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Reef fish abundance enhancement through low-cost coral reef restoration practices: a case study in Tioman island, Malaysia

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Abstract

Coral reefs and their inhabitants are threatened by several anthropogenic impacts worldwide. While active coral conservation activities have emerged aiming to maintain reef ecological functions and services, the effect of coral restoration practices on high trophic levels is still poorly investigated. We assessed the effect of small-sized concrete structures on the fish community by comparing the early fish colonization between a treated area and a coral rubble area (used here as control) over 12 months with a monsoon period in between. The treated area nearly doubled in total fish abundance and species richness compared to the control, showing that this method can locally enhance the community of reef fish. *Pomacentrus chrysurus* was the dominant species in all areas, although *Eviota distigma* contributed the most to the average between-group dissimilarity between areas (8%, $p = 0.02$). Fish abundance also exhibited differences between fish life stages (adult, juvenile) ($H = 28.852$, $p < 0.001$), among trophic categories (carnivore, omnivore, herbivore, planktivore, and corallivore) ($H = 90.615$, $p < 0.001$), and between community types (cryptobenthic and non-cryptobenthic) ($H = 52.084$, $p < 0.001$). These effects seem to be boosted by a monsoon period, as the increase of species richness and fish abundance was steeper after the monsoon. Our study shows a positive influence on the reef fish community from the deployment of small-sized concrete structures and highlights human intervention as an essential factor in recovering these iconic ecosystems.

1. Introduction

Coral reefs are among the most complex ecosystems on Earth hosting up to 830.000 species worldwide (Fisher et al. 2015) but are also one of the most threatened marine habitats (Hughes et al. 2017; IPCC, 2018; Bindoff et al., 2019). Rising water temperatures (Bellwood et al. 2004; Hughes et al. 2018), ocean acidification (Hoegh-Guldberg et al. 2007), and other anthropogenic disturbances such as destructive fishing practices or water pollution (Burke et al. 2011) are causing coral reef degradation, as these accelerate reef erosion and decrease the overall structural complexity of this ecosystem (Graham and Nash 2013). Consequently, reef-associated species are rapidly losing their habitats (Jones et al. 2004), thus leading to a stark biodiversity decline in the iconic ecosystem formed by corals.

Coral reef restoration (hereafter abbreviated as “CRR”) is a common tool aiming at recovering damaged reefs (Boström-Einarsson et al. 2020; Hein et al., 2021) that is defined as an active intervention aimed to assist the recovery of reef structure, function, and key reef species in the face of rising climate and anthropogenic pressures, promoting reef resilience and the sustainable delivery of reef ecosystem services (Hein et al., 2021). Some researchers state that coral reefs as we know them may disappear in a scenario of up to 2°C warming due to future coral bleaching, and up to 90% of coral reefs could be lost even with an increase of 1.5°C (IPCC, 2018; Bindoff et al., 2019; Bellwood et al., 2019; Morrison et al., 2020). In this context, many experts argue that coral reef restoration is merely a band-aid solution and a distraction from global actions on threat reduction. Therefore, CRR techniques may not prevent the deterioration of coral reefs caused by climate change or ocean acidification (Normile 2009; Hein et al., 2021), but they have proven to be a successful tool in quickly recovering the ecological functions and

ecosystem services of degraded reef regions that may not be resilient enough to recover by themselves (Clark and Edwards 1999; Lirman and Schopmeyer 2016). Artificial reefs (hereafter abbreviated as “ARs”) are used to quickly enhance structural complexity, and by providing substrate for recruitment, they can over time lead to increased coral cover. In combination with direct transplantation, deploying ARs in damaged reefs where rubble patches are abundant contributes to increasing coral cover (Ceccarelli et al. 2020) and enhancing fish populations or fisheries productivity (Ferse, 2009; Lowry et al. 2014; Becker et al. 2018; Folpp et al. 2020). ARs seem to positively affect reef fish communities (Yap 2009; Richardson et al. 2017), especially in areas of low complexity and low coral cover (Ferse, 2009) as they provide shelter for many fish species due to their higher complexity compared with adjacent damaged areas (Lingo and Szedlmayer 2006; Ferse, 2009; Fadli et al. 2012; Hylkema et al. 2020). Moreover, food availability increases significantly for reef species as the hard surfaces of the structure are rapidly colonized by bryozoans and hydrozoans, followed by coral recruits, algae, and other macro-invertebrates (Schuhmacher, 1977; Cresson et al. 2014; Fariñas-Franco and Roberts 2013; Lozano-Cortés and Zapata 2014), attracting fish species. While this early colonization is most likely due to the result of the dispersal and recruitment of propagules from local hard substrata (Herbert et al. 2017), the reason why other mobile organisms colonize artificial habitats in sequential patterns remains poorly understood. Some researchers showed that the colonization of artificial reefs by fish is initially rapid (Bohnsack et al., 1994; Golani and Diamant, 1999), primarily by adult fish from adjacent areas (Walsh, 1985).

Reef fish play an important role in the dynamics of coral reefs, as they constitute an essential component for mitigating the impacts of coral bleaching and influencing their susceptibility/resilience and recovery (Chase et al., 2018), as for preserving reef resilience (Bellwood et al. 2006), its ecosystem functions and services (Bellwood et al 2019). For instance, they regulate community structure by enhancing or inhabiting other reef species (Loch et al. 2010; Young and Bellwood 2012; Boaden and Kingsford 2015), removing sediment to contribute to the maintenance of coral-dominated reefs (Goatley and Bellwood 2010; Chase et al., 2020), preventing changes from coral to macroalgal dominance (Cheal et al. 2010; Vergés et al. 2012; Plass-Johnson et al. 2015) and mitigating coral disease (Raymundo et al. 2009). To recover coral reef ecosystems, it is thus essential to re-establish the fish communities. Herbivorous fishes are essential for successful CRR programs as they prevent algae from smothering coral transplants (Edwards and Gomez 2007; Hylkema et al., 2023). In addition, they help to make more free space for coral recruits as algae are removed, and they contribute to reducing coral-algal interactions (Abelson et al. 2016). Cryptobenthic reef fishes, a group of 17 reef-associated fish families characterized by species less than 5 cm in length (Brandl et al. 2018, 2019), also seem to play a critical role in reef trophodynamics, as these tiny fishes feed on many of the large predators in coral reefs (Goatley and Brandl 2017; Brandl et al. 2019). However, very few studies have considered monitoring cryptobenthic fishes in CRR programs (Komyakova et al. 2019).

In recent CRR efforts, the use of small structures has emerged aiming at either combining substrate stabilization with the provision of habitat structure or providing structure only (Ceccarelli et al. 2020). Concrete structures are widely used in CRR programs (Ceccarelli et al. 2020; Razak et al. 2022), as they are one of the most suitable materials for transplanting coral fragments (Chen et al. 2018). Generally,

CRR research has focused on investigating coral growth and survival rates of transplanted coral fragments (Sabater and Yap 2002; Young and Bellwood 2012; Lirman and Schopmeyer 2016; Williams et al. 2019), and more recently on investigating the response of the fish communities to the deployment of artificial reefs in near-shore environments (Lowry et al. 2014; Opel et al. 2017; Paxton et al. 2018; Lemoine et al. 2019; Hylkema et al. 2020). While artificial reefs support comparable levels of fish density, biomass, species richness, and diversity to natural reefs (Paxton et al., 2020), differences in the ecological benefits of different types of artificial reef habitats are still poorly understood. Artificial reefs should be considered strategically based on location-specific scientific assessments and resource needs to maximize habitat enhancement (Paxton et al., 2020).

In Malaysia, coral reefs are in fair condition, with an average live coral cover of 40.63% in 2019 (ReefCheck Malaysia 2019). Moreover, the east coast of Peninsular Malaysia and the islands nearby receive the impacts of an annual wet monsoon period (from November to March each year), causing powerful waves and tidal surges (Yaacob and Hussain 2005). Environmental changes during storm events influence the behavior of fishes depending on their morphological and physiological characteristics, and these sensitive species may migrate from the area or seek refuge in the reef substrate to weather the storm (Munks et al., 2015). Tropical storms affect the fish community due to a reduction in species diversity (Jalal et al., 2012) and density (Gavriel et al., 2023), but also to habitat loss. Coral reefs regularly experience fragmentation episodes due to disturbances, including storms, contributing to the accumulation of rubble on the substrate (Fox and Caldwell, 2006; Viehman et al., 2018). Indeed, around 15% of Malaysian coral reefs´ substrate consists of coral rubble. Small fragments – and potentially coral recruits –, regardless of their functional morphotype, may not recruit in areas of strong wave action where unconsolidated substrates are frequently mobilised, but may persist in rubble beds in lower-energy environments (Keyton et al., 2020). Human intervention aiming at stabilizing rubble substrates may be necessary if several requirements are met (Keyton et al., 2023).

ARs have been used since the 1980s aiming at mainly enhancing fish abundance and increasing habitat quantity and structural complexity of denuded reefs (Clark and Edwards, 1999; Fox and Caldwell, 2006; Cabaitan et al., 2008; Higgings et al., 2022). Of these, ARs, although popular for fish enhancement, have not been used as extensively for coral restoration (Boström-Einarsson et al., 2020). Whereas the link between artificial reefs and fish communities has been investigated for years, there is still limited research on fish community responses to reef restoration practices (Hein et al., 2020; Seraphim et al., 2020). In this study, we aimed to investigate the fish response to newly deployed small-sized concrete structures as part of the reef rehabilitation efforts conducted in a local coral reef in Tioman Island, Malaysia. We assessed the fish communities by comparing two areas, one consisting of two sites where we applied restoration efforts, and one non-treated site considered as control. Following natural colonization processes, fish prey such as algae, crustaceans, or mollusks, rapidly colonize new habitats (Cresson et al. 2014). Thus, we expected to observe a faster colonization in the treated area by these organisms and therefore, a higher abundance of fish from all trophic groups (herbivorous, carnivorous, and omnivorous) compared to the control. The potential shelter provided by the artificial structures in the restoration sites led us to hypothesize that: (1) the treated area would show higher richness and

abundance of fish species than the control over time; (2) species composition would change between the treated vs control area, and (3) total abundance grouped but also the composition of trophic groups.

2. Material and Methods

2.1. Study site

The study was conducted in Middle Reef (2º 47.369´ N, 104º 12.259´ E), located in the centre of Juara Bay, located on the east coast of Tioman Island (Fig. 1). This reef was selected for rehabilitation purposes due to its high coverage of unconsolidated substrates, such as coral rubble and sand. It has an average depth of 5 m, reaching a maximum depth of 7 m. Tioman´s waters extending to 2 nautical miles from the coastal line were declared Marine Park in 1994 under the Fisheries Act (1985) to protect its biodiversity (DMPM 2011). Nevertheless, the increase in tourism and coastal development of adjacent areas has the potential to negatively impact the reefs and lead to low live coral coverage at some of the reef sites reported on the western coast of Tioman (Shahbudin et al. 2017; Akmal et al. 2019). Furthermore, reef sites in the offshore area of Tioman suffered a bleaching event in 2010, resulting in partial mortality of approximately 50% of live corals (Tan and Heron 2011). The east coast of the island is the least developed and has the highest mean coverage of live corals. Additionally, Juara Bay receives very little wave action and smooth currents during the dry season from April to October. However, a monsoon period hits the east coast of Tioman every year between November and March, which creates a period of rough seas and strong wave energy. A combination of these events and several bleaching events turned some large regions of living coral into coral rubble.

Figure 1 Location of Middle Reef (in green) in Juara Bay, on the east coast of Tioman island

2.2. Experimental design

Three damaged reef areas of 10 m^2 each were delimited at Middle Reef, two for reef rehabilitation purposes and one for control. the reef flat at Middle Reef has large unconsolidated substrate zones (rubble) where hard coral colonies try to grow each year. This reef is subject to strong waves during the monsoon period, so it was necessary to secure the seafloor to increase coral cover (Keyton et al., 2023). We placed twenty-four equally spaced structures made from concrete of 50x30x20 cm in size and about 30 kg. in weight each. on each area delimited for restoration purposes (Fig. 2). Each block was made by pouring concrete into a wooden frame with 6 to 8 glass bottles. These were placed upright into the concrete so that once the concrete dried the narrow part of the bottle stuck out approximately 8 cm. Concrete structures have been used in reef restoration to stabilize unstable areas or unconsolidated substrates (Ceccarelli et al. 2020; Hammond et al. 2020) since it is a heavy and durable material. In recent CRR efforts in Malaysia, concrete structures have been selected as the most suitable material for planting coral fragments (Chen et al. 2018).

Live coral fragments between 5 to 10 cm in length were collected from the surrounding seafloor by SCUBA or freediving. Only loose fragments were chosen to minimize damage to already existing coral

colonies (Bowden-Kerby 2001), and they were collected at the same depth range as the study area. 24 hours prior to their transplantation, coral fragments were placed close to the rehabilitating areas to minimize the stress from this process. We transplanted eight coral fragments belonging to eight different reef-building coral species in each block, making a total of 158 transplanted fragments in each restoration area. Coral fragments were attached to the bottleneck of the glass bottles using cable ties since this fastener is more effective for increasing fragment self-attachment than other materials (García-Baciero et al., 2023). The bottle opening was left open to increase the development of fish communities, for which holes and hole sizes in artificial reefs are important aspects (Hixon and Beets, 1989). The selected species included the acroporids Acropora digitifera (Dana 1846), Acropora muricata (Linnaeus 1758), Isopora brueggemanni (Brook 1893), and Montipora sp.; the pocilloporids Pocillopora damicornis (Linnaeus 1758), and Stylophora sp., and the poritids Porites cylindrica (Dana 1846) and Porites lobata (Dana 1846). We selected these species to provide more structural complexity to the rehabilitation site since they all have different growth forms, but also due to their higher availability in the donor sites. As several studies have reported that macroalgae can decrease coral survival rates (Hughes et al. 2007; Cetz-Navarro et al. 2015; Webster et al. 2015; van Woesik et al. 2017), maintenance activities were performed twice a week using SCUBA and toothbrush to reduce coral-algae competition and accelerate the growth of transplanted fragments.

Fig 2. Reef restoration area consisting of concrete blocks with glass bottles and iron rebar sticks on Middle Reef, Tioman Island, Peninsular Malaysia

2.3. Fish assemblage monitoring

Each area was surveyed by two researchers using SCUBA and underwater slates between July 30th, 2019, and June 21st, 2020. Fish surveys only stopped during the monsoon period, which lasted from November 2019 to mid-April 2020, resulting in a total of 36 fish surveys conducted in each area. All surveys started right after the structures were deployed and all the fragments were transplanted and continued till almost 11 months after deployment. Surveys were performed by two researchers who were trained in species identification prior to data collection. Surveys were conducted using SCUBA and were performed between 10 am and 2 pm, to minimize time-related effects on fish counts. For each survey, an underwater visual census (UVC, English et al. 1997) was conducted consisting of a 5 min stationary point-count (Bohnsack and Bannerot 1986; Lowry et al. 2012) followed by a thorough search of the internal spaces of the artificial reefs to record all cryptobenthic fish. All fish within a virtual cylindrical column, extending 1 m sideways of the reef and extending 2 m upward from the bottom were included in the survey.

Fish individuals were grouped by life stage (i.e., juvenile or adult) according to their size and body pattern colour. Subadult individuals as well as those transitioning from juvenile to adult were considered as adults. Fish species were identified to the lowest taxonomic rank using Indo-Pacific Reef Fish identification guides (Bergbauer et al. 2014; Allen et al. 2005) and FishBase website (Froese and Pauly 2020). We collected photos of fish species using an Olympus Tg-5 underwater camera to assist with identification afterward when uncertainty in *in situ* species identification. Additionally, fish were classified into trophic groups related to their diets in the adult life stages, following previously defined dietary

categories for reef fishes (Sandin and Williams 2010; Mouillot et al. 2014; Morais and Bellwood 2019; Siqueira et al. 2020). Five major groups were used: carnivores (including piscivores, invertivores), omnivores, planktivores, herbivores (including macroalgivores, grazers, and detritivores) and corallivores (fishes that feed on coral polyps). Moreover, fish were classified into cryptobenthic and non-cryptobenthic communities according to their cryptic appearance and/or behaviour.

2.4. Data analysis

Total fish abundance for each area was calculated as the highest number of individuals from each species, counted at once during the survey. This is a modification of the metric *MaxN*, which typically is a conservative count of the maximum number of fish observed on a single video frame (Cappo et al. 2004). Then, we calculated the average total fish abundance per treatment type. Statistical analyses were performed in R version 4.2.2 (R Development Core Team 2022). Generalized Linear Models (GLM) using the "MASS" package (Venables and Ripley 2002) in R were used to test whether the total abundance and species richness differed between treated areas and the control, accounting for temporal change (i.e., sampling date), the effect of the monsoon, and their interaction term (fixed factors). Since we observed over-dispersion when summing Pearson's residuals (Lindén and Mäntyniemi 2011) in the total abundance data, we used a negative binomial distribution in the GLM instead of a Poisson distribution (suitable for count data). On the other hand, a Poisson distribution was used in the GLM for species richness.

Differences in species composition occurring in different areas, including their nestedness and turnover components, were examined through the Jaccard abundance-based index (Legendre 2014), using the package "BAT" v. 2.9.2 (Cardoso et al. 2015). The percentage of variability in the community composition of fish species observed across samples was assessed through a permutational analysis of variance (PERMANOVA) using distance matrices and the R package "vegan" v. 2.6-4 (Oksanen et al. 2015). Sampling data was also included in the latter model to account for the temporal variation in the reef at structuring differences of species assemblage. To address the contribution of each species to the average dissimilarity between areas, a similarity percentage analysis (SIMPER) (Clarke 1993) with 1000 permutations was performed, using the "vegan" package. Moreover, to test for differences in total fish abundance grouped by life stage, trophic categories, and community types between treated and non-treated regions, a non-parametric Scheirer-Ray-Hare test was performed using the *rcompanion* package (Mangiafico 2022) in R. The effect of the monsoon on total fish abundance grouped by each category was tested using a Kruskal-Wallis test in R.

Model selection among the candidate GLM models was conducted using Akaike Information Criterion (AIC) values based on minimum AIC, with the best model having the lowest AIC score. Residuals were plotted to examine if the model assumptions were met. Wald χ^2 tests were also performed for statistical inference of the fixed factors using the *Anova* function in the "car" package (Fox and Weisberg 2019). In the results, only the final models are shown, and the outcomes of the candidate models are not reported. P values < 0.05 were considered statistically significant and reported values are means ± se.

3. Results

3.1. Total abundance and species richness

A total of 51 different species and a total abundance of 1,866 individual fish were observed throughout 36 surveys. 804 fish individuals belonging to 47 species were observed in the treated area, whereas only 278 fish individuals from 21 species were observed in the control (Table SI1). *Pomacentrus chrysurus* (White-tail damselfish) was the most abundant species both in the treated area and control, followed by *Eviota distigma* (Twospot dwarfgoby) and *Halichoeres melanurus* (Tail-spot wrasse) in the treated area, and by *Pomacentrus siamsiang* (Blueback damsel) in the control (Table SI1).

The results from the GLM model indicated that total fish abundance improved positively and significantly in the treated area compared to the non-treated (IRR = 13.31, $p < 0.001$), so did species richness (IRR = 7.31, $p < 0.001$) (Table 1). Fish abundance increased significantly in both areas over time, but slightly faster in the treated area than in the control (IRR = 1.01, $p < 0.001$), whereas species richness increased at the same rate in both areas over time (IRR = 1, $p = 0.0293$).

Figure 3 Regression lines accounting for temporal variation of total abundance (A) and species richness (B) between pre- and post-monsoon periods in each area. Coloured bands indicate the standard error of the regression lines

There was a significant reduction in both total fish abundance and species richness at the beginning of the post-monsoon period compared to the end of the pre-monsoonal state (Fig. 3). For both parameters, the effect of time was stronger after the monsoon, showing a faster growth following the monsoon (Fig. 3).

3.2. Fish community composition

Fish species composition accounting for total fish abundance varied between areas (PERMANOVA, F = 12.8787, $p = 0.001$), over time (PERMANOVA, F = 4.7510, $p = 0.001$), as well as with the interaction term between area and time (PERMANOVA, F = 5.0345, $p = 0.001$). Moreover, the differences in species composition were mostly due to differences in species richness between areas (~ 99%).

Fig. 4 The species composition of fish communities in the reef, showing their abundances (expressed in logarithm) from light to dark blue. Rows indicate the samples of blocks (purple) or control (yellow), whereas columns denote the species; the dendrogram was built using the method '*complete*' by function '*hclust*'

The species *Eviota distigma* exhibited the highest contribution to the overall dissimilarities between communities (SIMPER, 8.00%; $p = 0.02$), followed by *P. chrysurus* (SIMPER, 6.79%; $p = 0.001$), as their abundance was much higher at the treated area compared to the control. The species *P. siamsiang* also contributed highly to these dissimilarities (SIMPER, 6.75%, $p = 0.04$) though this species was more

abundant in the control area. The main species that accounted for most of the dissimilarities between areas are shown in Table 4.

Significant differences in total fish abundance grouped by fish phase (adult vs. juvenile) were found between life stages (Scheirer-Ray-Hare, H = 28.852, $p < 0.001$), between areas (Scheirer-Ray-Hare, H = 55.314, $p < 0.001$), as well as between their interaction (Scheirer-Ray-Hare, H = 6.671, $p = 0.0098$). A much larger number of juvenile and adult phase individuals was observed in the treated area than in the control (Fig. 5). The monsoon significantly affected the total abundance grouped by fish phase (Kruskal-Wallis test, $\chi^2 = 8.3305$, $p = 0.0039$). In addition, the number of juvenile individuals after the monsoon was significantly higher in both areas than before this period, despite the observed enhancement in the abundance of juveniles in the treated area.

Differences in total abundance between trophic categories were found (Scheirer-Ray-Hare, H = 90.615, $p < 0.0001$), between treated and control areas (Scheirer-Ray-Hare, H = 43.435, $p < 0.0001$), as well as between their interaction (Scheirer-Ray-Hare, H = 9.187, $p = 0.0269$). Carnivore, omnivore, and planktivore were the most influenced groups by treatment, whereas the effect was less clear for herbivores and almost inexistent for corallivores (Fig. 5). The monsoon did not affect significantly the total abundance grouped by trophic categories (Kruskal-Wallis test, $\chi^2 = 0.6050$, $p = 0.4367$).

Total fish abundance grouped by type of community also showed significant differences in community types (Scheirer-Ray-Hare, H = 52.084, $p < 0.0001$), between areas (Scheirer-Ray-Hare, H = 38.205, $p < 0.0001$), but no between their interaction (Scheirer-Ray-Hare, H = 2.890, $p = 0.0891$). The non-cryptobenthic community was much more abundant than the cryptobenthic community in both areas (Fig. 4). There were also significant differences in total abundance grouped by community types due to the monsoon effect (Kruskal-Wallis, $\chi^2 = 6.6881$, $p = 0.0097$). Indeed, more individuals from both community types were found after the monsoon (Fig. 5).

Fig. 5 Total fish abundance differences between treated and non-treated areas during pre- and post-monsoon periods, grouped by fish life stages, trophic groups and type of community

4. Discussion

The present study tested the influence of the deployment of concrete structures with coral fragments directly transplanted on them on the total fish abundance and species richness during 11 months after transplantation. Although it is a short-term study, this research showed an early colonization of the artificial structures by reef fish species. Indeed, total fish abundance and species richness increased by nearly 50% in the treated area compared to the control by the end of the study. The positive effect of the artificial structures in damaged reef areas matched our expectations, as any addition of hard substrate provides shelter and food availability (Graham and Nash 2013), enhancing ultimately fish communities nearby. While here we transplanted 158 coral fragments on each plot, we believe that the effect of the coral fragments themselves was negligible since their initial size prior to transplantation was around 5

cm in length. While it may be possible that by the end of the study the size of coral fragments contributed to enhancing fish abundance and species richness, it has been shown that coral colonies ≤ 10 cm in height do not appear to contribute to reef habitat, underscoring the value of colony height as an indicator of fish habitat (Fisher, 2023). Additionally, the size of the coral fragments during most of this study was too small to influence on fish behaviours such as predator avoidance or foraging. Fisher (2023) investigated the relationship between fish population measures (density, taxa richness and biomass) and both biological (taxa richness and live surface area) and physical (number, size and structural complexity) attributes of scleractinian coral populations. He found that few pomacentrid and labrid fish species correlated with coral colony height, whereas many others did not, suggesting that colony height appears useful for estimating fish density but not fish species. Therefore, other factors such as season, location, depth, currents, time of day, and weather are among a variety of plausible reasons for the differences.

4.1. Effects of blocks on the abundance and species richness

The artificial structures were set in areas where most parts of the substrate were unconsolidated (rubble) and that were subset to strong wave energy every year. Since small fragments and probably coral recruits may not succeed in recruiting in frequently mobilized rubble beds (Keyton et al., 2020), these structures aimed first at stabilizing the seafloor, while increasing the reef complexity. Complex concrete structures with different-sized holes and crevices have shown positive effects on colonising fish species (Sherman et al. 2002; Lemoine et al. 2019; Hylkema et al. 2020). Thus, considering that coral fragments were too small to be the key factor in increasing fish abundance and species richness, the existing spacing among the structures and between the blocks and the seafloor could explain the differences observed in total abundance between areas. Admittedly, many researchers agree that the structural complexity of artificial structures is one of the key components that lead to a rapid increase in fish abundance (Lemoine et al. 2019; Hylkema et al. 2020). Previous studies have shown that placing artificial reefs in areas of low complexity with a low percentage of coral cover resulted in very high fish abundance over time (Edwards and Clark 1993; Clark and Edwards, 1999; Ferse, 2009). Moreover, fish abundance seems to be correlated with habitat availability (Lingo and Szedlmayer 2006; Hackradt et al. 2011; Hylkema et al. 2020). The materials used to build the structures (White et al. 1990; Masuda et al. 2010; Bateman 2015), as well as the type of structures, contribute to enhancing the reef fish abundance (Hein et al. 2020; Hylkema et al. 2020). Our results confirm this and suggest that these complex structures are advantageous to mitigate, on a small scale, the effects of extreme climate events on the reef fish community. While the differences in oceanographic conditions and methodologies among reef restoration practices make them hard to compare, this variability of results emphasizes the need for further research to better understand the reef fish communities' response to reef restoration, notably the combination of artificial structures and direct transplantation methods.

4.2. Effects of monsoon and time on the abundance and species richness

In this study, we show that fish abundance differed between the pre- and post-monsoon periods. While total fish abundance and species richness increased in the treated area before the monsoon, both variables exhibited a steeper increase after the monsoon indeed. This post-monsoon increase might be due to a strong influence of monsoon-induced seasonality (Abesamis and Russ 2010; Abesamis et al. 2015), which affects resource availability (Hee et al. 2020). Nevertheless, both variables showed lower values at the beginning of the post-monsoon period comparing them with the end of the pre-monsoon period. This decrease in abundance and richness at the beginning of the monsoon might be due to the fish's tendency to find refuge under strong current conditions (Eggertsen et al. 2016). On the east coast of Peninsular Malaysia, the northeast monsoon (November to March each year) brings big waves and strong currents (Yaacob and Hussain 2005). Whilst none of the present studies addresses the monsoonal effects on reef fish communities in Tioman Island, Jalal et al. (2012) reported a reduction in species diversity in an estuary located on the east coast of Peninsular Malaysia, which could be extrapolated to the nearby islands. Moreover, a storm can cause a decrease in fish density of more than 50% compared to a pre-storm state (Gavriel et al., 2023). Nevertheless, we reported here the effects of only one monsoon season during the early fish colonization. Thus, further research having a longer monitoring period would be advisable to assess its effect on the fish assemblages. Finally, we found a significant increase in pomacentrid species in both areas after the monsoon, but steeper in the treated area. Indeed, a study by Abesamis and Russ (2010) recorded the highest recruit densities in four pomacentrid species during the southwest monsoon, which is associated with calmer and more stable conditions compared to the northeast monsoon, whereas the lowest densities were observed during the northeast monsoon. Still, there is limited research on the potential relationships between the spawning or recruitment of coral reef fish and monsoon periods in the South-East Asia region (Abesamis and Russ 2010; Claydon et al. 2014 but see Hamilton et al. 2011).

4.3. Effects of blocks on fish community composition

In the treated area we observed a higher representation of pomacentrid, labrid, and gobid species, such as *Pomacentrus chrysurus*, *Eviota distigma*, or *Halichoeres melanurus* than in the control. Indeed, Pomacentridae, Labridae, and Gobidae are some of the most abundant reef-associated families (Lirman 1999). An important contributor to increasing reef fish diversity and abundance is the restoration-driven efforts to enhance coral cover (Edwards and Clark 1993; Cabaitan et al. 2008; Yap 2009; Fadli et al. 2012). Coral fragments transplanted in high-density plots (Cabaitan et al. 2008; de la Cruz et al. 2014) or used in a combination of both artificial structures and direct coral transplants in high densities (Ferse 2008) have shown a positive effect in increasing fish diversity and abundance. While the presence of coral transplants can significantly affect the associated fish community within days to weeks (Ferse 2009), we believe that the coral fragments transplanted here were too small to make a significant effect on total fish abundance and species richness that cannot be explained by the structures themselves. Indeed, coral-related fish, such as corallivorous fish and apogonid species did not show significant

differences between areas. Nevertheless, it is most likely that over time there will be a significant increase in fish abundance accompanied by the growth of transplanted coral species.

Increased food availability could play an important role in determining fish composition. The concrete structures used in this study had gaps in between which were occupied by small invertebrates and small fishes that may have attracted larger fish in the first place. This could explain why carnivore and omnivore fish were more predominant in the treated area than in the control, as well as herbivorous fishes that might have been attracted after rapid colonization of the structures by filamentous algae.

4.4. Implications

The findings shown in this study suggest that coral reef species are benefited from deploying small-sized concrete structures for reef rehabilitation purposes, as new stable habitat is created. This method is affordable and accessible for low-income organizations or institutions working in reef rehabilitation, especially those located in remote areas. Indo-Pacific countries are experiencing a tremendous decline in coral cover due to anthropogenic pressures (Bruno and Selig 2007). Precisely, many of them are low-income countries that are fighting the effects of climate change differently than highly developed countries (Taconet et al. 2020). Therefore, this study implies a cost-effective alternative for funding constraints conservation entities that are willing to protect coral reefs and their inhabitants.

4.5. Conclusion

Our study showed that small-sized concrete structures applied in areas of low coral cover rapidly enhance total fish abundance and species richness. Furthermore, it shows that this increase is further accentuated after the monsoon period. As presented here, the differences in species composition among areas were in 99% due to species richness rather than species replacement. The treated area also exhibited differences in fish abundance grouped by life stage, trophic categories, and community type compared to the non-treated area.

Statements and Declarations

Compliance with Ethical Standards

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Author contributions

AGB conceived the main idea and experimental design. Material preparation and data collection was performed by AGB, AGH, MA, and EHB. Data analyses were performed by AGB, JGZ, and GGG. The first draft of the manuscript was written by AGB and MA, and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Data Availability

The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Ethics approval

This article does not contain any studies with human or animal subjects

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Tables

Table 1 Summary of the results from the logistic regression, showing the predictors that significantly affected total fish abundance. Fixed effects are Area (Blocks/Control, categorical), Time (days, continuous), Monsoon (before/after, categorical), and their interaction term

Total abundance				
Predictors	*Incidence Rate Ratios*	*std. Error*	*CI*	*Statistic*
(Intercept)	13.31 ***	1.30	10.97 – 16.15	26.41
Area [Control]	0.35 ***	0.03	0.29 – 0.42	-11.71
Time	1.01 ***	0.00	1.00 – 1.01	3.97
Monsoon [After]	0.01 ***	0.00	0.00 – 0.02	-7.09
Time * Monsoon [After]	1.01 ***	0.00	1.01 – 1.02	4.80
Observations	107			
R^2 Nagelkerke	0.949			
AIC	681.017			
Species Richness				
Predictors	*Incidence Rate Ratios*	*std. Error*	*CI*	*Statistic*
(Intercept)	7.31 ***	0.69	6.06 – 8.76	21.18
Area [Control]	0.36 ***	0.03	0.30 – 0.44	-10.58
Time	1.00 **	0.00	1.00 – 1.01	2.68
Monsoon [After]	0.10 ***	0.07	0.03 – 0.35	-3.61
Time * Monsoon [After]	1.01 *	0.00	1.00 – 1.01	2.18
Observations	107			
R^2 Nagelkerke	0.917			
AIC	476.933			
	* p<0.05 ** p<0.01 *** p<0.001			

Table 4 Summary of the results of SIMPER (Similarity percentage procedure), showing the species accounting for most of the dissimilarities between areas.

SIMPER						
Species	Family	Trophic group	Avb	Ava	% Contribution	P value
Blocks vs Control						
Eviota distigma	Gobiidae	Carnivore	2.01	0.89	8.0	0.02**
Pomacentrus chrysurus	Pomacentridae	Herbivore	3.38	2.31	6.79	0.001***
Pomacentrus siamsiang	Pomacentridae	Herbivore	0.93	1.86	6.75	0.04*
Halichoeres melanurus	Labridae	Carnivore	1.65	0.25	6.08	0.001***
Thalassoma lunare	Labridae	Carnivore	1.5	0	5.47	0.001***
Stethojulis trilineata	Labridae	Carnivore	0.74	0.33	2.91	0.008**
Ecsenius yaeyamaensis	Blenniidae	Herbivore	0.67	0.03	2.7	0.008**
Pseudochromis fuscus	Pseudochromidae	Carnivore	0.58	0	2.52	0.001***
Halichoeres erdmanni	Labridae	Carnivore	0.53	0.08	2.16	0.008**
Cephalopholis boenak	Serranidae	Carnivore	0.60	0	2.03	0.001***
Cheilinus chlorourus	Labridae	Carnivore	0.43	0.14	1.78	0.04*
Pomacentrus cuneatus	Pomacentridae	Herbivore	0.38	0	1.75	0.001***
Meiacanthus grammistes	Labridae	Carnivore	0.35	0	1.16	0.01**

% Contribution = Species contribution to average between-group dissimilarity; Avb = average abundance at Treated area; Ava = average abundance at Control

Figures

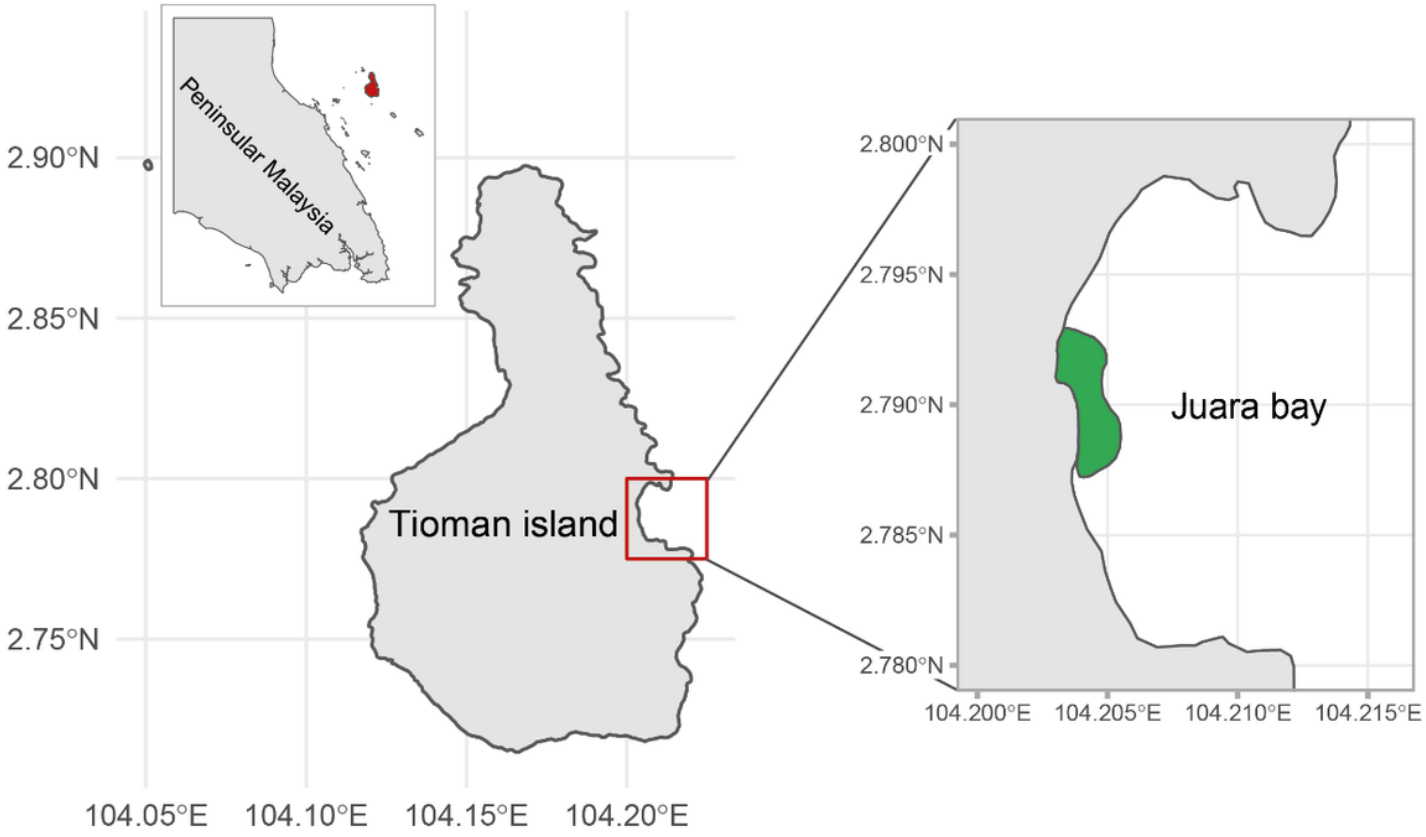


Figure 1

Location of Middle Reef (in green) in Juara Bay, on the east coast of Tioman island



Figure 2

Reef restoration area consisting of concrete blocks with glass bottles and iron rebar sticks on Middle Reef, Tioman Island, Peninsular Malaysia

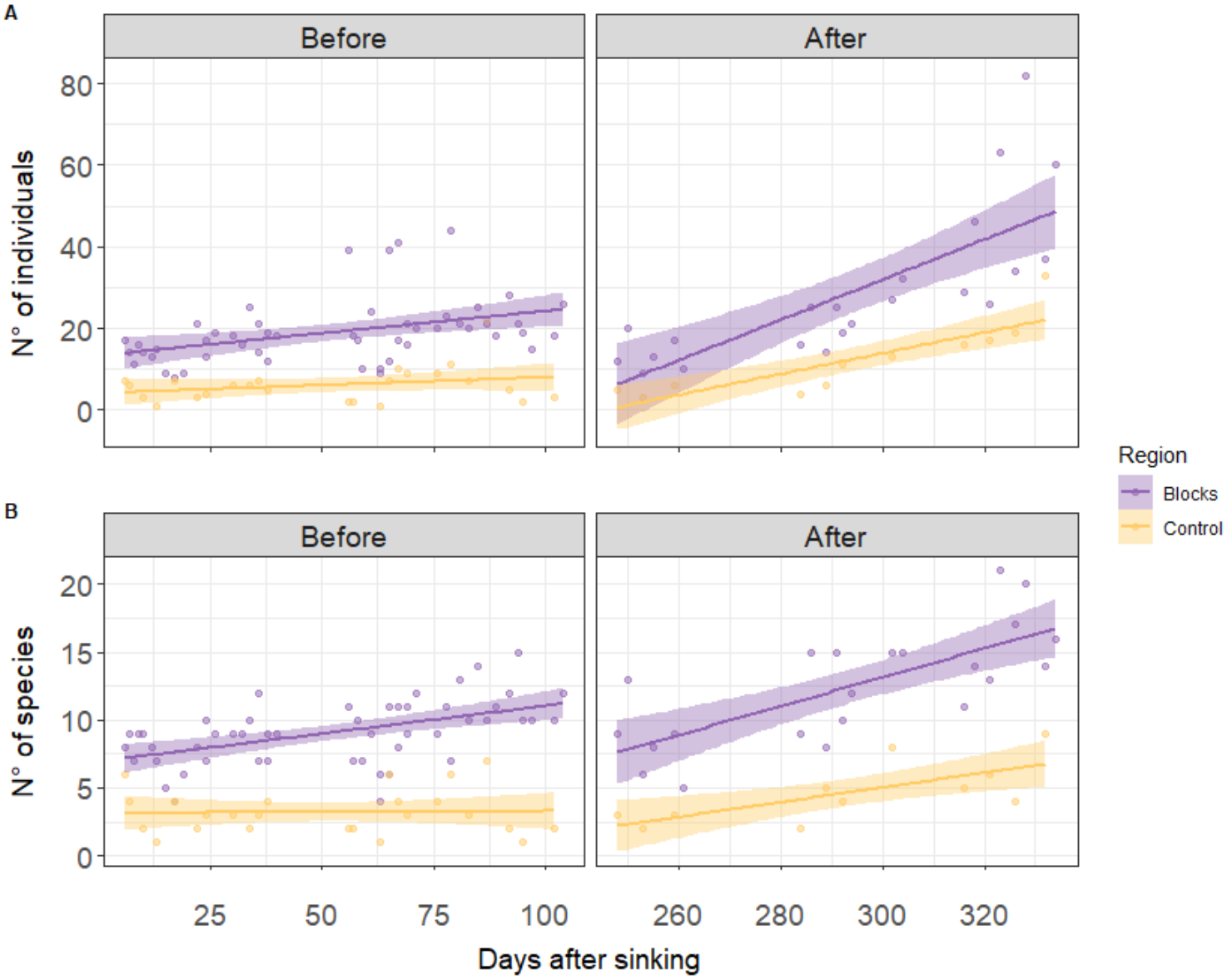


Figure 3

Regression lines accounting for temporal variation of total abundance (A) and species richness (B) between pre- and post-monsoon periods in each area. Coloured bands indicate the standard error of the regression lines

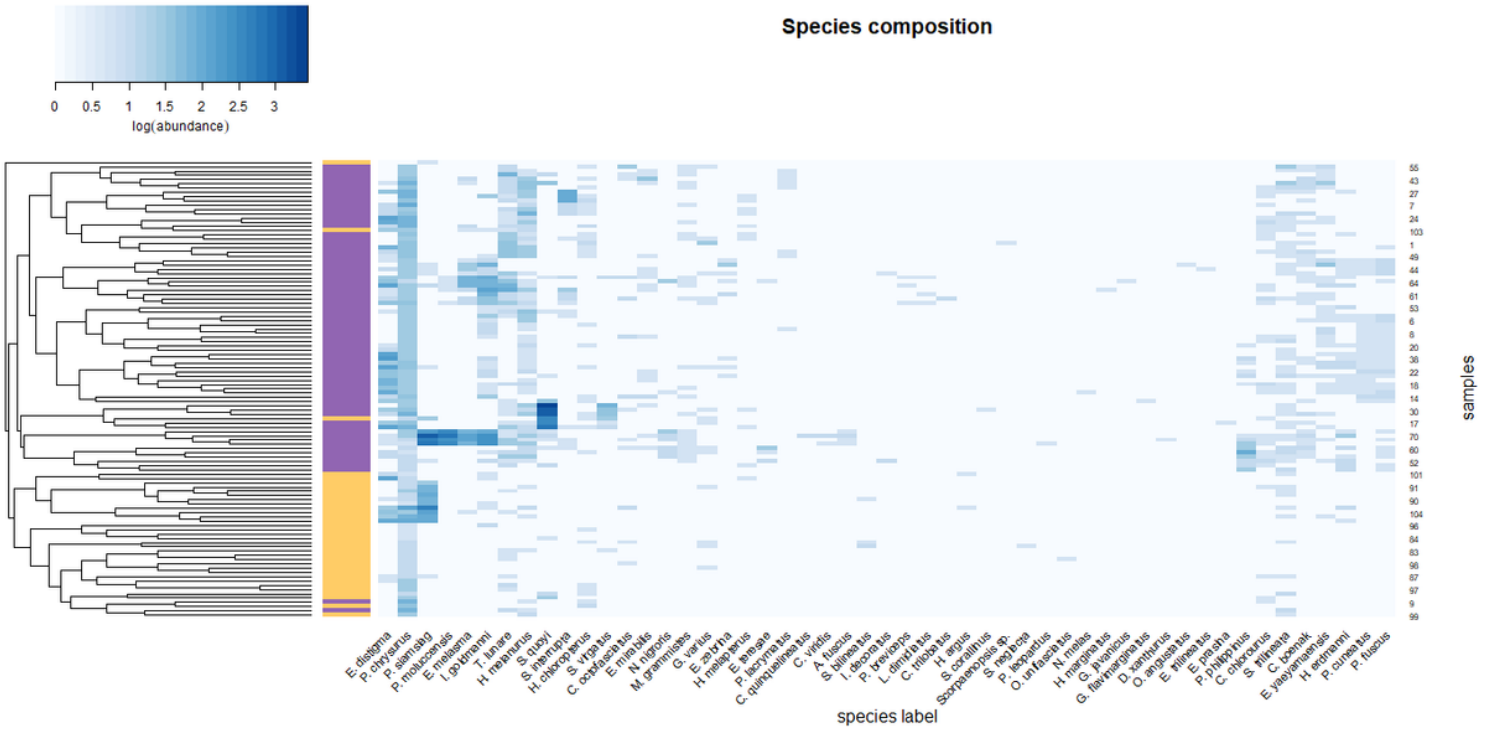


Figure 4

The species composition of fish communities in the reef, showing their abundances (expressed in logarithm) from light to dark blue. Rows indicate the samples of blocks (purple) or control (yellow), whereas columns denote the species; the dendrogram was built using the method '*complete*' by function '*hclust*'

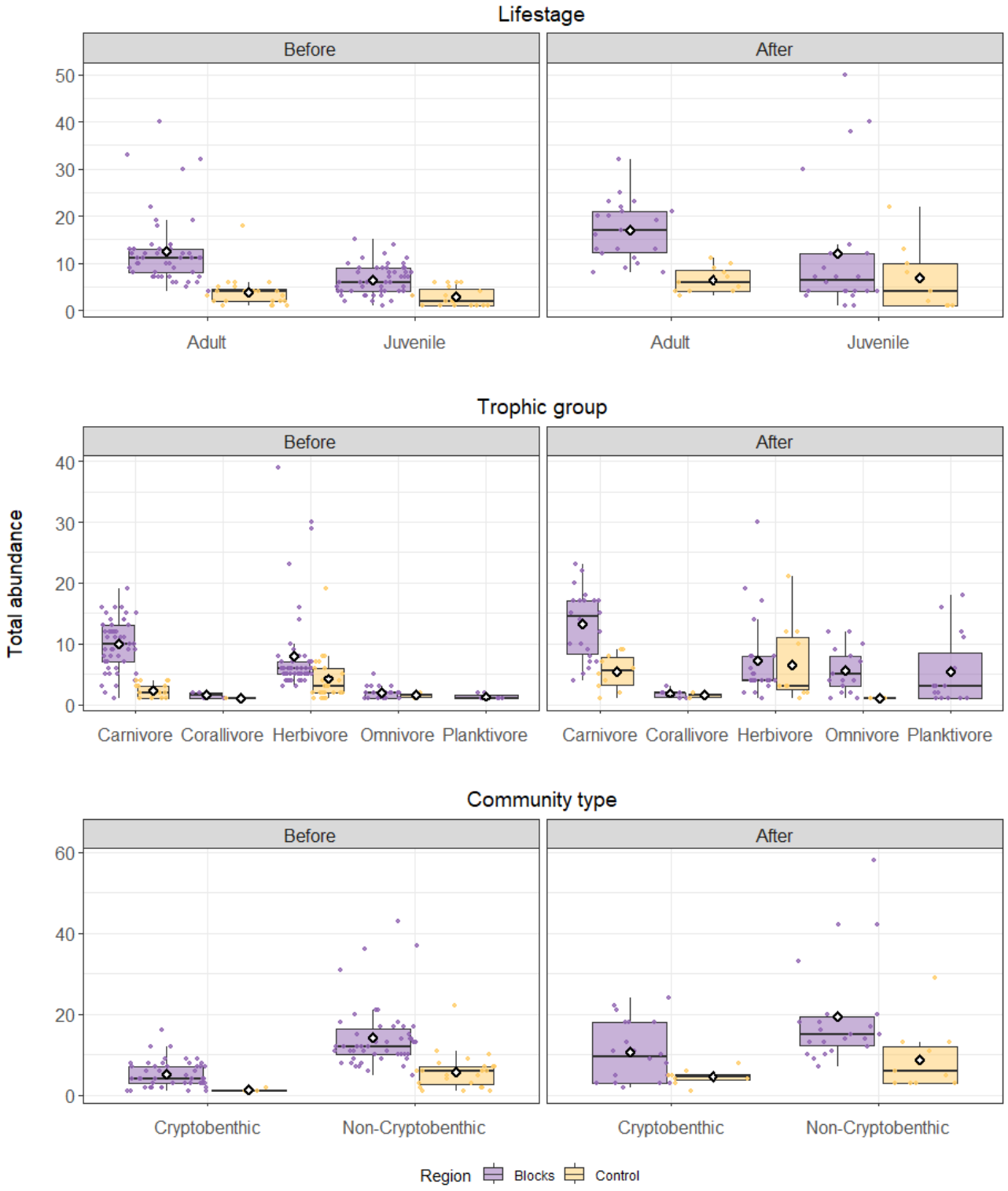


Figure 5

Total fish abundance differences between treated and non-treated areas during pre- and post-monsoon periods, grouped by fish life stages, trophic groups and type of community

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- TableSI1.docx