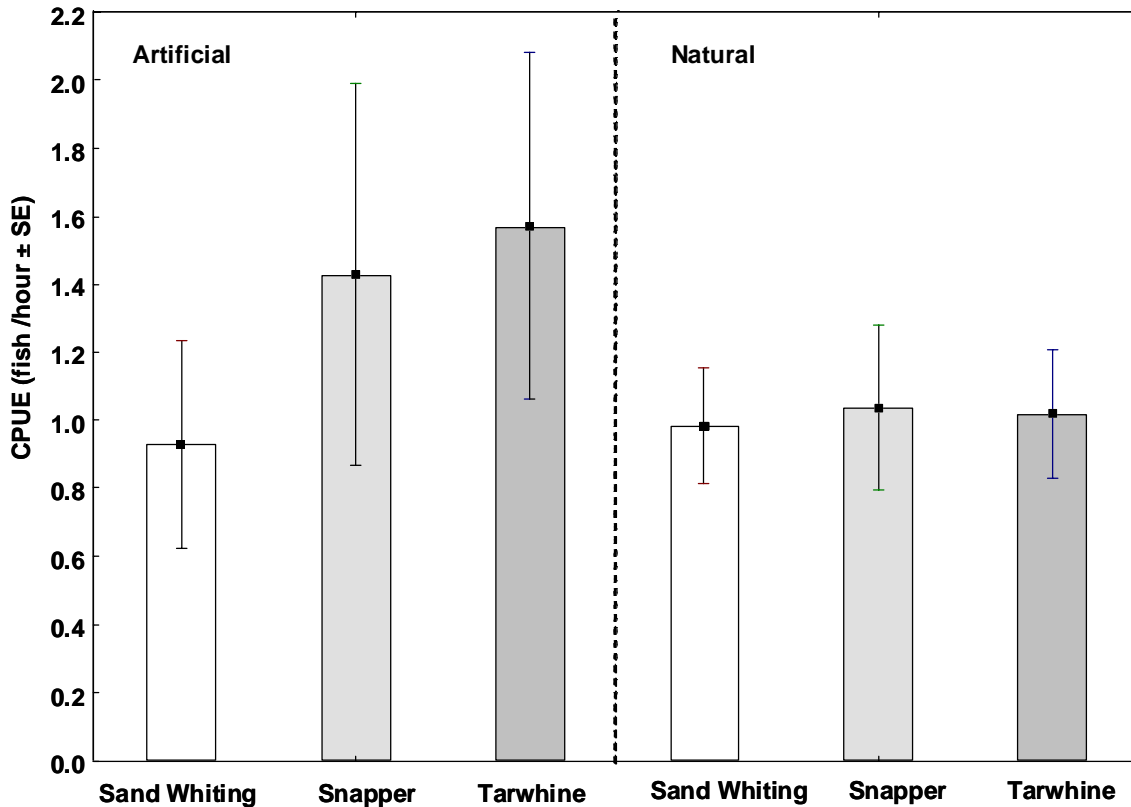
Figure 15. Comparison of mean catch per unit effort (CPUE) (±SE) for primary species between artificial and natural reefs.

Analysis of fish length and fish weight data for the three most abundant species (tarwhine, snapper and sand whiting) indicated a variable response between species and locations (Figure 16). Artificial reefs did however record the highest mean weight and length for tarwhine and sand whiting.
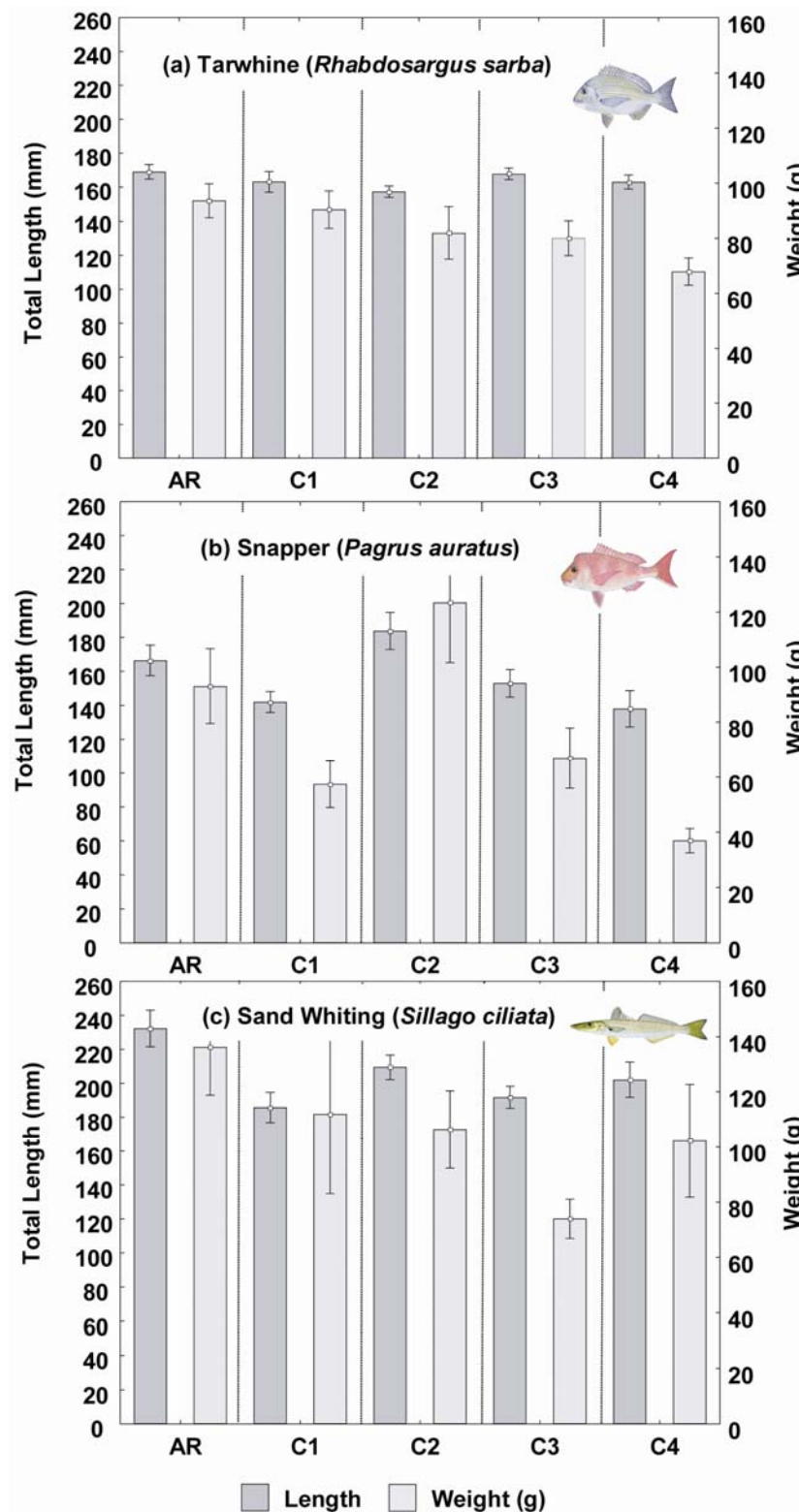
Figure 16. Comparison of mean total length (cm) and weight (g) (±SE) for tarwhine, snapper and sand whiting between artificial (AR) and control locations (C1 – C4).

3.4. Epibenthic community

Recruitment of the epibenthic community was relatively rapid with all taxa identified within the first year post deployment. The community associated with the artificial reefs was characterised by low diversity with only one algal species (*Spyridia filamentosa*) and three species groups (filamentous turfing algae, polychaetes and echinoderms) recorded. There were no significant differences identified in the percentage cover between artificial reefs. Filamentous turfing algae (FTA) and *Spyridia filamentosa* were the dominant taxa recorded on all reefs with a mean percentage cover for the entire study period of 76.8% and 22.3% respectively.

Figure 17. Extensive growth on artificial reef (AR5) 23 months after deployment.

Analysis of the percentage cover information with reef age (days post-deployment) identified a significant reduction in the abundance of FTA with an increase in reef age. An increase in the coverage of FTA was linked to the proportional coverage of polychaete species. Conversely a reduction in the coverage of FTA with an increase in reef age was identified with variation in the coverage of *Spyridia filamentosa* and low abundance of polychaetes.

4. DISCUSSION

4.1. Recruitment and succession of fish

The development of the fish community associated with the artificial reef was characterised by an initial rapid increase in recruitment of species over the first year post-deployment, followed by a moderate decline and levelling off over the remainder of the two year sampling period. The observed pattern of recruitment is consistent with a range of studies which have identified a similar response from a diverse range of species as a result of artificial structures deployed in a variety of locations worldwide (Bohnsack & Talbot 1980, Bohnsack & Sutherland 1985, Matthews 1985, Walsh 1985, Haughton & Aiken 1989, Cummings 1994, Golani & Diamant 1999, Manderson & Able 2003, Markevich 2005). What is consistent between all these studies, regardless of the artificial reef material, location, size, or time of deployment, is the rapid initial recruitment, followed by a levelling off and in some cases a reduction in the number of fish species identified.

The colonisation and subsequent changes in the structure of fish communities associated with artificial reefs has been defined by several theories, the most widely accepted being the theory of island biogeography (MacArthur & Wilson 1967). Classic island biographic principles indicate that the pattern of succession will be a result of the rate of movement of the colonising species, the distance from the source of new recruits and the size of the area being colonised (Walsh 1985). While models based on the concepts of island biogeography provide some of the guiding concepts associated with recruitment and community development, understanding the complexities of fish communities associated with artificial reef systems requires a more detailed assessment of physical and biological aspects of the environment and the associated fish assemblage.

The rapid colonisation to artificial reefs is often hypothesised 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). Sale (1980) hypothesised that the resultant community structure of artificial reefs is primarily influenced by opportunistic recruitment determined by factors such as the time of deployment (seasonality) and the proximity of the deployment site in relation to other sources of recruitment. Variation in the ontogenetic, physical and behavioural parameters of species (Pizzolon *et al.* 2008) as well as the proximity and degree of connectivity among suitable habitat patches in relation to the artificial reef, have also been demonstrated as factors that will influence the rate of colonisation and the resilience of the developing community to respond to post settlement processes such as competition and predation (Walsh 1985, Herrera *et al.* 2002, Fernandez *et al.* 2007).

While some reef species are capable of moving over bare sand for feeding (Ambrose & Anderson 1990), others are reluctant to cross it (Coll *et al.* 1998, Chapman & Kramer 2000, Fernandez *et al.* 2007). Medium sized mobile fish are least influenced by reef isolation or low habitat connectivity (Ault & Johnson 1998, McClanahan & Mangi 2000, Fernandez *et al.* 2007) with extensive sand patches perceived as barriers of variable permeability in relation to the size and vagility of fish species (Bell & Westoby 1986, Stamps *et al.* 1987, Coll *et al.* 1998). Striped trumpeter, yellowfin bream, snapper and tarwhine were all species identified as being either frequently or permanently associated with artificial reefs and were probably recruited from areas of adjacent natural habitat where they were known to inhabit as post-settlement juveniles, sub-adults or adults (Miskiewicz 1987, Hannan & Williams 1998). The rapid colonisation of these species to the artificial reefs is a result of the ability of these species to move relatively large distances over a variety of habitat types that act as a barrier for other less mobile reef associated species.

While the potential of species to colonise the reef structure may be mediated by behaviour 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 a 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 *et al.* 1983, Shulman 1985, Hixon & Beets 1991, Hixon 1993, Overholtzer & Karen 2004). While the effect of predation in structuring artificial reef communities remains poorly understood, more recent studies indicate significantly higher visitation rates and a greater diversity of larger predators 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 more likely to respond to larger aggregations of prey species (Stewart & Jones 2001, Connell 2002) which may result in an increase in the proportional mortality of aggregated prey, since the predators may feed at a greater rate (Connell 2000, Connell 2002). Numbers of striped trumpeters, a highly abundant schooling species were found to decrease significantly with increasing reef age. Striped trumpeter were initially identified around artificial reefs in large numbers, with mean relative abundance estimates of greater than 150 which decreased to less than 70 over the sampling period. Scars and injuries (e.g. bites and partial removal of fins) were regularly observed on individuals during analysis of BRUV data. Herrera (2002) observed that specific predator species were chiefly responsible for controlling the structure of artificial reef fish assemblages. Two important transient predators – the greater amberjack and yellowtail kingfish were regularly identified by diver observations and on BRUV tapes in the vicinity of the Lake Macquarie artificial reef complex. These species are more commonly associated with inshore and coastal shelf waters (particularly greater amberjack*s*), not usually encountered in the upper reaches of coastal estuaries (Kailola *et al.* 1993) and either not identified (yellowtail kingfish) or encountered less frequently (greater amberjack) at the natural reef locations. 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 striped trumpeter, 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 striped trumpeter may have potentially presented an opportunity for new species to recruit to the reef or existing reef species such as yellowfin bream in significantly higher numbers. However, when interpreting these results, we were careful to note that although a significant decrease in numbers of striped trumpeter was detected, any corresponding increase in other species (e.g. yellowfin bream) may potentially be an artefact of inherent bias associated with the use of BRUV systems. The 'density saturation effect' of species such as striped trumpeter which appear in large numbers as soon as the BRUV system arrives on the sea floor has been demonstrated to under represent species such as yellowfin bream, snapper and tarwhine particularly in the initial phase post-deployment when numbers of striped trumpeter were highest.

4.2.   Comparison between artificial and natural reefs

Evaluation of the relationship between natural and artificial structures is critical in developing an understanding of the dynamics associated with the development of artificial reefs and the importance that natural structures play in the development of fish communities associated with artificial structures. The results of this study support previous findings which demonstrate that artificial reefs are rapidly colonised and usually develop fish communities with comparable or higher abundance and diversity than surrounding natural reefs (Bohnsack *et al.* 1994, Stephens Jr *et*

al. 1994, Pickering & Whitmarsh 1997, Rilov & Benayahu 2000, Pondella *et al.* 2002). Although similarities between artificial reefs and naturally occurring reefs were evident, variation in relative abundances and species diversity were observed. It is unclear whether this is a result of inherent structural differences between the artificial and naturally occurring reefs or that the results represent an initial phase in the succession of the artificial reef assemblage as it moves to a "climax community".

Variation in fish communities between natural and artificial reefs are often explained by differences in reef size and age, however many artificial reefs are either to small or not monitored for long enough periods to provide an adequate comparison. More recent studies (Burt *et al.* 2009) that have compared large (>400,000 m^3) artificial reefs with similar areas of natural reef have identified that at least for coral reef systems, significant differences in fish assemblage persist between natural and artificial systems, even over extensive periods of time (>25 years). More detailed information regarding the movement patterns of fish within the artificial reef system and between artificial reefs and other naturally occurring habitat is required to evaluate the contribution of artificial structures to the productivity of environments in which they are deployed.

4.3.   Recreational fisheries enhancement

The information collected by the independent angling component of the study represents a "snapshot" obtained at a relatively early stage (between 8 and 15 months) post deployment. The survey work represents a starting point in developing a more comprehensive understanding of how artificial structures can act to enhance recreational fisheries within estuarine systems. Higher total catch rates and increased catch rates of the more mobile species such as snapper and tarwhine support the findings of previous similar studies (Buckley & Hueckel 1985, Matthews 1985) and the more general findings of long-term monitoring which consistently identified higher number of species and higher mean catch rates on artificial reefs compared to control locations (D'Anna *et al.* 1994, Fabi & Fiorentini 1994, Bombace *et al.* 2000, Zalmon *et al.* 2002, Santos & Monteiro 2007).

Anglers support for artificial reef programs is based on the expectation that the deployment of artificial reefs will result in an improvement in 'fishing quality'. Understanding how indices of fishing quality such as catch rates and the number and type of species that can be retained are influenced by the deployment of artificial reefs is central to the assessment of artificial reef programs that have fisheries enhancement as one of the primary objectives. While the ability of artificial reefs to enhance recreational fisheries through increased production remains an open question, it is evident that artificial reefs have the ability to provide habitat to a broad range of species and that catch rates for the majority of key recreational species are not significantly different to natural reefs. Comparison with previous 'access point' surveys of recreational anglers within Lake Macquarie (Steffe & Chapman 2005) identify similar patterns in diversity and relative proportions of species harvested, however direct comparisons of catch rates and size information are limited difficulties in comparing catch information collected independently compared to harvest estimates which are ultimately constrained by current bag and size limits.

Survey work carried out as part of the broader study (UVC and BRUVs) was limited in its ability to provide accurate size information of the fish assemblage associated with artificial reefs. It is recommended that future surveys incorporate methods such as stereo photogrammetry that will provide accurate size frequency information to enable the integration of this information with management measures such as size and bag limits. Further recommendations include angler survey designs which enable assessments of effort directed to artificial reefs in order to better understand the patterns of use and the potential social and economic benefits associate with the development of artificial reef deployments.

4.4.   Epibenthic assemblage

Studies of epibenthic communities associated with sub-tidal artificial structures in NSW estuaries have been limited to assessments of non design-specific structures including marina seawalls and wharves (Glasby 1999, Glasby & Connell 1999, Chapman & Clynick 2006, Clynick *et al.* 2007). The results of these studies and other similar studies have established that benthic assemblages on hard reef surfaces (natural or artificial) are strongly influenced by a wide range of environmental variables which include water depth (Moura *et al.* 2007, Rule & Smith 2007), orientation in relation to prevailing currents (Baynes & Szmant 1989, Abelson 1994), orientation of surfaces (Glasby & Connell 2001, Knott *et al.* 2004), complexity of surfaces and structure (Edwards & Smith 2005, Moura *et al.* 2007) and are linked to the establishment of migratory and stationary fish assemblages (Aburto-Oropeza & Balart 2001, Edwards & Smith 2005, Chapman & Clynick 2006, Clynick *et al.* 2007, Redman & Szedlmayer 2009).

Factors that are particularly important in the early stages of development of the epibenthic communities associated with artificial structures are the proximity to other forms of habitat and the speed and direction of currents. Isolated artificial structures are considered to be closed or semi-closed systems, the distance from natural reef or other artificial substrata which act as a source of supply of juvenile settling larvae or spores is critical in determining the patterns of epibenthic succession (Connell & Slatyer 1977, Bellan & Bellan-Santini 1991, Bombace *et al.* 1994). Flow rates determine to a large extent how far these propagules and larvae will travel (Denny 1985, Butler 1991, Judge & Craig 1997, Hurd 2000, England *et al.* 2008). Water motion will also affect epibiotic community development by determining the rate at which inorganic nutrients and carbon dioxide (CO_2) are consumed by epibiotic species (Denny 1985). Low water movement reduces the rate at which this metabolic process takes place and therefore inhibits growth whereas increased water motion facilitates the metabolic process to enhance growth (Denny 1985, Judge & Craig 1997).

Lake Macquarie is classified as a 'wave dominated' estuary constricted by a single narrow opening to the sea resulting in low tidal and fluvial flow rates. The remote location of Lake Macquarie's artificial reefs in relation to any extensive areas of naturally occurring reef and the low levels of water movement associated with this estuary explain the relatively low diversity and slow development of the epibenthic assemblage associated with the artificial reef complex. The low flow conditions and the relative high sediment loads associated with the Lake Macquarie system have also influenced the development of the community. The ability of filamentous turfing algae to trap sediments enables them to thrive in environments with high sediment loads (Connell 2003, Connell 2005). Our results are in agreement with previous descriptive and experimental studies that document the dominance of filamentous taxa in subtidal areas subjected to high rates of sedimentation (Airoldi 1998, Irving & Connell 2002, Gorgula & Connell 2004, Balata *et al.* 2005).

The majority of estuarine systems in south eastern Australia are similar to Lake Macquarie i.e. low energy, wave-dominated estuaries. Additional deployments of artificial reefs in these areas will therefore most likely result in the development of epibenthic communities characterised by low diversity. Priorities for future work include ongoing monitoring of epibiotic communities associated with estuarine artificial reefs with a more detailed assessment of the interior of these structures, combined with an integrated fish monitoring program to better understand the longer-term relationship between epibiotic communities and fish assemblages associated with artificial structures.

4.5.   Evaluation of methodology

Results indicate that BRUV is an effective method for recording species associated with artificial reefs with the exception of cryptic species that are located within the reef structure itself. The results of this study underline previous findings (Connell & Kingsford 1998, Willis *et al.* 2000, Cappo *et al.* 2004) which emphasise the importance of using complementary methods to obtain accurate estimates of species diversity and abundance. UVC methods are essential for the collection of baseline data i.e. the number and type of species associated with artificial structures to provide a comprehensive understanding of how artificial structures differ from natural areas of reef, and how the assemblages associated with these structures vary in space and time.

BRUV techniques complement UVC by providing increased coverage of species known to be diver averse as well as providing important information regarding behaviour of the species identified. Given the limitation of each method, it is recommended that monitoring plans for artificial structures should adopt a multi-method approach utilising BRUV and UVC where possible. This will aid in establishing a comprehensive picture of spatial and temporal variation in the structure of species assemblages and enable 'calibration' of the bias associated with each method. The allocation of resources to each method will largely depend on logistical considerations and the objectives of the program.

5.   CONCLUSIONS

On the basis of the above findings, the following conclusions and recommendations are made:

1. This study has provided a clearer understanding of the type of fish and epibenthic assemblages that result from the deployment of an estuarine artificial reef and is the first study to demonstrate that 'design specific' artificial structures can be effective at extending the habitats of a variety of fish species in temperate estuarine systems in southeast Australia. Results show a rapid recruitment of a diverse range of fish species over the first year post-deployment, followed by a moderate decline and levelling off over the remainder of the two year sampling period.

2. Factors that may be influencing the rate and composition of the fish assemblage include the proximity of the artificial reef complex to naturally occurring habitat and the ability of species to move between existing habitat and the artificial structures. Post-settlement processes, particularly predation is believed to play an important role in the observed changes in the fish community associated with the artificial structures over time.

3. Results indicate that BRUV is an effective method for recording species associated with artificial reefs with the exception of cryptic species that are located within the reef structure itself. Given the limitation of each method, it is recommended that monitoring plans for artificial structures should adopt a multi-method approach utilising BRUV and UVC where possible.

4. A critical element in understanding how artificial reefs can be integrated into a more general marine resource management framework is an ability to evaluate their performance. Priorities for future work include ongoing monitoring of epibiotic communities associated with estuarine artificial reefs combined with an integrated fish monitoring program to better understand the longer-term relationship between epibiotic communities and fish assemblages.

5. It is recommended that observations using non-destructive photogrammetric techniques such as stereo-video are incorporated into future monitoring programs to provide accurate size frequency information of species associated with the artificial reefs and determine how compatible this fishery enhancement initiative is with existing management input controls such as bag and size limits.

6. While the ability of artificial reefs to enhance recreational fisheries through increased production remains an open question, it is evident that artificial reefs have the ability to provide habitat to a broad range of species and that catch rates for the majority of key recreational species are not significantly different to natural reefs.

7. A more detailed assessment of movement patterns of fish within the artificial reef complex and between the artificial reef and other natural habitat is required to better understand the role that artificial reef systems play in influencing the ecology of areas in which they are deployed. A more comprehensive understanding of movement and residency times of the species associated with artificial structures will enable an assessment of the competency of artificial structures as habitat and a better understanding of their contribution to fisheries production.

6.   APPENDICIES

Appendix 1.   Mean diver counts (UVC) and Max N (BRUVS) values ± standard error (SE), 'ecological class' (A, B, C) and residency class (permanent, frequent and scarce) for each species identified on artificial reefs (AR1 – AR6) only.

Common Name	Species	Class	DIVER			BRUV		
			Residency	(UVC)	± se	Residency	(Max N)	± se
Striped Cardinalfish	*Apogon fasciatus*	A	Permanent	208.090	17.062	Scarce	0.53	0.18
Striped Trumpeter	*Pelates sexlineatus*	B	Permanent	97.521	11.552	Permanent	19.47	2.02
Snapper	*Pagrus auratus*	B	Permanent	4.667	0.457	Permanent	2.75	0.19
Yellowfin Bream	*Acanthopagrus australis*	B	Frequent	4.292	0.579	Permanent	4.48	0.36
Tarwhine	*Rhabdosargus sarba*	B	Frequent	4.076	0.491	Permanent	2.83	0.24
Leatherjacket (Fanbelly)	*Monacanthus chinensis*	A	Frequent	1.854	0.174	Scarce	0.24	0.05
Stripey	*Microcanthus strigatus*	A	Frequent	1.583	0.193	Scarce	0.36	0.08
Leatherjacket (Yellowfin)	*Meuschenia trachylepis*	B	Scarce	0.535	0.107	Frequent	0.61	0.10
Silver Biddy	*Gerres subfasciatus*	B	Scarce	2.944	0.915	Rare	0.20	0.11
Leatherjacket (Yellowtail)	*Meuschenia flavolineata*	B	Scarce	0.569	0.106	~	~	~
Trumpeter Whiting	*Sillago maculata*	B	Scarce	0.903	0.291	~	~	~
Yellowtail Scad	*Trachurus novaezelandiae*	B	Scarce	4.139	1.076	Scarce	1.94	0.39
Fortescue	*Centropogon australis*	A	Scarce	0.313	0.092	~	~	~
Moray Eel	*Gymnothorax prasinus*	B	Scarce	0.188	0.042	Rare	0.01	0.01
Diamondfish	*Monodactylus argenteus*	B	Scarce	0.188	0.046	Rare	0.02	0.01
Porcupine Fish	*Dicotylichthys punctulatus*	B	Scarce	0.174	0.042	Rare	0.04	0.02
Blue-lined Goatfish	*Upeneichthys lineatus*	A	Rare	0.146	0.085	~	~	~
Silver Trevally	*Pseudocaranx dentex*	B	Rare	0.326	0.132	Rare	0.05	0.03
Eeltail Catfish	*Cnidoglanis macrocephalus*	A	Rare	0.118	0.065	~	~	~
Longfin Bannerfish	*Heniochus acuminatus*	B	Rare	0.063	0.020	~	~	~
Rabbitfish	*Siganus nebulosus*	B	Rare	0.306	0.203	Rare	0.03	0.03
Gloomy Octopus	*Octopus tetricus*	A	Rare	0.063	0.023	Rare	0.01	0.01
Amberjack	*Seriola dumerili*	C	Rare	0.153	0.095	Rare	0.11	0.04

Common Name	Species	Class	DIVER			BRUV		
			Residency	(UVC)	± se	Residency	(Max N)	± se
Leatherjacket (Six-spine)	*Meuschenia freycineti*	B	Rare	0.097	0.058	Scarce	0.16	0.04
Silver Sweep	*Scorpis lineolata*	B	Rare	0.139	0.087			
Mado	*Atypichthys strigatus*	A	Rare	0.056	0.026	Rare	0.01	0.01
Maori Wrasse	*Ophthalmolepis lineolatus*	A	Rare	0.056	0.024	~	~	~
Magpie Morwong	*Cheilodactylus gibbosus*	A	Rare	0.042	0.017	~	~	~
Bartail Goatfish	*Upeneus tragula*	A	Rare	0.174	0.127	Rare	0.01	0.01
Gobies	*Gobiidae sp.*	B	Rare	0.069	0.050	~	~	~
Tailor	*Plotosus lineatus*	A	Rare	2.083	1.194	~	~	~
Sand Whiting	*Sillago ciliata*	B	Rare	0.208	0.175	Rare	0.04	0.02
Luderick	*Girella tricuspidata*	B	Rare	0.069	0.047	~	~	~
Dusky Flathead	*Platycephalus fuscus*	A	Rare	0.042	0.026	~	~	~
Crested Morwong	*Cheilodactylus vestitus*	A	Rare	0.021	0.012	~	~	~
Pineapple Fish	*Cleidopus gloriamaris*	A	Rare	0.021	0.015	~	~	~
Old Wife	*Enoplosus armatus*	B	Rare	0.014	0.010	Rare	0.01	0.01
Ocellate Butterflyfish	*Parachaetodon ocellatus*	A	Rare	0.014	0.010	~	~	~
Red Rockcod	*Scorpaena cardinalis*	A	Rare	0.014	0.010	~	~	~
Stout Longtom	*Tylosurus gavialoides*	C	Rare	0.007	0.083	Rare	0.02	0.01
Giant Trevally	*Caranx ignobilis*	C	Rare	0.028	0.028	~	~	~
Scats	*Scatophagidae sp*	B	Rare	0.021	0.021	~	~	~
Longsnout Flounder	*Ammotretis rostratus*	A	Rare	0.007	0.007	~	~	~
Rays	*Dasyatidae sp.*	B	Rare	0.007	0.007	~	~	~
Leatherjacket (Mosaic)	*Eubalichthys mosaicus*	B	Rare	0.007	0.007	~	~	~
Ocean Jacket	*Nelusetta ayraudi*	C	Rare	0.007	0.007	~	~	~
Serpent Eel	*Ophisurus serpens*	A	Rare	0.007	0.007	~	~	~
Spotted Grubfish	*Parapercis ramsayi*	A	Rare	0.007	0.007	~	~	~
Batfishes	*Platax sp.*	A	Rare	0.007	0.007	~	~	~
Flattail Mullet	*Liza argentea*	B	~	~	~	Rare	0.01	0.01
Sea Mullet	*Mugil cephalus*	C	~	~	~	Rare	0.01	0.01

Appendix 2.   Mean Max N and percentage frequency results for species identified on artificial and naturally occurring reefs.

Common Name	Species Name	ARTIFICIAL		NATURAL	
		MAX (N)	% frequency	MAX (N)	% frequency
Striped Trumpeter	*Pelates sexlineatus*	21.195	76.4	7.126	61.6
Yellowfin Bream	*Acanthopagrus australis*	3.652	82.8	7.042	72.0
Snapper	*Pagrus auratus*	2.303	81.6	0.765	36.8
Tarwhine	*Rhabdosargus sarba*	2.243	70.8	1.622	44.0
Yellowtail Scad	*Trachurus novaezelandiae*	1.491	26.6	0.050	3.2
Striped Cardinalfish	*Apogon fasciatus*	0.566	11.2	0.017	1.6
Leatherjacket (Yellowfin)	*Meuschenia trachylepis*	0.431	25.5	0.252	12.8
Stripey	*Microcanthus strigatus*	0.266	15.0	0.050	4.0
Silver Biddy	*Gerres subfasciatus*	0.247	7.1	0.160	10.4
Leatherjacket (Fan-bellied)	*Monacanthus chinensis*	0.176	13.9	0.168	14.4
Leatherjacket (Six-spine)	*Meuschenia freycineti*	0.112	8.6	0.067	4.8
Silver Trevally	*Pseudocaranx dentex*	0.056	3.4	0.151	2.4
Greater Amberjack	*Seriola dumerili*	0.056	3.4	0.008	0.8
Three-bar Porcupinefish	*Dicotylichthys punctulatus*	0.045	4.5	0.017	1.6
Yellowtail Kingfish	*Seriola lalandi*	0.030	1.5	~	~
Sand Whiting	*Sillago ciliata*	0.026	1.5	0.017	0.8
Diamond Fish	*Monodactylus argenteus*	0.015	1.5	~	~
Black Spinefoot	*Siganus fuscescens*	0.015	0.4	0.008	0.8
Gloomy Octopus	*Octopus tetricus*	0.011	1.1	0.8	0.011
Stout Longtom	*Tylosurus gavialoides*	0.011	1.1	~	~
Sea Mullet	*Mugil cephalus*	0.007	0.7	0.185	6.4
Bartail Goatfish	*Upeneus tragula*	0.007	0.7	~	~

Appendix 2. *Continued*

Common Name	Species Name	ARTIFICIAL		NATURAL	
		MAX (N)	% frequency	MAX (N)	% frequency
Mado	*Atypichthys strigatus*	0.004	0.4	~	~
Old Wife	*Enoplosus armatus*	0.004	0.4	~	~
Green Moray	*Gymnothorax prasinus*	0.004	0.4	~	~
Flat-tail Mullet	*Liza argentea*	0.004	0.4	0.160	4.8
Serpent Eel	*Ophisurus serpens*	0.004	0.4	~	~
Dusky Flathead	*Platycephalus fuscus*	0.004	0.4	~	~
Large-toothed Flounder	*Pseudorhombus arsius*	~	~	0.008	0.8
Striped Scat	*Selenotoca multifasciata*	~	~	0.008	0.8

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