Using ecological evidence to refine approaches to deploying offshore artificial reefs for recreational fisheries

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ABSTRACT

Artificial reefs have many applications but are best known for their deployments to enhance fisheries, particularly recreational fisheries. Artificial reefs present opportunities to create new habitat in areas where natural reef is otherwise limited. The expectation is that assemblages of fish will take up residence on artificial reefs and that these assemblages will become at least similar, if not more diverse and abundant, to those on natural reefs. Although designed, purpose-built artificial reefs are becoming more widely used in support of recreational fisheries and many of the historic issues have been resolved, conservation practitioners and managers still face challenges as to the type, number and arrangement of structures and where to deploy them to maximize benefits and minimize risks. The ecological literature was reviewed to develop and enhance contemporary principles of artificial reef best practices for utilization including for goal setting and monitoring, selection of target species and determining optimal strategies for design, arrangements and siting.

INTRODUCTION

Artificial reefs have been deployed in many countries to prevent bottom trawling, to enhance recreational diving experience, surfing, for coastal defense purposes, for aquaculture, habitat restoration, or as a disposal option for hard waste (Grove et al. 1991, Collins et al. 1995, Baine 2001, Black and Mead 2009, Ajemien et al. 2015, da Silva et al. 2020). They are also well known for their deployments to enhance recreational fishing opportunities (e.g. Hueckel et al. 1989, Fabi et al. 2011, Smith et al. 2016, Recfishwest. 2017). In some jurisdictions, these deployments have been justified as a response to declining catch (Pauly and Chua 1988, Milon 1989) or a response to declarations of marine protected areas that have excluded recreational fishers from popular natural reef fishing spots (Fabi et al. 2015). In many countries, artificial reefs have become important elements of frameworks for integrated fisheries or coastal management (Baine 2001, Fabi et al. 2011, Moura et al. 2006, Ramos et al. 2007, Kim et al. 2008, Leitão et al. 2007, 2009, Tessier et al. 2015, Becker et al. 2017, Vivier et al. 2021) however Baine (2001) asserted that 50% of programs had not achieved their objectives, mainly due to poor design and planning (see also Pickering and Whitmarsh 1997, Jan et al. 2003, Campbell et al. 2011, Hackradt et al. 2011, Lima et al. 2019).

Wherever reefs are deployed to enhance recreational fisheries, they are expected to improve the fishing quality. Because recreational fisheries are social-ecological systems the product of recreational fishing is an experience with both catch and non-catch motivations present, although the relative importance of these factors is highly variable among participants and participant groups (McPhee 2008). In theory at least, artificial reefs can increase the catch rate and diversity of species caught and provide an easy to access location (Sutton and Bushnell 2007) which can enhance catch-related motivations leading to greater satisfaction with the recreational fishing experience. However, siting and design must also consider the possibility of crowding by anglers which can enhance the possibility of depleting stocks in addition to reducing satisfaction levels (Sutton and Bushnell 2007).

To meet catch expectations of recreational fishers it is key that artificial reefs provide new habitat for preferably a diversity and abundance of sought after fish in an otherwise saturated environment. However, The new habitat needs to provide production as much as attraction otherwise it could potentially make popular species more harvestable by aggregating them in a known location, thereby facilitating increased fishing mortality (Polovina 1989, Carr and Hixon 1997, Baine, 2001, Wilson et al. 2001, Powers et al. 2003, Claudet and Pelletier 2004, Szedlmayer and Bortone 2020). The problem could be exacerbated if new reefs attracted fishers who previously did not fish due to a prior lack of availability, skill and accessibility, thus increasing overall fishing effort within a management area (Pickering and Whitmarsh 1997). Another potentially adverse effect is increased predation on fish associated with artificial reefs that leads to an overall increase in natural mortality to some species (Leitao et al. 2008). It is feasible that this effect could potentially decrease recruitment to populations if predators and prey are attracted to artificial reefs, increasing vulnerability for the latter. The opposite is also possible (i.e. where predators are fewer on artificial reefs compared to natural reefs) as a result of the isolated nature of artificial reefs.

In support of the production hypothesis, some studies such as Brickhill et al. (2005), Karnauskas et al. (2017), Gallaway et al. (2018) and Szedlmayer and Bortone (2020) reported that artificial reefs may act as nursery areas for juvenile or sub-adults of some important commercially or recreationally harvested species (e.g. red snapper, *Lutjanus campechanus*), but until recently there have been very few studies that have indicated whether artificial reefs potentially increase the local biomass of benthic invertebrates and fishes (but see Powers et al. 2003, Shipp and Bortone 2009, Fowler and Booth 2012, Syc and Szedlmayer 2012, Smith et al. 2015, 2016, Folpp et al. 2020). The question is further complicated given that it suspected that density-dependent changes to demographic parameters regulating populations may occur for some species as a consequence of individuals becoming concentrated on artificial reefs (Lindberg et al. 2006, Mason et al. 2006).

Regardless of whether or not artificial reefs increase overall production of a defined area, marine organisms colonize them for shelter or foraging (e.g. Rilov and Benayahu 2002). Structures that add substrata that are physically, hydrologically and chemically different from natural habitats can in some circumstances be more advantageous to non-indigenous than native species (see review in Dafforn 2017). Further, if artificial reefs mimic suitable habitat cues associated with natural reefs, but fail to mimic their complexity, diversity or other vital characteristics, they may become ecological traps to native species (Komyakova and Swearer 2019, Komyakova et al. 2021), leading to lower fitness outcomes such as reduced growth, reproduction or increased mortality rates to those individuals recruiting or moving onto an artificial reef (Schlaepfer et al. 2002, Kristan 2003, Battin 2004, Hale and Swearer 2016). Recent studies have provided evidence that ecological traps may result from a proliferation of artificial structures (Hallier and Gaertner 2008, Jaquemet et al. 2011, Reubens et al. 2013, Komyakova and Swearer 2019, Komyakova et al. 2021, Swearer at al. 2021).

Although reefs purpose-built for fishing have been deployed in Japan and Korea for over 50 years (Kim et al. 2008), these types of structures have only recently become popular elsewhere. Advantages of deploying purpose-built artificial reefs include incorporation of design elements that promote durability, account for local hydrodynamics and substratum types, use non-toxic or non-corrosive materials, and to tailor design to desirable species. Such tailored designs often have a positive effect on catch rates (Seaman 2002, Kim et al. 2008); however the broader risks described above still need to be considered.

Iterative trial-and-error approaches to collecting knowledge about how to improve designs, (size, complexity and arrangement), understand ecological processes, and determine the optimal deployment location have hindered the rate of development of artificial reefs for recreational fishing. Controlled experiments to evaluate various options have also been rare (but see Lindberg et al. 2006, Mason et al. 2006) or have used structures much smaller than those that would be deployed to enhance fishing, and the scalability of such results is unclear. Notwithstanding this, contemporary guidelines for artificial reef deployments for recreational fisheries have used the plethora of available studies to develop best practice deployment frameworks based on the weight of evidence (e.g. see USDC NOAA 2007, Reef Ball Foundation 2008 in the USA; London Convention and Protocol UNEP 2009, Fabi et al. 2015 in Europe; Diplock 2010, Recfishwest 2017 in Australia). The purpose of this paper is to review some ecological concepts that are fundamental to these guidelines and to refine, where appropriate, key criteria for designing, siting and deploying artificial reefs for recreational fishing. We have focused on a subset of principles within contemporary guidelines that are important to maximize the success and minimize risks in artificial reef programs:

1. Definition of goals and quantitative measures of success that reflect desirable outcomes but minimize adverse impacts;
2. Prioritization and selection of target species, based on both desirability for fishers, ecological factors (including life history and predator-prey interactions), and vulnerability to excessive harvest;
3. Objective-specific and species-driven reef module design;
4. Optimization of module arrangements;
5. Siting of deployments to optimize enhancement and avoid unintended consequences; and
6. Adoption of best practice in operational monitoring.

(1) DEFINING QUANTITATIVE GOALS

Although most artificial reef guidelines recommend defining program objectives the specific objectives or goals of past reef deployments have not always been obvious, or were not always stated clearly (Baine 2001). Even if objectives and goals are not explicit, the general expectation for an artificial reef program designed to enhance recreational fishing is that it would provide additional fishing opportunities in terms of consistent or enhanced yield of one or more popular species, diversity of catch and accessibility and safety for recreational fishers. Given that other stakeholders may potentially use the area where a reef could be deployed for purposes other than recreational fishing (e.g. traditional foraging, commercial fishing or as shipping lanes), deployments must also consider the needs of other users (Tunca et al. 2014).

In a review of artificial reef programs, Becker et al. (2018) found that where goals had been given they were generally a qualitative measure, like increasing fishing yield, reducing illegal trawl activity, mitigating habitat losses or increasing fishing opportunities for recreational fishers. Although this would allow for some assessment of artificial reef performance, success or failure may be difficult to determine. Becker et al. (2018) proposed that artificial reef programs should include quantitative goals with specific indicators for determining success and clear definition around how these indicators should be measured. Indicators could include for example, meaningful and measurable targets for modelled regional production or CPUE, or economic performance as measured against a pre-reef baseline. Importantly, less successful or even damaging practices that become apparent, should be discontinued in future deployments.

In the same way that quantitative tools now exist to help guide the objectives of marine stock enhancement and assess release scenarios against objectives prior to large investments being made (Lorenzen 2008, Blount et al. 2017), similar tools could be developed for artificial reef programs. These may include modelling expected trophic relationships and production rates based on existing knowledge of species biology and trophic interactions (e.g. Campbell et al. 2011, Smith et al. 2016), density-dependent processes (e.g. Mason et al. 2006), modelling successional processes and also potentially modelling colonization or recruitment of desirable species to the new habitats based on knowledge of migration and source-sink dynamics.

Following on these principles, artificial reef deployment programs that are intended to enhance recreational fisheries, should as a minimum, include two fundamental considerations:

- A fit-for-purpose design and configuration that recognizes the requirements of recreational target species and their prey, while minimizing the potential for adverse impacts on the ecosystem; and
- Optimal siting that facilitates ease of access for recreational fishers but lowers the risk of undesirable outcomes to the environment and other stakeholders (see Site selection).

These should be supported by specific quantitative performance indicators, with robust monitoring and reporting arrangements (see Monitoring). Objectives should also link with broader management arrangements for the relevant fisheries to ensure that production rates and catches at the artificial reefs can be suitably incorporated into stock assessment.

(2) PRIORITIZING AND SELECTING TARGET SPECIES

Historically, the habitat requirements of local species were not explicitly considered in the design of artificial reefs (Baine 2001). Programs in Japan and Korea (Kim et al. 2008) represent key exceptions, and the advances made in these programs have supported the development of the purpose-built concrete or steel modules that are increasingly common throughout the world (e.g. Smith et al. 2015, Recfishwest 2017, Lemoine et al. 2019). Such tailored designs consider durability, account for local hydrodynamics and substratum types, use non-toxic or non-corrosive materials, but also incorporate design features that make them attractive for desirable species (Seaman 2002, Kim et al. 2008). Prioritizing and selecting desirable species is a key component of refining design characteristics to achieve program goals and consultation with the fishers is an important early step in this process. Recreational fishers often preferentially target and prioritize carnivorous species (Taylor and Suthers 2021), although desirable species often encompass a diverse assemblage that is dominated by reef species (Freire et al. 2020). Kim et al. (2008) and Bortone (2011) classified reef-associated fish, in the context of artificial reefs, into the following groups (see also Fig. 1):

> **Type A – Reside within or on the reef**: species with a very close connection to reef structures through physical contact (thigmotaxic) or visual excitation. They are generally more sedentary and reside on or within cavities in reefs (e.g. Muraenidae, Gobidae, Blennidae, Apogonidae and Scorpenidae).

> **Type B – Demersal reef associated**: reef fish that reside in close proximity to structures, and are closely linked to these structures due for provision of shelter and/or prey availability (e.g. Sparidae, Sciaenidae, Lethrinidae, Lutjanidae, Haemulidae, Epinephelidae, Serranidae and Labridae).

> **Type C – Pelagic transients**: species in the middle or upper water column that usually maintain a certain distance from the artificial structures, with association driven by sound excitation from the structures, prey aggregation or production, and current stream refuge (e.g. pelagic carnivores such as the Scombridae, Mugilidae and Carangidae).

> **Type D – Pelagic residents**: species in the middle or upper water column that usually maintain a certain distance from the artificial structures, but may link closely and semi-permanently to them because of shelter or food (e.g. pelagic planktivores such as Hemirhamphidae, small Carangidae, Atherinidae and Clupeidae, Acanthuridae, Kyphosidae and Scorpididae).

> **Type E – Reside on soft sediment**: found in, on or just above the adjacent soft sediment substratum, often supported by the halo of productivity that surrounds the reef structure (e.g. Bothidae, Platycephalidae and Mullidae).

Bortone (2011) and Kim et al (2008) considered that Type A species have the strongest production linkages to a reef but recreational fishers tend to target types B, C and E (and perhaps Type D to use as bait). The most abundant trophic group of fish around artificial reefs and natural reefs is often the Type D planktivores (Edgar and Stuart-Smith 2014, Truong et al. 2017, Becker et al. 2019). Given that planktivores are an important pathway linking lower trophic levels with exploited species (Champion et al. 2015), it is important for artificial reef designs to consider not only the habitat requirements of the targeted carnivores, but also these small pelagics which serve as their prey. Likewise, other smaller site-attached, non-target species (Type A) may be an important food source for carnivores, or play a role in maintaining a reef ecosystem; their habitat requirements also require some consideration. Consequently, we suggest the following basal considerations regarding species selection in artificial reef programs:

- Explicit identification of desirable species or species 'types' that are the primary targets of the deployment;
- Review of taxa-specific habitat requirements over a range of life history stages; and
- Incorporation of habitat requirements of target species and their prey into reef design features and arrangements.

(3) MODULE DESIGN and ENGINEERING

GENERAL CONSIDERATIONS: Niche requirements of some fish species may overlap, while other fish species (or life stages) may be highly specialized in terms of their habitat (Munday et al. 1997, Gardiner and Jones 2005, Wilson et al. 2008, Coker et al. 2014, Komyakova et al. 2018, 2019b). Variation in habitat associations among species has led to the hypothesis that purpose-built artificial reefs can be designed specifically to accommodate a particular species or a suite of species (Bell et al. 1989). This has been demonstrated through iterative design of modules in Japan and Korea (Kim et al. 2008). In Baine's (2001) review of artificial reefs, 36 papers (14%) noted the importance of complexity and configuration of artificial reefs, their size, volume and area to maximising abundance and diversity. The provision of shelter through refuges and crevices was highlighted as important in 6% of studies, particularly in relation to juvenile fish or shellfish. Important design elements for specific species included the amount of void space, bottom relief, height and shading. Other factors considered important to the success of a deployment included the type of material used for construction, structural integrity and stability. Similar factors were considered important in Kim's et al. (2008) review of artificial reef types in Japan and Korea. Other researchers have also considered that it is not only reef attributes that regulate reef fish abundance (e.g., density-dependent habitat selection) but also trophic interactions, and physiological performance (growth and condition) (Lindberg et al. 2002, Lindberg et al. 2006, Mason et al. 2006). Engineering for protection against toppling, scour and sliding depends on local conditions however these design requirements are outside of the scope of this review.

MATERIALS AND LIFESPAN: Early artificial reef deployments principally used materials of opportunity. Artisanal fisheries tended to use natural objects such as rocks and wood (Thierry 1988, Grove et al. 1991, Baine 2001, Fabi et al. 2011) and one of the earliest artificial reefs deployed for recreational fishers consisted of four vessels with other nondescript material (McGurrin et al. 1989). Since then, various materials have been used, such as purposeful and accidental ship wrecks, car and train wrecks, construction waste, metal, and plastic (Grove et al. 1991, Baine 2001, Fabi et al. 2011). Some of these, such as repurposed oil and gas platforms and potentially other abandoned subsea oil and gas infrastructure have been shown to be productive habitats (Claisse et al. 2014, 2015, Ajemian et al. 2015, McClean et al. 2017), while others, such as car tyre reefs, are identified as sources of marine pollution and contamination (Pollard 1989, Kerr 1992, Day et al. 1993, Collins and Jensen 1995, Collins et al. 1992, 1994, 1995, 2002, Wik and Dave 2009; Verschoor et al. 2016, Boucher and Friot 2017, Kole et al. 2017, Heery et al. 2017).

Purpose-built artificial reefs tend to be made of concrete, iron and steel, reinforced concrete (concrete and steel), ceramic, plastic, plastic concrete (concrete mixed with polyethylene, polypropylene sand and iron) and fibre-reinforced plastic (O’Leary et al. 2001). Concrete and steel modules have longevity of over 30 years (Recfishwest 2017, Fisheries WA 2010) and these materials are most commonly used, particularly the less toxic, high-strength marine-grade reinforced concrete (Baine 2001, Spieler et al. 2001). Concrete (via moulding) and steel have the required flexibility to tailor design attributes and to provide for more suitable surface textures for colonizing organisms, such as corals (see below). Welded steel is the preferred material for very large purpose-built artificial reef modules (Kim et al. 2008, Diplock 2010, Recfishwest 2017).

Note that different versions of concrete can leach undesirable substances or be less amenable to settlement of invertebrates than others (Becker et al. 2020), or, be more susceptible to colonization of non-indigenous species (Dafforn 2017). Some researchers have proposed potential ecoengineering approaches to these problems (Dafforn 2017). Materials used to construct artificial reefs are under continuous examination and evaluation by reef developers and environmental regulators (USDC NOAA 2007) particularly with respect to the types of concrete and its reinforcement material.

SIZE AND SURFACE AREA: The size of an artificial reef module imposes physical limits on the abundance of fishes that can be accommodated, while smaller reefs may be harder to detect by recruiting fish (Brown and Kodric-Brown 1977, Hale et al. 2015). Several companies are now patenting artificial reef modules of various sizes. In general, smaller, low-relief artificial reefs (e.g. ‘reef balls’) are often deployed in sheltered estuaries or bays (Folpp et al. 2011, 2020), whereas larger artificial reefs are generally deployed in offshore waters (Reeds 2017, Kim et al. 2008). Currently, the largest artificial reefs for recreational fishing are sunken ships, including those that were unintentionally sunk for recreational fishing (e.g. Lemoine et al. 2019), or decommissioned oil rigs (e.g. Ajemian et al. 2015). There are many repurposed rigs for fishing in the Gulf of Mexico (Ajemian et al. 2015), with similar options being investigated in the North and Adriatic seas (Løkkeborg et al. 2002, Sayer and Baine 2002, Fabi et al. 2004) and Australia (Fowler et al. 2014). Per unit area of seafloor, sunken ships or oil platforms are among the most productive artificial marine habitats, often exceeding natural habitats (Claisse et al. 2014, Lemoine et al. 2019) and Gallaway et al. (2019) reported some oil platforms to hold a diverse array of recreationally important species with total abundances in the order of tens of thousands. Although a high level of production for these larger structures is also related to their vertical extent (see below) we note that some recreationally important species may not always prefer larger reefs (see Lindberg et al. 2006).

Some of the largest purpose-built artificial reefs consist of high-relief, complex steel structures deployed in deep water, such as those designed to augment fish populations in Japan and Korea (Seaman Jr 2002, Kim et al. 2008, Ito 2011). The first ‘designed’ large steel artificial reef in Australia was deployed off the coast of Sydney, Australia, in 2011, and its success, popularity and productivity has paved the way for numerous multi-component reefs throughout Australia (Keller et al. 2016, 2017, Smith et al. 2016, Cardno 2018). In Japan and Korea, where there are hundreds of purpose-built artificial reef deployments and module types, the majority of steel reefs are less than 10 m tall; most concrete reefs are < 8 m tall (Kim et al. 2008).

The surface area of an artificial reef can be proportional to its size, but total surface area and bulk volume is important to productivity and diversity (Kim et al. 2008, Lemoine et al. 2019). Surface area available for the settlement of habitat forming epibiota is directly related to the abundance of food available for benthic feeding (Type A), which enhances productive capacity (London Convention and Protocol/UNEP 2009).

RUGOSITY AND VOID SPACE: Bohnsack and Sutherland (1985) suggested that complexity is an important consideration in the design of artificial reefs given that it promotes diversity of species and biomass. There have been many studies of fish associated with natural and artificial reefs supporting this hypothesis (Rilov and Benayahu 2000, Wilhelmsson et al. 2006, Wilson et al. 2007, Hackradt et al. 2011, Komyakova et al. 2013, Lemoine et al. 2019). Complexity can be considered in terms of external complexity, or ‘rugosity’, and in the case of artificial reefs, internal complexity, or ‘void space’.

Rugosity is the state of roughness or irregularity of a surface. Greater rugosity can provide direct cover for smaller (Type A and B) reef fish (e.g. Gratwicke and Speight 2005, Kuffner et al. 2007, Walker et al. 2009). Areas of great rugosity are also more suitable for attachment for algae and sessile invertebrates (Harlin and Lindbergh 1977, Hixon and Brostoff 1985, Mumby 2006). This is particularly so for horizontal surfaces where sessile invertebrates more easily attach to more elevated areas because they are less affected by accumulations or movement of sand along the substratum (Friedlander et al. 2003). Horizontal surfaces also provide diversity of habitat, having shaded and light-exposed surfaces. Even on vertical surfaces some sessile biota, such as coral larvae, appear to preferentially recruit to areas with greater rugosity (Rogers et al. 1984). Given that there are strong associations among some sessile communities, for example corals, and the diversity and structure of Type A or B reef fish communities (Komyakova et al. 2013), rugosity can therefore indirectly increase diversity and abundance of fish. Granneman and Steele (2015) showed that artificial reefs that had relatively low vertical relief and rugosities were structurally similar and had similar fish assemblages to the low-profile natural reefs in the region but artificial reefs with greater rugosities and relief than the natural reefs had fish assemblages that were approximately two- to five-fold more dense and had two- to three-fold more biomass. Similar associations between low vertical relief artificial reefs and low vertical reef natural reefs have been observed elsewhere (Komyakova et al. 2019a).

In terms of void space, many studies have found that reef blocks with greater area and more holes were characterized by greater species richness, abundances or biomasses of Type A or B fish than those blocks with less holes (Kellison and Sedberry 1998, Sherman et al. 2002, Lindberg et al. 2006, Hackradt et al. 2011). Holes on artificial reefs can also provide important habitat for invertebrates (Langhamer and Wilhelmsson 2009).

The optimal amount of void space is highly species-dependent (Bohnsack et al. 1991; Spieler et al. 2001). Small scale voids/holes may be relevant to small, site-attached Type A fishes, whereas large scale voids/holes may be suited to large fish species including sit and wait (Type B) species (e.g., large serranids). Large voids may be less desirable than smaller voids because they offer less shelter and less niches. Shulman (1984) and Hixon and Beets (1993) confirmed that the number and size of refuges significantly influenced the number, size, and species richness of Type A and B fishes. In addition to differences in habitat requirements among species relating to their size, many species also show ontogenetic shifts in habitat utilisation as they grow (Lindberg et al. 2006, Snover 2008, Wilson et al. 2008, Giffin et al. 2019, Komyakova et al. 2019b). Several studies have noted the importance of hole size relative to body size of Type A or B reef fishes as a means of predator exclusion (e.g. Hixon and Beets 1993, Almany 2004a, 2004b). Kellison and Sedberry (1998) considered that the smaller numbers of species and individuals on artificial reefs without holes might have been due to less juvenile and adult recruitment to those units. Indeed, some tropical studies have demonstrated that smaller-bodied individuals (e.g. recruits) tend to occupy coral with smaller branching space (Komyakova et al. 2018, 2019b).

In addition to size, the shape of the void and void position on a reef can also be very important, particularly for habitat specialists (Gardiner and Jones 2005, Lindberg et al. 2006). Kerry and Bellwood (2012) reported close association of all but one of the 11 families of large Type B reef fishes observed (including haemulids and lutjanids, along with lower counts of the serranids and mullids) with tabular corals relative to other coral forms, supporting similar findings by Shibuno et al. (2008). Given their canopy, it is intuitive that tabular corals should outperform both branching and massive corals in providing concealment or shade for large Type B reef fishes, but branching corals provide highly complex microhabitat, which is often utilized by smaller reef fishes or early ontogenetic stages of larger species for shelter. From the perspective of reef design, Kerry and Bellwood (2012) found artificial shelter units and tabular corals were functionally equivalent, supporting fish communities that were not significantly different, and with comparable occupancy rates for large Type B reef fishes. Notably, large Type B reef fishes preferred opaque rather than translucent canopies. Other research has shown that large fishes cued to tabular corals for concealment and/or shade (Almany 2004b). In contrast, smaller Type A fishes (e.g. pomacentrids, gobids, blennids and apogonids) were associated mainly with artificial reef units that did not visually obstruct their view. It was suggested that this is because smaller bodied species are more likely to be subjects of ambush predation (Almany 2004a, 2004b), and hence benefit from being able to see in every direction

VERTICAL RELIEF: Natural reefs that offer vertical relief are often characterized by greater taxonomic diversity of Types A-D fishes relative to their surroundings (Fagerstrom 1987) and there is ample evidence to suggest that if artificial reefs have sufficient vertical relief they too can support greater taxonomic diversity (Ogawa 1967, Molles 1978, Beets 1989, Bohnsack et al. 1994). Similar positive correlations between abundance and vertical relief have been demonstrated for artificial reefs (e.g. Thorne et al. 1989, Nakamura and Hamano 2009). Boswell et al. (2010) reported that large aggregations of fish underneath a decommissioned oil and gas platform were closely associated with the vertical slopes in the structure. Davis and Smith (2017) assessed proximity effects of small natural and artificial vertical walls on patterns of fish assemblages, testing whether wall size and type affected assemblages. Fish assemblages in the immediate vicinity of both natural and artificial walls had significantly more species and abundance of fish than those on surrounding, low-relief reefs. The size of the effect generated by walls was found to be proportional to the size of the wall, with species richness and abundance generally increasing with wall height and length. Differences between natural and artificial walls were detected, but these were confounded by differences in size between wall types. The study builds on previous work by showing that, within reefs, local areas of great species richness and abundance can occur in the vicinity of small but important reef features such as vertical walls, suggesting that walls appear to act as localized biodiversity ‘hotspots’.

Vertical relief also plays an important role in recruitment, at least for Type A - C coral reef fishes. Granneman and Steele (2015) showed that a difference in the size of fish on artificial and natural reefs was potentially driven by the enhancement of the recruitment of small, young fish to the higher relief and structurally more complex artificial reefs, coupled with the presence of older, bigger fish on natural reefs. Rilov and Benayahu (1998, 2000, 2002) tested the hypothesis that high-relief artificial reefs had more recruitment of coral reef fishes, mainly Type D planktivores, than near-bottom, low-relief artificial reefs. Recruitment was approximately two orders of magnitude greater for the experimental vertical installations than for the near-bottom ones. Most of the initial recruitment occurred at the upper sections of the vertical installations, which may indicate near surface movement of fish larvae as they approach the structure.

Some species also show post-recruitment differences in affinity for vertical structures. Red sanpper (*Lutjanus campechanus*), for example, recruit to high-relief vertical structure as age-2 fish in late summer and fall but prior to this age juveniles prefer low-relief habitats with shell or gravel substrata as do older fish s (Gallaway et al. 2009, Karnauskas et al. 2017).

UPWELLINGS AND VORTICES: Species preferences to different hydrological effects such as upwelling, eddies and slipstreams can enhance habitat, move nutrients and create feeding opportunities (Kim et al. 2008, Recfishwest 2017). Evidence is building that these effects are important drivers of abundance and diversity on artificial reefs in tropical and temperate environments, particularly for Type D planktivores. In their study of vertical relief on artificial reefs, Rilov and Benayahu (2002) suggested increased abundances of fish around the upper sections of the vertical installations may have resulted from preference by Type D planktivorous species for areas with greatest water / plankton flux. Zooplanktivorous fishes such as Yellowtail Scad (*Trachurus novaezelandiae*) position themselves around natural reefs relative to prevailing current conditions to maximize feeding opportunities (Hamner et al. 1988, Kingsford and MacDiarmid 1988), with similar locational preferences by this species also observed on a purpose-built artificial reef in south-eastern Australia (Becker et al. 2019).

Metal panels can also be incorporated into the design of steel reefs to take advantage of currents and tides to create upwelling that increases primary productivity (food sources for larval fish). Steel lattice-like structure added to steel reefs can also provide shelter and safe areas for baitfish (Type D) to congregate (Recfishwest 2017).

Optimal design criteria are summarized in Table 2.

(4) OPTIMIZING SPATIAL ARRANGEMENTS

MODULE NUMBER AND SPACING: Module arrangements can have an influence at the seascape scale on the effectiveness of the artificial reef. Individual artificial reef modules can be arranged within clusters to form multi-component reef ‘complexes’ or patch reefs that increase the effective footprint of the artificial reef system. Spatial complexity plays a prominent role in the ecological effectiveness of artificial reefs, and spatial configuration of the reef field has received considerable attention in recent decades to identify optimal characteristics in different contexts (Lindberg et al. 2002, Jordan et al. 2005, Lindberg et al. 2006, Mason et al. 2006, Biesinger et al. 2011, Campbell et al. 2011, Smith et al. 2017, Becker et al. 2019). Optimization can be complex and is necessarily context specific, requiring consideration of recruitment and colonisation processes, foraging behaviour of desirable species, connectivity and the expected recreational fishing effort. Decision makers usually need to balance multiple objectives, outcomes and impacts within a finite budget, and careful consideration of the spatial arrangement of the reef field is an important way to achieve an optimal outcome.

Determining the appropriate distances between artificial reefs and the number of modules primarily requires an understanding of how far Type B and D fish move away from modules to forage, the halo of productivity surrounding particular reef structures and the hydrodynamic environment that is desired within the reef field itself (as structures can locally dampen wave and current energy). An artificial reef is inhabited by predators and prey and all require shelter (either for ambush or safety) and need to forage. In short, shelter limits local densities of predators (e.g. Lindberg et al. 2006), foraging competition drives predators and prey away from shelter and predation risk drives prey toward shelter (Biesinger et al. 2011). The tradeoff between these two sets the population distributions for predators and prey around the reef.

Arrangements can also create interstitial zones between modules that in theory are safe pathways for fish to migrate between modules and are liveable space for some species (Jordan et al. 2005). Not all fish on an artificial reef obtain energy directly from biota living on the structure. Some Type B or D species will use the reef simply as a refuge and leave it to feed elsewhere (Coleman and Mobley 1984), whereas others, like the majority of Type D planktivores, are likely to source food around the structure (Becker et al. 2019). This has led to a better understanding of the optimal spacing among modules so that foraging areas would not overlap and fish would not be competing for food resources, particularly benthic food sources, and creating areas of intense prey depletion (‘foraging haloes’) around the reef structures (Lindberg et al. 1990, Frazer and Lindberg 1994, Campbell et al. 2011, Reeds et al. 2018).

While large steel purpose-built artificial reefs are generally deployed as solitary structures, smaller concrete modules are more usually deployed in clusters to create a sufficiently large reef footprint (Kim et al. 2008). The proximity between artificial reef units within reef clusters had been an important consideration for researchers, given the multitude of biological and ecological factors that affect how a cluster of reefs will function (Jordan 2005, Lindberg et al. 2006, Campbell et al. 2011). In creating a cluster, Kim et al. (2008) suggested that placing reef modules too close together can impact water flow in such a way that it adversely influences fish occupation, whereas Jordan et al. (2005) suggested that modules placed apart by a certain distance combined to function as a larger individual reef. Some researchers have developed more sophisticated approaches to determining spatial configurations and numbers of artificial reef units. The Korea Fisheries Resources Agency (FIRA) has been studying spatial configurations for many years, and Lan et al. (2004) developed a model that can optimize an arrangement by considering the costs, the budget and the deploying distance. As a general rule, optimal module spacing within a cluster should be 3-4 times base diameter of modules, as this both encourages fish occupancy, and supports fishing within and around the cluster, rather than simply on top of it (Cardno 2018). Modules of various types should also be arranged in such a way to achieve the complexity and niches within the overall reef that are required to support desirable species.

Clearly, the scale of an artificial reef cluster must be large enough to develop a stable assemblage structure and facilitate fishing activity simultaneously. Highly connected natural reefs can have a greater abundance and diversity of reef resident species (Vega Fernaˊndez et al. 2008). Large artificial reef mosaics may also accommodate more fishers who might use a reef simultaneously, and facilitate more diversity and abundance of fish (Jordan et al. 2005, dos Santos et al. 2010). In contrast, Campbell et al. (2011) showed that there are diminishing returns on abundance and biomass with very large increases in number of modules. In Korea, the area of clusters comprised of multiple modules varies but is generally between 100 – 1,000 m^2 (Fisheries WA 2010). Approximately 400 m^2 is an optimal footprint, given this is sufficient to incorporate at least four larger concrete modules, or many smaller modules.

MODULE CLUSTERS: Analyses by Biesinger et al. (2011), Becker et al. (2019), Gallaway et al. (2018, 2019) showed that fish abundance decreases rapidly with distance away from a reef-field; such close associations with artificial reefs have been shown to differ for Type C and D pelagic (Boswell et al. 2010, Scott et al. 2015) and Type A and B reef associated species (dos Santos et al. 2010). Further, Scott et al. (2015) found that a fish assemblage associated with an artificial reef is unlikely to be detected 30 m away from that reef. Biesinger et al. (2011) and Becker et al. (2019) suggested that such patterns indicate the value of areas near the reef-field as habitat for many observed fishes, highlighting a trade-off between foraging competition and the risk of predation, with fish more likely to forage in the area immediately around the reef-field in close proximity to the shelter provided by the modules. This holds true for reef (Type A), demersal (Type B) or pelagic (Type C or D) species (Truong et al. 2017). The patterns may also depend on other factors, such as the size of the artificial reef, the composition of the fish assemblage, the propensity of particular species to travel far from an artificial reef (possibly related to their ability to find their way back to those structures, density of prey or density-related competition for resources), and perhaps most importantly, the proximity of an artificial reef to other natural or artificial structures.

Optimally arranged artificial reef clusters would ideally take advantage of small-scale movements of fish while also limiting potential foraging overlap. Consequences of resource depletion caused by the overlap of foraging haloes are a reason why the deployment of artificial reefs should include consideration of how clusters are spaced. The resource mosaic hypothesis predicts (in part) that as reef spacing decreases, access to prey that inhabit the soft-bottom area around the reefs also decreases (Frazer and Lindberg 1994). Given some Type B or E species feed on non-reef-associated demersal prey they can create areas of intense prey depletion (‘foraging haloes’) around the reef structures, and benthic prey depletion has potential to increase as reef spacing decreases because of the greater overlap of foraging activity (Lindberg et al. 1990, Frazer and Lindberg 1994, Campbell et al. 2011). The feeding haloes may have negative effects on abundance, growth, and residence time of fish on artificial reefs if the fish are forced to forage outside of the halo area. For some species, more competition for food and a requirement to forage further afield may increase the risk of predation but some reef-associated fish tend to trade off this risk by limiting their forage range (Lindberg et al. 2006, Biesinger et al. 2011). Notwithstanding this, it appears intuitive that more widely spaced reefs should result in decreased halo overlap and leading to an increased density of potential prey species in surrounding soft-bottom habitat and therefore increased foraging opportunities.

Becker et al. (2019) found that a spacing of 50 m between clusters of modules created foraging grounds within the reef-field similar in size to those suggested by previous studies (Frazer and Lindberg 1994, Scott et al. 2015), while creating an increased refuge area for smaller Type A or B species (Champion et al. 2015). Fish occupy the spaces between the clusters both in the epi-benthic zone and the water column, indicating that reefs act as a single unit, and given that many researchers indicate that the total reef effect amounts to between 20-50 m (Fabi and Sala 2002, dos Santos et al. 2010, Scott et al. 2015, Smith et al. 2017), a 50 m spacing among clusters is likely to be appropriate for a well-connected reef-field. Becker et al. (2019) suggested that although it may be possible to further extend this spacing, this could reduce connectivity and risk the creation of isolated reef clusters. Scott et al. (2015) suggested a separation distance of 60 m would avoid overlapping distributions of associated fish, while still promoting a necessary level of connectivity. Given the findings highlighted here, optimal design arrangements are summarized in Table 2 and some examples for different sized modules are presented in Fig. 2.

(5) SITE SELECTION

GENERAL CONSIDERATIONS: The choice of optimal locations for deploying artificial reefs for recreational fishers is challenging for planners, given that they may need to consider the range of local conditions, alongside socioeconomic factors and legislative requirements. It would serve no purpose to deploy artificial reefs in areas that fish are known to actively avoid (e.g. areas where bottom water is anoxic or where there are other deterrents to fish), where natural recruitment is limited (unless seeding of the reefs is to occur), or, in the case of artificial reefs designed for fishing, where fishing is limited or locations are difficult to access (e.g. strong currents may prevent fishing, or a location may be distant to access points, or in a shipping lane). Other factors to consider include local habitat type, sediment type, protected habitats, current strength and direction or wave action, oceanographic parameters such as water temperature and depth, exclusion zones such as spoil grounds, port limits, marine protected areas, communications routes, proximity to culturally sensitive areas, and planning and permitting requirements (Pickering and Whitmarsh 1997, Baine 2001, USDC NOAA 2007, Fabi et al. 2015, Becker et al. 2018).

Separation or co-location of artificial reefs and natural reefs is a source of debate. Separation may create additional production to local natural reefs (rather than simply attracting fish away from natural reefs), however co-location may produce multiplicative impacts. It is thought that more isolated artificial reef will have a greater species diversity and be used by more Type C or D pelagic fish (Walsh 1985, Jordan et al. 2005, Vega Ferna´ndez et al. 2008), whereas highly connected artificial reefs will have more resident Type A or B reef species (Vega Ferna´ndez et al. 2008, dos Santos et al. 2010). Optimal distances for separation depend on the relative sizes of nearby natural reefs, the fish community structure (Brickhill et al. 2005, Kim et al. 2008, Komyakova et al. 2019a), the ability of fish to detect a reef, or foraging behaviour (Shulman 1985a, 1985b, Workman et al. 2002, dos Santos et al. 2010, Abecasis et al. 2013). Further complicating resolution of the production/attraction debate is the possibility that artificial reefs, were they to include a diverse range of species, function as fish habitat at temporally or spatially variable intermediate states between attraction and enhancement. Powers et al. (2003) estimated annual production enhancement (per 10 m^2 of artificial reef) under the various scenarios, and found it ranged from 0 kg under the attraction scenario, or a net decline with fishing, to 6.45 kg with no attraction, or 4.44 kg with fishing.

Artificial reef may well function as ecological stepping stones, or provide alternative foraging or shelter opportunities (e.g. Westmeyer et al. 2007, Lowry et al. 2017), thus increasing the connectivity between other non-reef habitats and the dispersion and recruitment of species (Westmeyer et al. 2007, Shipley and Cowan 2011, Keller et al. 2017). To date, the overwhelming majority of reef fish studies have been conducted at relatively small spatial scales, limiting our ability to identify these potentially important habitat linkages in a landscape context. However, many reef species (i.e. several lutjanids) exhibit multiple ontogenetic shifts in habitat use (e.g. Appeldoorn et al. 2003, Gallaway et al. 2009) while others (e.g. haemulids) may migrate daily to forage (e.g. Tulevech and Recksiek 1994). Grober-Dunsmore et al. 2007) found that the availability of seagrass habitat near natural reef patches appears beneficial for recruitment, settlement, survivorship, abundance and/or coexistence of certain juvenile reef fish at close distances but between 500 m and 1 km for adults.

Given the risks of artificial reefs attracting popular species from nearby natural reefs and thus increasing their vulnerability to fishing are yet to be disproven, we precautionarily recommend that managers deploy artificial reefs far enough away from natural reefs if they are focused on eliminating this potential risk. Under this approach, based on the likely species present and species-specific behaviours, proposed optimal separation distances of between 500 m to 1000 m (Brickhill et al. 2005, Kim et al. 2008, Topping and Szedlmayer 2011) would be adequate. Notwithstanding this, given the potential benefits of co-location of artificial reefs with natural reefs or other stepping-stone non-reef habitats is a focus area of current research we recognize that a shift in best practice is possible in the future.

WATER DEPTH: Few studies have explored the impact of water depth on the diversity and abundance of fish on artificial reefs. In Portugal, Santos et al (2013) showed there were slightly higher densities of fish recorded on deeper relative to shallow reefs, but other investigations focusing on particular species have been confounded by potential ontogenetic shifts in fish associated with habitat type. For example, in a study of red snapper (*Lutjanus campechanus*) in the Gulf of Mexico, there were significantly more small fish (<33 cm TL) at shallower depths (<35 m) and on small artificial reefs than at deep sites (>35 m) (Jaxion-Harm and Szedlmayer 2015). In Japan and Korea, Kim et al. (2008) reported that artificial reefs are chiefly installed in water depths of less than 40 m to favour the most habitable water depth for the majority of Type B target species and their Type A or C prey. Water depth may also influence the ability of some recreational fishers to effectively fish an area.

MULTICRITERIA ANALYSIS: With so many considerations to siting, a decision analysis tool is needed that can compare positive and negative effects or values against a list of relevant criteria to determine preferred areas or alignments. Multi-criteria analysis (MCA) is one approach that can integrate unquantifiable and intangible factors, such as expected impacts of an activity on marine benthic communities, with strictly measurable data (Mendoza and Macoun 1999, Herath and Prato 2006).

MCA can identify potential sites for artificial reef deployments within a broad study area and was used successfully to recently deploy purpose built artificial reefs for recreational fishing in the Northern Territory of Australia (Cardno 2018). An MCA requires the following steps:

1. Desktop review – to define the overall environmental and social characteristics of the region of interest
2. Identification of evaluation criteria – including environmental, social and engineering constraints and opportunities (Table 1)
3. Data review – identify available data to represent the evaluation criteria identified in Step 2. For each data set, the accuracy and currency of the data are evaluated.
4. Analysis
    a. Assign performance weightings
    b. Weighting of criteria
    c. GIS analysis.

Step 4 can be repeated to include stakeholder workshops to refine weightings of criteria. Indicative criteria and rationale used to identify potential artificial reef deployment areas are listed in Table 1.

Performance ratings for each criterion:

- Highly Constrained – highly constrained and unsuitable for further consideration (for example, in the proximity of an existing pipeline, at a wreck site)
- Moderately Constrained – characteristics that could restrict or are considered to represent an option that would require considerable additional investigation or justification
- Slightly Constrained – characteristics that while not restricting are considered less than ideal

- Least Constrained – characteristics that are in the opinion of specialists consulted pose no constraint.

MCA requires consideration of the relative importance (weighting) of each criterion compared with other criteria (see Stevens 1997) and the level of constraint for an area is assigned according to the sum of all weighted scores for criteria (see Cardno 2018 for more detail). Once one or more areas are identified, ground truthing and further stability analysis may also be required. In this stage it is important to verify that modules are designed to withstand the existing conditions of waves climate, current velocity, tides and extreme weather events such as cyclonic activity and 1 in 100-year storm events (Recfishwest 2017). Many of the above considerations are summarized in the optional site selection criteria proposed in Table 2

(6) MONITORING

In the past, there has been a general lack of monitoring to test effectiveness of, and evaluate risk associated with artificial reefs. Research and monitoring programs to assess artificial reefs against their goals will, however, become increasingly important. This is principally driven by growing environmental awareness and compliance with a 'social license' based on the expectation of rigorous evaluation. Demonstrating the performance of artificial reefs against quantitative goals is likely to support this social license into the future (Becker et al. 2018).

A monitoring program is integral to evaluate not only the assumptions made about the positive impacts of artificial reefs but also how negative impacts have been minimized, and in some instances in the event of an undesirable outcome, how this could be mitigated (or at least not repeated in the future). For example, if it became apparent that an artificial reef was attracting popular fish species and fishers so that there was a risk of an undesirable level of fishing mortality, then a bag or size limit could be implemented or adjusted. Monitoring should not be constrained to environmental indicators or catch but should be broad enough to consider socio-economic aspects of the artificial reef and its maintenance. In this paper we have focused on the environmental aspects of monitoring given sufficient guidance for other operational aspects of artificial reefs are provided for in artificial reef guidelines (e.g. USDC NOAA 2007, Fabi et al. 2015).

Reference sites will need to be incorporated into monitoring programs for environmental indicators to provide an essential context for observations on the artificial reef itself (Carr and Hixon 1997, Brickhill et al. 2005). While artificial reefs may not necessarily mimic the structure of natural reefs (Hueckel et al. 1989, Hackradt et al. 2011, Folpp et al. 2013), the inclusion of reference sites provides a broader picture of temporal process within the region of study and can assist interpretation of patterns.

Although some have advocated the use of MBACI (Multiple-Before-After-Control-Impact) as applicable sampling designs for fisheries projects because they have an environmental impact (albeit beneficial for fishers) (Kingsford 1999, Lincoln Smith et al. 2006), in reality, given the cost of artificial reef construction and deployment it is likely the overall number of artificial reef deployments with remain comparatively small. As such, artificial reef monitoring will inherently need to incorporate an asymmetrical sampling design (i.e. a single artificial reef sampling location and multiple control or reference locations). Such an asymmetrical design allows for comparison of variability of indicators within and among reference locations compared to those associated with the artificial reef. Notwithstanding, if multiple artificial reefs are deployed within a locality over time, it may be possible to use the same reference locations for each artificial reef and undertake meta-analysis of data for each new artificial reef (and the references) with the existing ones. Monitoring programs will also need to be aware of non-independence of samples, such as occurs where one sample in space or time influences another. For example, if artificial reefs are deployed very close together the fish may swim between them. If fish are sampled by net or line fishing at an artificial reef on one day, sampling the next day may be non-independent if many of the fish were removed on the previous day. Use of appropriate sample replication and avoidance of pseudoreplication are also very important. Where possible, for every type of module (or cluster) deployed and monitored there should be replicate modules providing a measure of among-module variability.

In summary, for environmental indicators we recommend monitoring against quantitative goals (for verifying benefits and undesirable outcomes) and in an asymmetrical design that includes sufficient reference sites.

DISCUSSION

Only a few decades ago, the opinion of fisheries managers suggested major concerns as to whether the desired "positive effects" of artificial reefs were possible (Murray 1994). At the time, *ad hoc* approaches to deployments, poor choice of material, design and site selection were significant points of contention because poorly designed reefs were still *in situ*. Unlike other tools used for fishery enhancement, such as aquaculture-based stock enhancement (where adaptive strategies can include, for example, adjusting the releases of fingerlings as new lessons are learned), in any artificial reef program there is generally only one shot at deployment at a given site. Given that retrieval to adjust design or to redeploy an artificial reef to a more suitable area is impractical or cost-prohibitive, science-backed planning is essential to maximize return on investment and minimize the chance of undesirable outcomes. For recreational fisheries it could be argued that it is ever more critical (than for commercial or artisanal fisheries) to get the balance right given perceptions are so important to the sector and given catch and effort tare harder to control than for other sectors.

Although some previous deployments have suffered from poor planning, there are several examples of good planning, and this has been improving over time. Guidelines for siting, development and construction of patented 'reef ball' technology in the United States have been in place for many years (Reef Ball Foundation 2008), and given the recent interest in deploying larger, purpose-built artificial reefs for recreational fisheries, there have been efforts to also develop general guidelines for these structures (USDC NOAA 2007, London Convention and Protocol/UNEP 2009, Diplock 2010, Fabi et al. 2015, Recfishwest 2017).

Contemporary deployments of artificial reefs commonly use designed, purpose-built structures, and positive outcomes have driven a resurgence of interest by fisheries managers and recreational fishers (Recfishwest 2017). Similar to aquaculture-based marine stock enhancement, artificial reefs also offers great opportunity to recreational fishers, but can come with considerable risks. The responsible approach to marine stock enhancement set a new standard for ensuring success, and avoiding poor decisions by embracing a logical and conscientious strategy for applying aquaculture technology to help conserve and expand natural resources (Blankenship and Leber (1995), Lorenzen et al. 2010, Lorenzen et al. 2014). Considering the ecological concepts that underpin best practice principles for artificial reefs (e.g. design, siting and deployment) that are outlined here, will ultimately support decisions that enhance recreational fishery outcomes and minimize risk.

We encourage programs to focus on developing goals that consider both catch and non-catch motivations given both are important to recreational fishers (Hunt et al. 2013, Arlinghaus et al., 2017, Wahyudin et al. 2018, Nieman et al., 2020, Solomon et al., 2020) and that have appropriate means for measuring success, selecting target species, and determining optimal strategies (in this case designs and arrangements). Additionally, we encourage an increased focus on the critical element of siting and determining optimal deployment locations, which can be aided by qualitative tools (such as MCA). This is especially important, as the neglect of 'composition, arrangement or location' increases the probability of a deployment failing to achieve desired outcomes (O'Leary et al. 2001).

We have suggested that there needs to be as much emphasis on setting goals for determining and analyzing failure, as there is on measuring success. This is particularly important given the risk of undesirable outcomes such as artificial reefs functioning as fish attraction, rather than fish production devices, and concentrating fishing effort on vulnerable species, facilitating colonization of non-indigenous species or becoming ecological traps. It is important for recreational fishery managers and fishers to be cognizant of, and responsible for, potential threats generated by their activity and how this underpins their social license to operate (SLO). The goals we propose provide the means for a mentality whereby we step away from reefs functioning as aggregation devices and head towards artificial reefs that also provide services not only for recreational fisheries, but for the ecosystem as a whole.

Choice of appropriate target species requires flexibility, given the global diversity in recreational fisheries and geographic variation in assemblages of fish but must be informed by a knowledge of which species are desirable to local recreational fishers. By designing artificial reefs to suit a variety of desirable species (and their prey), the fisheries enhancement element of reef communities may be more resilient to taxa-specific seasonal variation. However, to promote production, and sustainable exploitation, rather than attraction, the choice of species would best focus on reef-associated, demersal, philopatric (i.e. those that return to their place of origin to breed), territorial and obligatory reef species (Smith et al. 2016).

Generally, artificial reefs are designed considering engineering problems, such as durability (lifespan), stability (ability to withstand storms) and cost, whereas the suitability of the structure to target species is often a secondary consideration, if at all (Thierry 1988, Grove et al. 1991, Clark and Edwards 1999, Baine 2001, Fabi et al. 2011), even when the success of artificial reefs in providing suitable habitat for fish depends heavily on the design employed. It would seem logical that artificial reefs that can emulate local natural reef habitats, or improve on them, would have greater potential for production if they not only provided shelter for target species, but also a food sources by providing shelter for their prey (see Perkol-Finkel et al. 2006). The best purpose-built artificial reefs will require interdisciplinary collaboration between structural engineers, ecologists and fisheries managers. Although best practice may suggest decommissioning and module removal at the end of a reef’s proposed life is required, we have purposely not included specific advice on this. In practice, although small modules may be retrievable, large modules will unlikely not be retrievable given their integrity will most likely be compromised after 30 years. This and water depth will make retrieval costs unrealistic for most proponents. Given most artificial reefs will likely remain on the seabed once decommissioned it will be important to ensure that they are made from materials that once eroded, do not threaten marine ecosystems. Here it is worth noting the special case of oil rigs, which although not specifically designed to be fish habitat have become key hard bottom to some fisheries, such as for red snapper (*Lutjanus campechanus*) in the Gulf of Mexico (Shipp and Bortone 2009, Gallaway et al. 2019), although only 1,266 oil rigs, from a peak of about 4,000, remain in the Gulf of Mexico, due to removal of these structures on decommissioning. In many parts of the world there is now an emphasis on leaving some rigs *in situ* because of the known benefits to fisheries (e.g. Ajemian et al. 2015).

Although there will always be exceptions for some reef species (see Lindberg et al. 2006), bigger reefs generally hold more fish. Large, simple structures are poor fish attractants without some complexity of microhabitat (Kerry and Bellwood 2012). Optimizing shapes, vertical relief, void spaces and unit arrangement associated with a purpose-built artificial reef offer great opportunities for increasing volumes and diversity of catch to recreational fishers. Some compromises to design are likely to be required to ensure that artificial reefs are engineered sufficiently so that they do not move, topple or sink, and are built from suitable materials that promote longevity. Where there has been sufficient flexibility in the design a custom-designed artificial reef can be extremely productive and comparable to some of the most productive marine fish habitats (Smith et al 2016). In terms of the optimal arrangement, deploying more modules and using various types can increase the diversity and abundance of fishes. Reef modules should be arranged in clusters and given clusters have scalability the amount of deployments can cater for the expected amount of recreational fishers so as to avoid over-crowding.

Very much integrated with artificial reef design and configuration is the site selection and local environmental conditions. Selecting the appropriate locations to deploy artificial reefs is challenging given not all environments are conducive to increasing production and there is potential for competing use of some areas with other marine stakeholders. Consultation is an important part of site selection and success will be more likely when the demand is understood and the entire community (not just the recreational fishers) is committed to the chosen location, and is kept informed and involved during its selection process and its successes (Tunca et al. 2014). By inviting active communities' participation in the planning process, a program can deal effectively with the social and environmental challenges. The MCA tool presented here identifies the 'least constrained' areas, and has already been used effectively as a framework for the deployment of artificial reefs (Cardno 2018).

Measuring both the existing value and the impacts of any enhancement program is considerably challenging for a recreational fishery due to the diversity of motivational factors (Marta et al. 2001, Arlinghaus 2006, Young et al. 2016). Valuing the harvest caught by recreational fishers would considerably underestimate the value attributed to the activity by those fishers who are likely to fish for reasons independent of numbers or species caught. The potential to utilize market values of individual fish and harvests as an attempt to value catch by fishers is also problematic as many sport fish caught are released, though catch-release proportion is species-dependent (Tracey et al. 2013). Notwithstanding this, monitoring non-catch motivators in combination with quantitative indicators of the activity (e.g. catch rate and effort) and the fish assemblage will provide better understanding of success or failure of an artificial reef.

Marine stock enhancement and artificial reefs offer similar outcomes for recreational fishers, but both can also be potentially damaging to ecosystems if not properly executed. Even if abundance on an artificial reef were to increase, it does not necessarily confirm that biomass has also increased, or even been maintained, at a regional scale (Bohnsack et al. 1994, Powers et al. 2003) particularly if demographic parameters driving population dynamics (such as growth and reproduction) are compromised as has been shown for some species on experimental artificial reefs (i.e. Lindberg et al. 2006, Mason et al. 2006). Wilson et al. (2001) and Powers et al. (2003) suggest that both attraction and production are likely to interact in driving artificial-natural reef complexes and that much of the question relates to the role of larval supply and density-dependence that drive fish dynamics in general (Hixon 1998, Tupper and Hunte 1998). Osenburg et al. (2002) also considers that attraction and production are not mutually exclusive and can be considered as extremes along a gradient. While artificial reefs may simply attract and aggregate some species, they may promote the production of others and the situation is likely to lie between the two extremes (Powers et al. 2003, Bohnsack 1989 in Leitäo et al. 2008). If artificial reefs are integrated into a recreational fishery to become key pieces of habitat and fishing locations, such complicated effects will need to be incorporated into regional stock assessment models. This will be a key challenge to managers and scientists.

Just as the responsible approach to marine stock enhancement provided a conceptual framework that stimulated the evolution of aquaculture-based fisheries enhancement into a justifiable and complementary fisheries management tool, good guidance for artificial reef programs should have similar effect to recreational fisheries. Advocates of the responsible approach to stocking indicate that not all of the principles are relevant under all circumstances, but they urge proponents, where possible, to tackle all of the principles and to seek new processes for doing so (Lorenzen et al. 2010). We advocate a similar approach for managers of artificial reef programs for recreational fishing. While some of our advice around key criteria used to derive goals, select species and monitor effectiveness could probably be applied as they stand to any program, others require flexibility. Our advice for optimizing design and arrangement of modules is based on the best available information at the time of our review; as new information becomes available these concepts can be refined. We also acknowledge that all locations will have different constraints and stakeholders may weight categories in our site selection method differently, but our approach to siting is sufficiently flexible to account for such differences.

While this review has focused on artificial reefs for recreational fishing, clearly there are other user groups to consider. In fact, many of the studies cited in this review examined reefs that were deployed for the benefit of commercial fisheries, or commercial and recreational fisheries combined. In many cases, the target species are the same for each type of fishing, so in theory a reef could be designed that would be equally suited to each type. The concepts we propose are equally applicable, regardless of the beneficiaries (although management rules may differ). Examples include accessibility and access rights, size and catch limits and safety and potential duty of care considerations for recreational fishers. Artificial reefs could play a vital role in artisanal fisheries, but would be constrained by cost and access. Notwithstanding, artificial reefs may be a crucial means of supporting fisheries productivity in the face of climate change (i.e. damage to natural reefs from water temperature, acidification and storms of increasing intensity) and population growth. In this context, artificial reefs can play a very important role in future for all forms of fishing.

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Table

Table 1 - Criteria and rationale used to identify potential artificial reef deployment areas

Constraint	Criteria	Rationale
Environmental	Sensitive non-reef benthic habitat (seagrass, sponges, macroalgae)	Loss of existing sensitive benthic habitat is avoided
	Conservation estate	Impacts on sites with legal conservation status or areas identified as important to threatened species are avoided
	Existing hard strata and fish populations	Impacts to hard substratum habitats should be minimized
Social	Existing uses	Impacts to the existing use of the area are minimized
	Wrecks (including war graves)	Wrecks, including known war graves are avoided
	Cultural heritage sites	Cultural Heritage sites are avoided
	Mineral or petroleum exploration areas	Impact on mineral or petroleum exploration activities are minimized
Engineering	Substratum type	Areas of rock and limestone are avoided
	Distance from access point or harbour	Artificial reef is accessible
	Water depth	Artificial reef is not exposed during low tide or present a navigation hazard

Constraint	Criteria	Rationale
	Interference with existing infrastructure	Interference with marine infrastructure is avoided
	Interference with established shipping channels	Interference with established shipping routes is minimized

Table 2. Optimal criteria for design, arrangement and site selection

Optimal Designs

- High-strength marine-grade reinforced concrete or welded steel are the optimal materials for modules given their strength and longevity.
- Larger modules are more effective than smaller modules. However, a combination of smaller modules that form larger overall reef can be a viable alternative.
- Completely smooth surfaces should be avoided, a level of small scale structural complexity may increase invertebrate community formation, which may be of benefit to the fish community.
- Greater rugosity can provide cover for some fish, as well as minimising the effects of mobilised sediment on these biota.
- The height of modules should be dictated by reefs stability in the local environmental conditions, boat traffic safety and fish species requirements.
- The size of the effect (to abundance and diversity of fishes) generated by vertical walls (vertical relief) is proportional to the dimensions of the wall, with species richness and abundance generally increasing with wall height and length. Higher vertical relief has also been shown to stimulate rapid recruitment of juvenile fishes.
- Greater complexity in physical structures (at several spatial scales) through increased surface area, number of void spaces, cracks and crevices is commonly associated with a diversity of niches, high abundance and high species diversity.
- The shape of a void and its position on an artificial reef is important for shelter. Tabular voids provide concealment or shade to larger roving fishes. Smaller fishes also use such shelters but prefer that the shelters do not visually obstruct their view.
- Whilst maximising void volume to total volume ratio it is important to allow transparency to currents and stop the accumulation of silt.
- Features that produce upwellings, eddies and slipstreams are important drivers of abundance and diversity of fish, particularly planktivores.

Optimal Arrangements

- Using more than one module maximises complexity and increases the potential for greater diversity of fish
- Modules of various types should be arranged in clusters to maximise complexity at the scale of cluster.
- The closer the modules are placed together, the more they would function as a single unit. Spacing of modules within a cluster should be 3-4 x base diameter of modules to encourage fishing within the cluster
- An optimal footprint for a cluster is ~ 400 m^2
- Clusters have scalability, and clusters should be 50-60 m apart to provide for adequate foraging space for associated fish, and a necessary level of connectivity among clusters for foraging. This distance also provides drift channels between the reefs for fishing.
- Although there are some signs that deeper artificial reefs have higher densities of fish than shallow artificial reefs, it is likely that densities are driven mostly by individual species' depth preferences which can also include ontogenetic preferences.

Optimal Siting

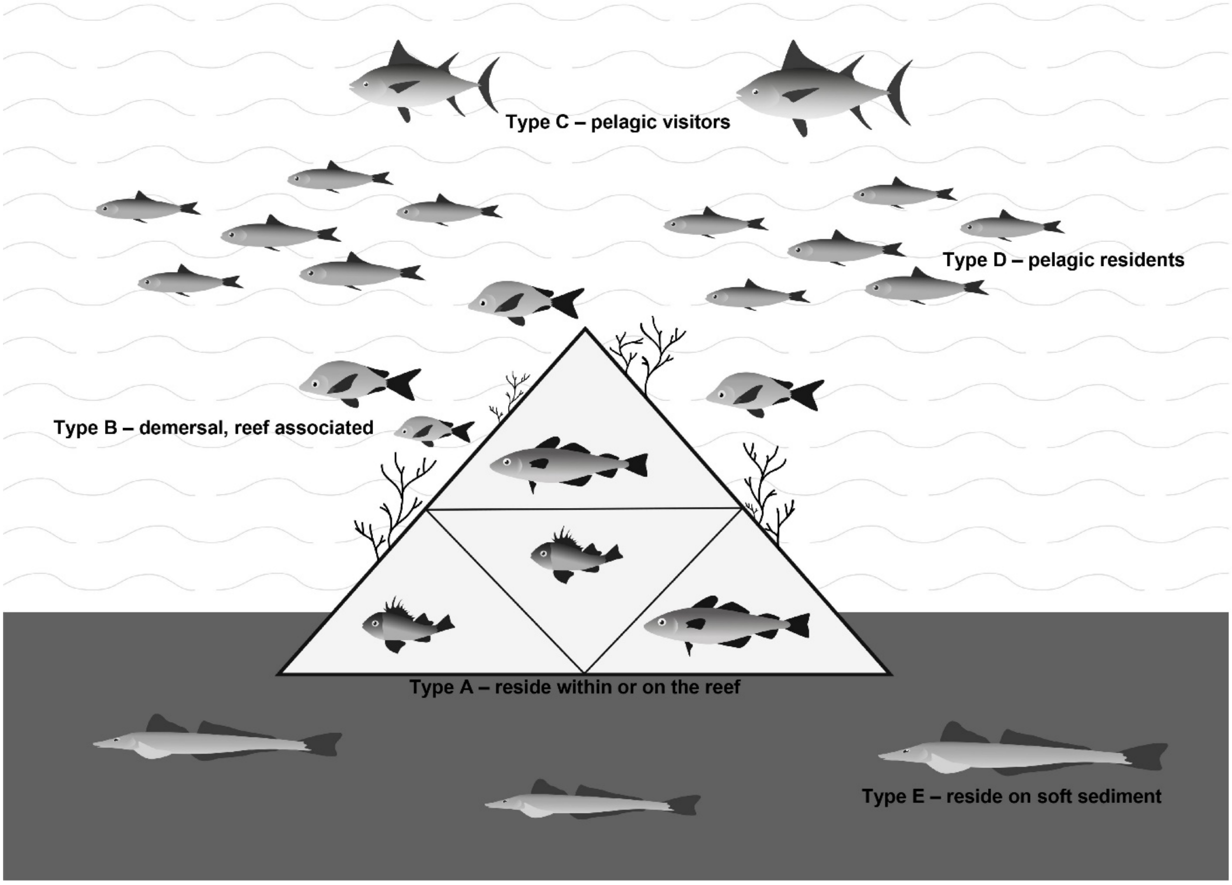
- Artificial reefs should be at least 500 m from natural reefs to avoid attracting fish.
- Environmental-
  - Avoid existing hard seabed
  - Avoid impacts to sensitive marine habitats
  - Avoid impacts to conservation estates
- Social-
  - Avoid impacts to existing users of the area
  - Avoid impacts to areas of cultural or historic heritage
  - Avoid impacts to mineral or petroleum exploration areas
- Engineering-
  - Avoid areas of rocky substratum of limestone
  - Avoid unstable seabeds
  - Accessible to recreational fishers
  - Artificial reef does not become exposed during low tides
  - Avoid interfering with marine infrastructure
  - Avoid interfering with shipping channels

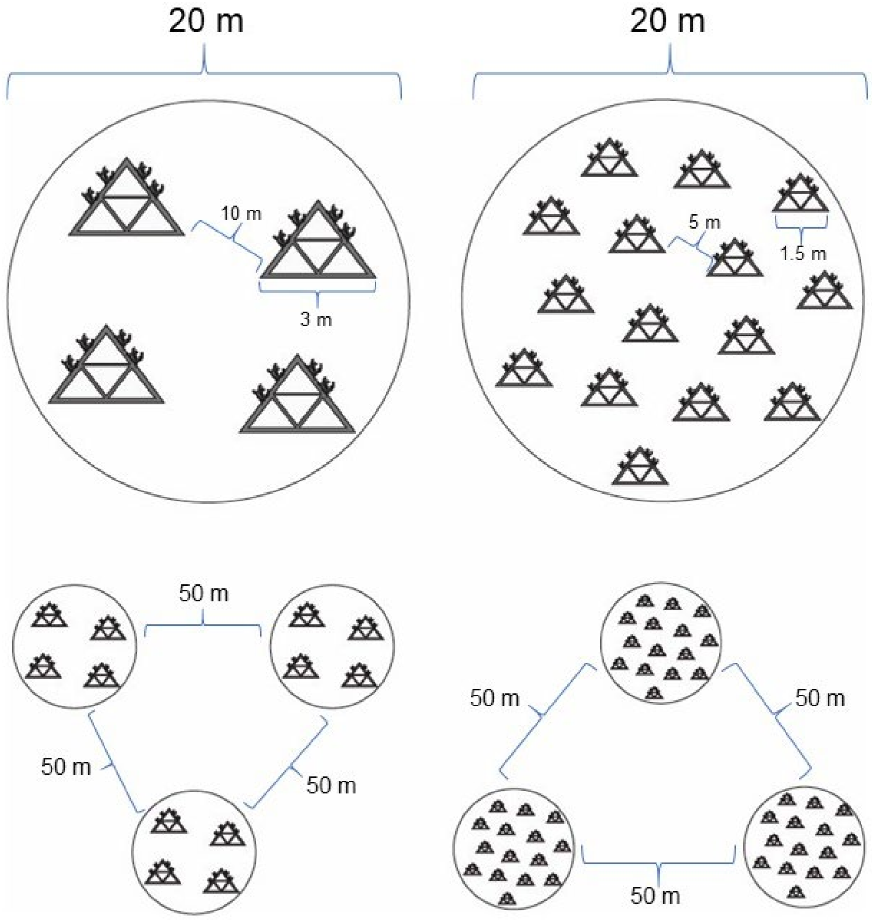
Figure Legends

Figure 1. Types of reef associated fish

Figure 2. Example optimal arrangements for artificial reef modules in a cluster where module base diameter = 3 m (left) or when base diameter = 1.5 m (left)

Figures


Type C – pelagic visitors
Type D – pelagic residents
Type B – demersal, reef associated
Type A – reside within or on the reef
Type E – reside on soft sediment


20 m
20 m
10 m
3 m
5 m
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