Original Article

Artificial reef structures and coral transplantation as potential tools for enhancing locally-managed inshore reefs: a case study from Wasini Island, Kenya

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Abstract
Many severely degraded reefs in the western Indian Ocean region show no signs of natural recovery and have remained for decades as barren, unconsolidated coral rubble fields with depleted commercially important fish groups. Consequently, several restoration techniques have been designed and developed to mitigate the localized impacts on coral reefs. Evaluating the efficacy of combined use of artificial reef structures and coral transplantation in enhancing habitat and recovery of key functions in severely degraded reefs is key to improved conservation of coral reefs. In this study, the survival of corals transplanted on reef structures is assessed, and changes in coral and fish abundance on artificial reef units and nearby natural reefs over time are compared. Coral cover on artificial reef structures increased from a mean of 17 % one year after initial attachment of fragments to 41 % after two years, with *Acropora* corals providing the highest cover. The artificial reef structures were also rapidly colonized by reef fish, with fish densities of 18±13 indiv./100 m^2 showing a three-fold increase compared to natural reefs after two years. Greater numbers of commercially important fish groups (e.g., Lutjanids and Acanthurids) were observed on artificial reefs while natural reefs harboured more small sized fish (Pomacentrids and Labrids). These findings provide insights for artificial reef projects that are capable of restoring the regenerative capacity of the human-induced coral rubble beds.

Keywords: coral transplantation, artificial reefs, community conservation areas (CCAs), reef restoration, reef conservation

Introduction
Coral reefs are among the most productive and biologically diverse ecosystems in the world (Burke *et al.*, 2011). While the value of coral reefs is intrinsic for many, there are also tangible physical and economic benefits especially for coastal communities including coastal protection, artisanal fisheries production, and tourism revenues (Burke *et al.*, 2011). The estimated value of Kenya's marine ecosystems is around US$ 2.5 billion per year (some 4 % of its GDP), of which 70 % is from tourism and reef-based fisheries, which are highly dependent on healthy reef ecosystems (Obura *et al.*, 2017a). Coastal tourism and subsistence fisheries are the two primary sources of livelihoods for coastal populations.

However, just like in many parts of the western Indian Ocean (WIO), Kenyan coral reefs have suffered from the severe impacts of human activities, resulting in long-term decline (Wilkinson, 2008; Obura *et al*., 2017b). These anthropogenic impacts include local stressors such as overfishing, land-based pollution, and global stressors such as climate change (Hoegh-Guldberg *et al.,* 2017; Mwaura *et al.,* 2017). Climate change-associated stressors, such as elevated seawater temperature and ocean acidification, are some of the global disturbances representing the greatest threat to coral reefs, over and above the many local threats (McClanahan *et al*., 2002; Hughes *et al*., 2018). In Kenya, previous inshore reef monitoring has shown that over 70 % reefs are in a poor condition (0-15 % live coral cover) and less than 5 % are in good condition (30-60 %) (Obura *et al.,* 2017b). The low status of live coral cover on most reefs is due to unusually higher ocean temperatures that cause stress to corals resulting to massive death (bleaching) of susceptible corals such as *Acropora* and other thermally sensitive and branching corals (McClanahan and Mangi, 2000; McClanahan *et al*., 2004). According to the Global Coral Reef Monitoring Network many coral reefs are in decline due to more frequent and severe bleaching events, forcing regime shifts to macroalgae dominated habitats (Shaver *et al*., 2020). In Kenya, large-scale coral bleaching events have been recorded in 1997/98, 2010, 2012, and in 2016, with many reefs experiencing very little or no natural recovery over time (Gudka *et al.,* 2018). The loss of coral cover following mass bleaching events can have a considerable impact on habitat complexity and associated fish populations over longer timescales (Wilson *et al*., 2006).

At the local level, loss of habitat caused by destructive fishing methods is one major threat to coral reefs (Burke *et al*., 2011). One of the most notable is the beach seine net fishing method, which destroys the structural complexity of the reef by shattering corals into pieces (McManus and Nanola 1997; Mangi and Roberts, 2006). The widespread use of destructive gears is also unsustainable as it not only harvests target fish species (i.e., commercially-important fish) but also non-target species (McManus *et al.,* 1997).

Several reef lagoons in Kenya including the Wasini Island shallow reefs have been affected by degradation. As indicated by a recent scientific report, these areas used to be dominated by fast growing branching corals such as *Acropora* and *Pocillopora* spp. (Karisa *et al*., 2020). But as a consequence of destructive fishing and large-scale bleaching of corals (*Acropora* spp.), most back-reefs are severely devastated and characterized by low hard coral cover and fish abundance (3 %, <10 individuals per 1000-m^2 area) , respectively (Mwaura and Murage, 2013). The absence of recovery is not due to a lack of larval availability; many reefs are generally well-connected and some outer reefs maintain a mixed coral community at over 40 % (Karisa *et al*., 2020). Some fragments usually survive after destruction, but after several days or weeks most are known to eventually die (Fox *et al*., 2003). Post-settlement mortality is usually high because of the mobility of loose rubbles that inhibits coral spat or juvenile attachment and growth (Fox, 2004). In such situations, the choice of restoration techniques and current state of the reef to be restored are fundamental considerations in reef conservation (De la Cruz *et al*., 2014).

Artificial reefs structures can provide additional, albeit unnatural habitat, and are increasingly being used to mitigate impacts on coral reefs (Fadli *et al*., 2012, Murage and Mwaura, 2015; Williams *et al.,* 2019). These artificial reefs which include man-made structures (e.g., shipwrecks, concrete structures, iron-rod structures), and sometimes with coral fragments attached, are intended to mimic natural reefs and enhance habitat availability for corals and reef-associated fish recovery (Abelson *et al*., 2006; Thanner *et al*., 2006). In severely degraded reefs, artificial reefs have been designed and developed to be used to rehabilitate their physical structure and function and consequently serve as a conservation tool (Williams *et al*., 2019). Specific conservation goals of artificial reefs include: restoration of 3-dimensional structures on degraded reef (Rinkevich, 2005), enhancement of commercially important fish (Fadli *et al*., 2012), and provision of firm substratum for coral transplantation and growth (De la Cruz, 2014). Successful reef rehabilitation using a combination of artificial reef structures and coral fragment transplantation have been undertaken with a view of enhancing coral and fish abundance on severely degraded reefs (Fadli *et al*., 2012; Williams *et al*., 2019). However, there is a paucity of information on their role as reef conservation or enhancement tools in severely degraded marine environments, especially in the WIO (Bostrom-Einarsson *et al*., 2020). To assess the efficacy of both artificial reef structures and transplantation of coral fragments as conservation tools, it is important to compare the key ecological changes or patterns between artificial and adjacent natural reefs.

An ecological justification for deployment of artificial reef structures onto which coral fragments are transplanted is that the area and condition of rubble-reef may limit reef fish abundance and firm substratum for coral recruitment (Fadli *et al*., 2012; Williams *et al*., 2019). In this study, it was hypothesized that the deployment of artificial reef structures in addition to coral transplantation will have a significant effect on the abundance of coral cover and reef fish, especially those targeted in the reef fishery, surpassing those on adjacent natural reefs.

The study objectives were therefore to 1) describe the technique for building the artificial reef structures, 2) document survivorship rate of transplanted corals, and 3) compare changes in coral cover and reef fish abundance on the artificial reef with the adjacent natural reef over time.

Materials and methods

Site description

The reef rehabilitation project was undertaken at Wasini Island, located on the southern coast of Kenya (Fig. 1). The rehabilitation work was initiated in response to funding by the GEF-Small-Grants Programme of United Nations Development Programme-Kenya, to support coral reef conservation in Wasini Island where a community-managed conservation area (CCA) had been set aside since 2010. The reefs around the Island support various tourism activities and are improtant artisanal fishing grounds.

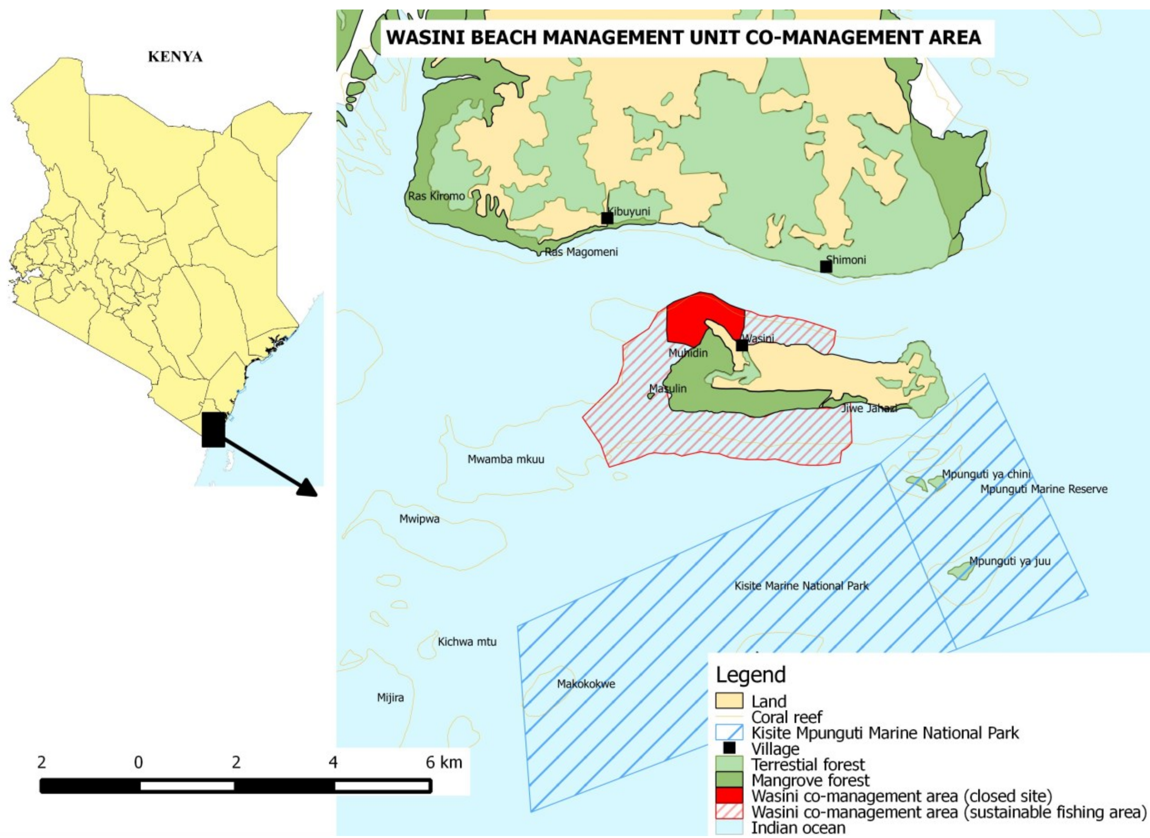

Figure 1. Map showing the location of the community conservation area at Wasini Island (no-take zone in red), southern coast of Kenya where the artificial reef project was undertaken in 2019 and 2021.

Community engagement in artificial reef work

The initiation of the artificial reef project started with a two-day workshop with key stakeholders and identification of their individual role/tasks in project implementation. The meetings also entailed awareness raising and training of stakeholders on basic coral biology and reef ecology, concepts of coral reef restoration, the activity objectives, transplanting techniques and the need for active restoration.

With the help of 30 local participants, the construction of artificial reefs was initiated by the making of rectangular wooden moulds, each with a dimension of 20x20x150 cm. A concrete mix was then made from three parts aggregate (predominantly coral boulders

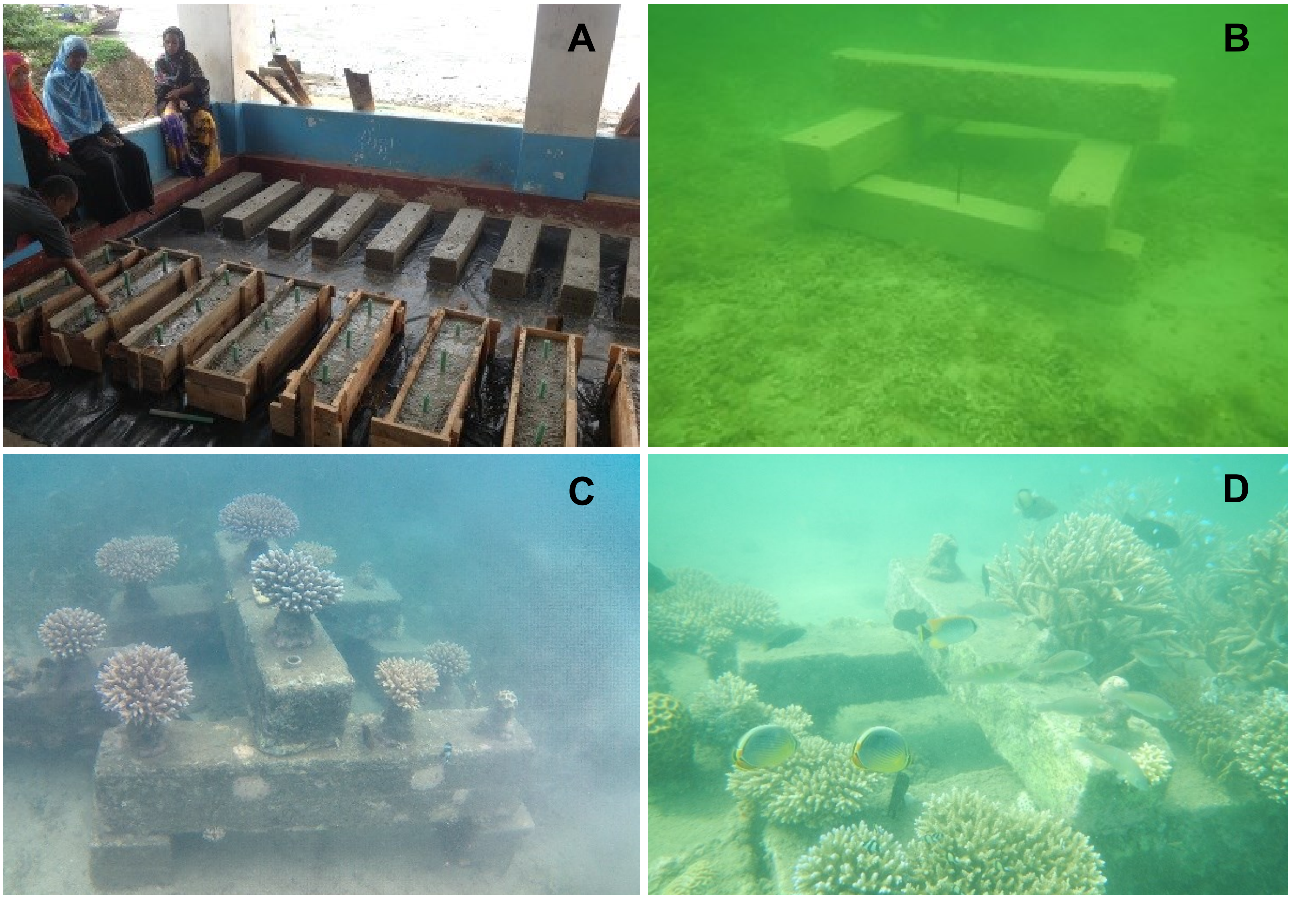

Figure 2. A - construction of artificial reef structures. B - deployed artificial reef structure units as replicates in the study. C- coral fragments attached to artificial structures after one year. D - abundant reef fish on artificial reef structures after two years.

crushed into particles, with a particle size of 2-20 cm) mixed with three buckets of sand and one bag of normal Portland cement. This concrete mix was then poured into the prepared wooden moulds to create a concrete block which was then reinforced using 8 mm steel bar (Fig. 2A). The constructed concrete blocks were then left on the beach to dry for 1-2 weeks. Divers then manoeuvred the blocks underwater and positioned them on bare-rubble habitat to form a pyramid-shaped reef structure (Fig. 2B). In this way, the artificial reef was made up of a network of over 20 groups of pyramid reefs deployed in similar depths within a no-take zone (Fig. 2B).

A few days prior to the transplantation activity, live coral fragments were collected by the authors assisted by about 20 trained community members. Fragments were sourced from a healthy reef on the northern side of Wasini Island (Fig. 1). The donor site was chosen on the basis that it had abundant and suitable branching coral species such as *Acropora* spp., *Porites*, and *Stylophora* which were targeted for use in this restoration, although other genera were also included. Fragments were augmented by loose coral fragments ("coral of opportunity") collected from the back reef at the rehabilitation site as they would otherwise perish from being buried in soft sediments or swept about by currents. Upon removal from the source reef, the harvested fragments were kept in plastic buckets filled with sea water and immediately transported by boat and laid down (≈2m deep) next to the artificial reef site. With the help of 20 local participants, more than 800 coral fragments were transplanted onto the concrete artificial reef surfaces. The transplantation was performed by attaching the coral fragment to the artificial reef surface using cement-sand mixed with seawater (i.e., cement balls). The fragments were placed 20-30 cm apart to avoid space competition among them as detailed by Omori and Iwao (2014). Periodic maintenance of transplanted corals was also carried out by local community participants for a period of 3 months, which involved cleaning/scrubbing of the concrete base of fragments, replacement of dead fragments and re-securing the dislodged ones.

Study design and data collection

This restoration project is a community-led project and the artificial reef was not intended as an experimental study. As such, this study design was superimposed onto the existing artificial reef to meet the objectives of this study.

The artificial reef network comprised over 10 groups of artificial reefs; only three of these artificial reef aggregations were selected as replicates for this study. To investigate the influence of artificial reefs, the changes in fish abundance and coral cover were monitored over time and compared to those in adjacent natural reefs. The monitoring also involved periodic observation on survivorship of transplanted corals which were tagged or labeled in order to track them. Survivorship of coral species was rated by number of corals that were alive compared to the total transplanted.

Changes in coral cover and fish abundance were assessed over three years, once before and twice after artificial reef structure deployment (i.e., before deployment, after 1 year, and after 2 years). As the artificial reef structures were deployed on sand-rubble habitat, the 'before' samples represented data prior to the deployment of the artificial reef structures. The percentage benthic cover within each of the three replicate artificial reef units was estimated using the 10 m line intercept method (PIT), following the protocol described in English *et al.* (1997). The observer recorded the benthic cover type under the tape at 0.5 m intervals.

The response of the fish community to the artificial reef treatment was assessed following modification of the line transect (Samoilys and Carlos, 2000). Each of the three artificial reef units was considered as a plot. The diver and transect layer swam along the three transects in one plot counting the fishes within a 5 m wide belt and 20 m long transect. The fish species counted at family level were later assigned into either "indicator" species (e.g. Chaetodontidae) and "target group" (e.g., Lutjanidae) and "other" families. The authors performed all the monitoring of the parameters. To compare the changes of benthic cover and fish abundance on the artificial reefs to those on adjacent natural reefs, three representative natural reefs were also monitored following the sampling protocol for the artificial reefs.

Statistical analysis

Statistical analyses were performed in R version 4.0.05 (R Core Team, 2021). One-way ANOVA was used to test for difference in mean coral cover and fish density before deployment, 1 year and 2 years after deployment, and on the natural reef substratum. Raw values of coral cover were used after examination of the residuals and revealed no major bias. Tukey's post-hoc tests were used to determine which treatments differed.

Fish family density was used to test the degree of change between the two reef types, i.e. natural and artificial reefs. Therefore, variations in fish family community structures was compared between natural and artificial reefs as treatment factors. A sample-family density matrix was developed with a sample size of six (6) for both natural and restored reefs. This was followed by a square root transformation to reduce species density variation within the dataset. Bray-Curtis similarity was used in the multivariate Permutation Analysis of Variance (PERMANOVA) to test for significance of differences between the two treatments. Permutation of Dispersion (PERMDISP) was used to test the degree of sample point variation in multivariate space, which were visualized in a non-metric Multi-Dimensional Scaling (nMDS) graph. Finally, Similarity Percentage (SIMPER) analysis was used to draw an understanding of the fish families responsible for the variation in community structure.

Results

Transplanted coral survival rate

After one year the overall survival rates of the coral transplants ranged between 30-100 % for the 15 genera with a high average survivorship of 76 % (Table 1). There was strong variation between genera, with higher mortalities being recorded in corals such as *Pocillopora*, G*oniopora*, and *Echinopora*. On the other hand *Acropora*, *Stylophora*, *Porites* massive and *Porites* branching exhibited higher survivorship (89-100 %). Generally, six months after transplantation, 86 % of the transplants survived well. One important observation was that it was critical to regularly clean the fragment base to avoid algal overgrowth on transplanted corals in the first three months after deployment. A few transplant mortalities were evident during the initial months and may be attributed to dislodgement from the concrete substrate due to poor cementing and accidental knocks/detachment by community members during cleaning, rather than natural mortality.

Coral cover

Mean percentage coral cover on artificial reef structures significantly increased from a mean of 3.3 %±SD 5.6 %, to 16.5 %±SD18.3 % one year after initial attachment of coral fragments to 41 %± SD20 % after 2 years

Table 1. Percentage survival rates of transplanted coral fragments.

Coral genus	Initial number of transplants	Live transplants observed	Survival rate (%)
Acropora	80	78	97.5
Favia	37	30	81.1
Favites	43	35	81.4
Hydnophora	25	18	72.0
Goniopora	5	2	40.0
Echinopora	34	17	50.0
Stylophora	7	7	100.0
Diploastrea	8	6	75.0
Platygyra	34	28	82.4
Pocillopora	50	15	30.0
Pavona	17	15	88.2
Lobophylia	10	8	80.0
Porites (massive)	46	43	93.5
Porites (branching)	38	34	89.5
Leptoria	19	14	73.7
Overall	453	350	75.6

(P<0.001; Fig. 3). In contrast, coral cover on the nearby natural reef remained low (4 % to 8 %), indicating no evidence of substantial change over the study period (Fig. 3). Coral cover of 41 %± SD20 % after two years of artificial reef deployment (Fig. 3) was almost entirely because of the increase in the cover of transplanted *Acropora* corals (Fig. 2C).

Fish abundance

The artificial reefs were quickly colonized by damsel and other small-bodied fish (see Fig. 2D). Pairwise differences revealed no evidence of differences in fish density both on artificial and natural reefs before deployment (Fig. 4). However, there were significant differences in fish density, with reef fish density increasing from a mean of 7± SD6.7 indiv./100 m^2 after 1 year and to 18± SD12.9 individuals/100 m^2 after 2 years on artificial reefs (Fig. 4). In contrast, there was no evidence of changes in fish abundance at the natural reefs over time.

Non-metric Multidimensional Dimensional Scaling (nMDS) showed a separation of sample points between natural and artificial reefs (Fig. 6). This was supported by PERMANOVA (Table 2) and PERMDISP, which both showed a significant difference between the two reef types (P<0.005). While the former demonstrated a variation in fish family community structure, the latter demonstrated a significant variation in the dispersal of sample points in multivariate space (Fig. 6). SIMPER results showed 57 % dissimilarity between natural and transformed reefs with Acanthurids (16 %),

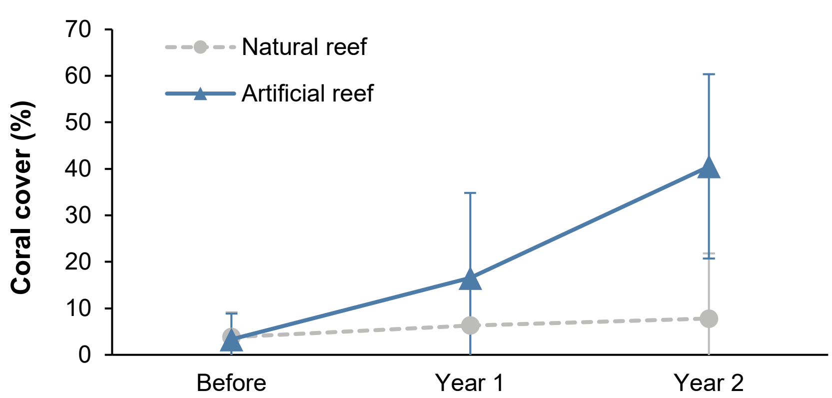

Figure 3. Percentage coral cover (m±stdev). Time steps are before artificial reef structure deployment, 1 year after and 2 years after artificial reef structure deployment. Dashed line with circles represent the natural reef and solid line with triangles represent the artificial reef.

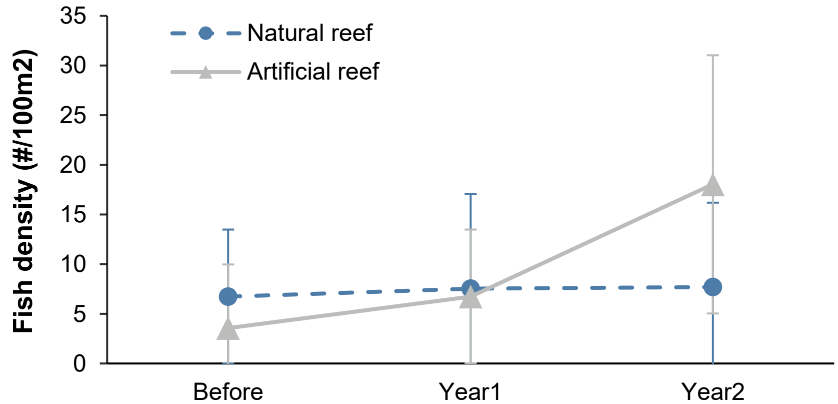

Figure 4. Average fish density (m+stdev). Time steps are before artificial reef structure deployment, 1 year and 2 years after artificial reef structure deployment. Dashed line with circles represent the natural reef, and solid line with trianges represent the artificial reef.

Pomacentrids (14 %) and Labrids (10 %) accounting for most of the variation; all three being more abundant on the natural reef (Table 3). However, commercially important families such as Scarids, Lutjanids, Serranids, Lethrinids, Haemulids, Siganids and Chaetodontids were all more prevalent on the artificial reef compared to the natural reef (Fig. 5).Two year-old artificial reefs had well-developed corals to the point where they had begun to attract reef health indicator fish species (i.e., Chaetodontids) (Fig. 5).

Discussion

This study reveals the potential of the combined application of artificial reef structures and coral transplantation as conservation tools in speeding up habitat restoration and recovery of key functions in a severely degraded reef system. It was found that before the deployment of artificial reef structures, the density of fish was similar to that at nearby natural reef. One year after the deployment and transplantation of coral fragments, fish abundance was significantly higher than those on natural reefs. Throughout the study, fish abundance on natural reef either slightly increased or showed no evidence of change over time. When viewed in conjunction with other research in the literature that showed higher fish density following the deployment of artificial reef structures (Fadli *et al*., 2012), this study provides compelling evidence that artificial reef structures create new reef habitat, that provides a potential basis that favours foraging opportunities and increases shelter availability for fish, (Raymundo *et al*., 2007; Charbonnel *et al*., 2002). Besides the effect of new reef habitat, differences in reef heterogeneity and increased niche partitioning could also explain the higher density on artificial reefs compared to the natural reef. In fact, even if the habitat complexity has not been quantified in this study, it should be noted that natural reefs is

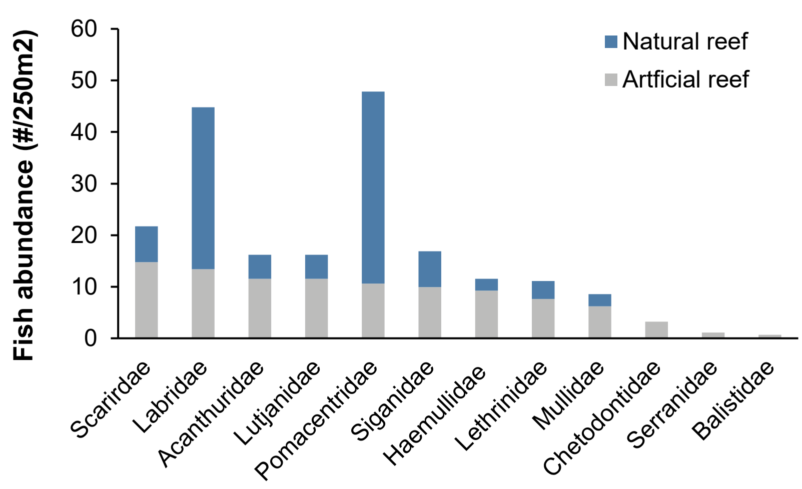

Figure 5. Proportion of fish abundance by Family two years after artificial reef deployment, and on natural reef.

Table 2. One-way PERMANOVA results comparing the community structure of fish families between natural and artificial reefs.

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Treatment	1	2682.2	2682.2	3.6618	0.004	411
Residuals	10	7324.7	732.47			
Total	11	10007				

a extensively rubble-dominated field with some few rocky boulders, whereas rehabilitated area is small (i.e., 0.012 ha) and comprises several deployed discontinuous artificial reef subsets. Coral reefs are complex biogenic habitats, combining both physical (e.g., high structural heterogeneity) and biological (e.g., live coral cover) characteristics (Walter and Haynes, 2006; Feary *et al*., 2007). Several fishes are dependent on live coral for food (Pratchett, 2005) and structural heterogeneity increases available habitat; providing refuge from predation and attracting fish recruits (Lindahl *et al*., 2001). As the current study showed, this effect is especially important when both the structural and biological component potentially influence the abundance and composition of the associated fish community on artificial reefs. The difference is perhaps linked to the lower coral cover in natural reefs given that coral dependent fish such as Chaetodontids require higher coral cover (Boström-Einarsson *et al.,* 2018).

Few studies have investigated the effects of adding new habitat using before and after reef construction samples. In the current study before and after reef construction was used to investigate the effects of artificial reefs on fish abundance. The study hypothesis proposed that if the addition of new reef habitat on sandy-rubble reef serves to overcome habitat limitation, then addition of corals on constructed artificial reef structures would promote greater fish abundance. Indeed, the density of fish after one year on artificial reef was not considerably different to before the deployment, but was significantly higher after two years, and higher than on natural reefs. It should be noted that the carrying capacity of these artificial reefs may be is bottlenecked by the lack of large corals which are so crucial for habitat provision in natural reefs (Holbrook *et al*., 2002). The artificial reefs were purposely deployed to optimize habitat complexity and rugosity but clearly cannot compare to healthy natural reefs in terms of habitat provision for fish. However, increased habitat area provided by artificial reef does show the effectiveness of these structures in their ability to facilitate survivorship and increased growth of transplanted corals. Where coral mortality is low, it would be expected that these corals would grow to larger sizes, increase habitat complexity and support larger fish populations (Halford *et al*., 2004). In this study, the artificial reefs were rapidly colonized after one to two years by numerous reef fish including those important in the fishery such as the Lutjanids, Acanthurids as well as ecological indicators such as Chaeotodontids. The exceptional increases in abundance for both indicator and target fish groups on artificial reefs suggest that these might have important positive implications for the ecological status of coral reefs and the livelihoods of the coastal fishing communities (Cabaitan *et al*., 2008; Spalding, 2016).

The probable key to success in rapidly creating new reef habitat in this study was the choice of a fast-growing coral species (*Acropora* spp.) that was resistant to handling, easy to transplant, survived well and grew

Table 3. SIMPER results showing the comparison in average abundance of fish families between the natural and artificial reefs.

Families	Natural Reef Av.Abund	Artificial Reef Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Acanthuridae	22.17	9.5	12.42	2.1	20.8	20.8
Pomacentridae	27.17	15	11.15	1.14	18.68	39.48
Labridae	10.67	17	7.13	1.6	11.94	51.42
Scaridae	9	14.17	5.45	1.28	9.12	60.54
Lutjanidae	7.83	10	5.11	1.8	8.57	69.11
Serranidae	0	6.17	4.25	0.73	7.12	76.23
Lethrinidae	2.5	11	4.07	0.87	6.82	83.05
Haemulidae	0	8.83	3.53	1.33	5.91	88.96
Siganidae	5.17	9.33	3.04	1.4	5.09	94.05

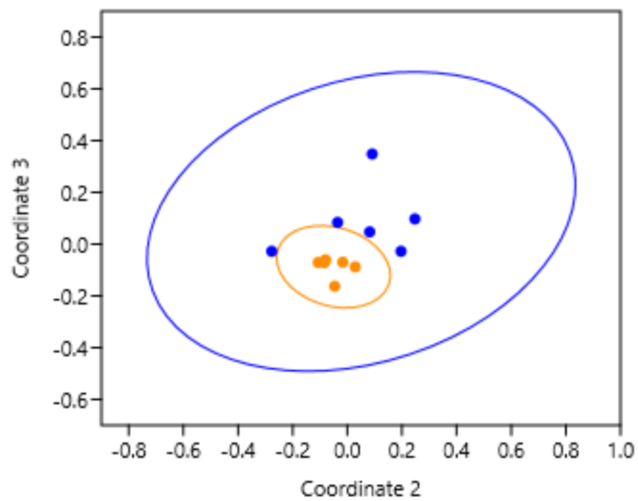

Figure 6. nMDS plot demonstrating variation in fish community structure among sample points where brown and blue dots represent natural and artificial reef samples, respectively.

rapidly. There was a clear pattern in terms of the coral cover of coral communities on the artificial reef over time. The cover of *Acropora* corals increased over time from deployment, perhaps due to the high survival of transplanted corals and also the artificial reef substratum being rapidly colonized by coral recruits (Rinkevich, 2000, pers obs.). One explanation for this is that *Acropora* corals have been identified as an opportunistic genus with life history traits that allow for the quick colonization of newer substrates (Jouval *et al.*, 2020). In this study, coral cover and habitat complexity may have improved to attract more reef fish within a relatively short time period as the project used mainly fast-growing *Acropora* coral species (Rinkevich, 2000).

The relatively high survival of transplanted corals could also be attributed to community participation in maintenance efforts (i.e., once a week for three months) on transplanted corals. Proper training and education of participating community members was important in order to maintain the structural integrity of artificial structures and transplanted corals to reduce impacts from potential competing taxa (i.e., macroalgae, sponges). A similar study has shown that higher survival of coral transplants is mostly related to the avoidance of adverse conditions, including algal overgrowth, by maintenance cleaning (Hernández-Delgado *et al.*, 2014).

Anecdotal reports suggest an immediate benefit of involving the local community in supporting artificial reef projects. The use of artificial reefs as a method to increase reef habitat as part of ecological restoration is a valid application, particularly in areas with tourists. Community members have increasingly been showcasing their restoration sites to tourists, thus providing an additional benefit that could develop into an alternative livelihood for local residents as indicated in another study (Cadiz and Calumpong, 2000). On average, there has been an 80-100 % increase in weekly income for the Wasini community members, from US 60 to US 220, during high tourism seasons (unpublished data).

In conclusion, the results from this study provide compelling evidence that the use of artificial reef structures in conjunction with coral transplantation represent viable restoration tools as they have the potential to restore habitat and enhance coral and fish abundance on severely degraded reefs. Additionally, this project demonstrates that local communities can be practically involved in restoration of their degraded reefs (e.g., regular cleaning of algal overgrowth on coral fragments) when provided with training on restoration skills, as it encourages their active participation and stewardship in reef restoration (as also observed in related studies, e.g., De la Cruz *et al.*, 2014). The deployment of artificial reef structures and subsequent transplantation of corals upon them has generally shown a positive trajectory of coral and fish recovery in a severely degraded reef area over a short timescale (2 years). Given the continued growth of transplanted corals and natural recruitment of corals on the artificial reef structures it would be expected that, in the long-term, the eventual development of large corals could support larger populations of fish (Halford *et al.*, 2004), and contribute to showcase the potential use of combined artificial reef structures and coral fragment transplantation as a conservation tool in severely degraded reefs.

The present study, being one of a few implemented in the WIO, raises many opportunities for reef researchers and local communities to continue partnering to develop this method further, as well as monitoring in order to understand fully the benefits and/or impacts of this reef restoration approach. If scaled-up with consideration of initial successes and lessons learnt, this combined use of artificial reef structures and coral transplantation can contribute towards the UN-proclaimed Decade of Ecosystem Restoration (2021-2030), which aligns with a wide range of Sustainable Development Goals (SDGs), including enhancing healthy of coastal ecosystems and biodiversity conservation (UNEP, 2019). In light of this global target, the ongoing assessment of this artificial reef can be used to gain insights into the effectiveness of artificial reefs as a conservation tool for habitat restoration and recovery in degraded coral reef ecosystems, and become an important focus for coastal communities.

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