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SHORT NOTES

Long-term monitoring describes unique fish communities across natural and artificial reef habitats in Indonesia

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Abstract

Artificial reefs (ARs) are commonly used as a tool in Indonesia to restore marine habitats and provide local communities with ecosystem services previously lost due to reef degradation, notably those related to fishing and tourism. Now, Indonesia has more reef restoration programmes than any other nation in the world. However, only 16% of reef restoration programmes in Indonesia are thought to monitor their reefs, and those that do often lack funding and capacity to conduct long-term (5 years or more) monitoring and to publish these results. We used Remote Underwater Video to monitor mobile assemblages over a 5 year period in Tianyar Bay, north Bali, Indonesia, in three habitat types: an artificial reef, degraded sand flat and natural coral reef to compare fish species richness and community structure between habitats over time. We found that (1) artificial reefs can support similar communities to coral reefs even after a short time (~ 1–3 years after deployment), (2) coral and artificial reef fish communities and trophic groups do not fully converge over a 5 year period, with the AR consistently supporting greater populations of predatory fish and the coral reef supporting greater populations of planktivorous and herbivorous fish, (3) both communities on the coral reef and artificial reef responded in a similar manner to localised environmental and ecological changes (e.g. recruitment, predator-prey interaction, environmental factors, unusual weather patterns) and/or external factors (such as fishing), and (4) the 5 year dataset allows us to draw different conclusions to what we would have after a 3 year dataset, much more commonly associated with research funding timescales. Ultimately, longer-term data sets are needed to make stronger conclusions on how artificial reefs can support similar communities to natural reefs. Given the continued, large-scale deployment of ARs as a habitat enhancement tool, collecting long-term data on their ecology should be prioritised if we are to better understand their effectiveness and full restoration potential.

Keywords Coral reef restoration · Artificial reefs · Fish community structure · Remote underwater video · Bali · Functional groups

Introduction

The coral reefs of Indonesia are recognised as a global conservation priority due to their high diversity and tourism value, however over 95% of Indonesian reefs are considered ‘threatened’ due to climate change and localised threats such as destructive fishing practices (e.g. dynamite fishing; Burke et al. 2012). In response, Indonesia has experienced a rapid growth in the establishment of reef restoration programmes(Razak et al. 2022) and now has more coral reef restoration programmes than any country in the world (Lamont et al. 2022). The construction of ARs is one of the most commonly utilised reef restoration tools in Bali, with hundreds of thousands units already deployed in

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degraded reefs on the coastlines of Bali's largest tourism hubs (Boakes et al. 2022a).

Ultimately, restoration of all reefs destroyed by climate change is impossible, but localised restoration such as AR deployment, may mimic many of the ecosystem services of coral reefs which local communities depend upon (Peixoto et al. 2024), especially services associated with fish diversity and abundance (e.g. supporting fisheries, reef-based tourism etc. (Chen et al. 2013; Ramos et al. 2019). However, there is a global lack of monitoring of these benefits by reef restoration programs (Eger et al. 2022). This has resulted in large knowledge gaps in understanding the effectiveness of these programmes and may have led to the same mistakes being repeated as lessons learned are not shared (Mills et al. 2017). More long-term monitoring is needed to understand how fish assemblages change on ARs over time (Hicks et al. 2025), as this will help to guide management decisions based on our understanding of how ARs can support key ecosystem services (Baine 2001).

In terms of the monitoring of ARs in a global context, Ries et al. (2021) discussed that long term (5 years or more), robust data sets on AR fish communities are rare, and the limited available mid-long term data sets particularly in terms of community structure and assemblages vary greatly in their results. For example, Mills et al. (2017) highlighted that ARs in south east Australia began to show community stability after just one year, whereas Hicks et al. (2025) found that a dynamic community on ARs In the Texas Gulf of Mexico stabilised after 5 years. Certain long-term studies have also aimed to assess how functional groups change on ARs over time, and the results for this also substantially vary between studies. For example, Cresson et al. (2019) observed that temperate ARs in the Mediterranean Sea experienced significant changes in fish functional groups over 6 years monitoring period, but despite this, the authors found no pattern of changes in terms of assemblages. In contrast Hicks et al. (2025) found that trophic structure on the ARs in their study was relatively stable, and suggested that functional groups may not reflect larger community shifts in ARs over time.

In the context of AR monitoring in Indonesia, only around 16% of coral restoration projects involve a post-installation monitoring framework, and even less publish on the long-term effectiveness of their reef restoration work (Razak et al. 2022). Consequently, despite the large amount of funding invested into artificial reef deployment programmes, there are still many unanswered questions in terms of how they perform, and if they are able to capture some of the key benefits of natural reefs which they aim to replace. In this study, we provide long-term (5 year) data on fish communities on three habitats (a natural coral reef, an artificial reef and a degraded sand habitat) in north Bali, to address this research gap and assess the potential of artificial reefs in mimicking coral reef fish diversity, community structure and trophic groups.

Methods

This research was conducted within a No-Take Zone (NTZ) Marine Protected Area (MPA) within Tianyar Bay, Karangasem Regency, north Bali, Indonesia (Fig. 1). The three habitat types (which were around 250 m apart) were survey. Following (Boakes 2024), these included:

1. **Coral Reef (CR)**: This site represented a relatively undisturbed coral reef, characterised by high biodiversity among both benthic and mobile species. A Reef Health Index (RHI) survey conducted at this site (following the methodology of Díaz-Pérez et al. (2016)—demonstrated that this coral reef had a RHI score of 4.5, categorized as "good" to "very good" (Boakes et al., *in review*). Local community members confirmed that this section of reef had not been subjected to past destructive activities, explaining its relatively pristine condition.
2. **Artificial Reef (AR)**: This site represented one of Indonesia's largest artificial reef deployment areas, which at the time of submission of this paper (May 2025), was made up of approximately 35,000 artificial reef structures. These structures were built by the local community using a three-part mixture of cement, calcium, and sand, forming units locally referred to as *roti buaya*. Each unit measured approximately 1 × 0.5 m and resembled a small table with a textured surface—including bumps, scratches, cracks, and crevices—to facilitate the natural recruitment of benthic species. ARs were arranged in clusters of 30 units, each covering about 10 m^2. Within each cluster, the structures were stacked in a similar yet random configuration to maximize habitat complexity, creating features such as holes, tunnels, and caves to provide shelter for fish.
3. **Sand Flat (SF)**: This site consisted of a sandy seabed with minimal hard substrate and a limited fish assemblages. According to local knowledge, this area was once a vibrant natural reef, which was destroyed decades ago due to anchor damage, coral harvesting, and destructive fishing methods. Over time, erosion and sedimentation buried the remnants of the reef beneath a layer of sand, resulting in the current "sand flat" state. This habitat was included in the study as a control site, as it represented the AR site before reef structures had been introduce, and therefore provides a baseline for evaluating ecological changes driven by AR deployment.

Fig. 1 Location of the three sampling sites (Sand Flat (8°11’24.0"S 115°29’34.4"E), Artificial Reef (AR; 8°11’26.3"S 115°29’42.2"E) and Coral reef (CR; 8°11’28.1"S 115°29’51.5"E)) within Tianyar Bay, Bali, Indonesia. Map copyright of OpenStreetMap contributors (openstreetmap.org/copyright) and Carto (https://carto.com/attribution)

In a similar study design to Vida et al. (2024), our research compared fish communities on the AR to those on adjacent CR and SF habitats. All habitat types were surveyed over a 5 year period between 2020 and 2024, during the months of April to September. Three sample sites (herein subsites – which were consistent for each habitat across the whole study period) were established in each of the three habitat types. For the SF and CR, sub-sites were chosen haphazardly (approximately 50 m apart from each other). For the AR, sub-sites (which contained 30 artificial structures of the same age) were chosen based on age of the structures, with AR1 = 1 year old, AR2 = 2 years old and AR = 3 years when monitoring started in 2020. On the AR, sub-sites were similar in terms of AR design, placement and number of structures. The monitoring then continued to survey the same sub sites over the 5 year period to allow comparison of reef communities over time. All habitats were surveyed within a similar depth range (6–9 m), following Jackson et al. (2012).

Remote Underwater Video (RUV) was used to survey mobile communities in all three habitat types in Tianyar Bay. Video samples were taken one by one using a GoPro Hero 9 HD 1080p underwater camera (fixed to a steel frame) on sampling days (of varying tidal conditions), only on calm mornings (between 8-10am) when underwater visibility was at least 15 m (measured using a visibility measuring line). Following Boakes et al. (2023a), RUV videos were recorded for a duration of 25 min, allowing for an initial 5 min settlement period and 20 min of analysis time (resulting in 90% species saturation levels – Boakes et al. (2022b). The cameras were deployed using by snorkellers/divers, who swam at least 100 m away whilst the camera was recording in order to decrease chance of influencing presence/absence of some fishes. Videos were taken twice from each subsite across the sampling season, giving a total of 18 RUV samples across all three habitat types per year (90 in total over the 5 year research period). Fish were identified to species level, allowing for the calculation of maximum number of species as a measure of species richness, and maximum number of individuals seen in any single frame (MaxN) as a measure of abundance (as per Hall et al. (2021).

A generalised two-way mixed model ANOVA (using a Poisson link function) was run for species richness as a dependent variable with habitat, year (and the interaction term) as fixed factors in the analysis and subsite as a random factor in the analysis to account for potential temporal autocorrelation in repeatedly sampling the same locations (diagnostic plots of residuals against fitted values indicate the model assumptions were appropriate for the data used). Differences between habitat types and years were examined using post-hoc tests (using uncorrected values in the emmeans package—(Lenth 2021). Then, two way

Permutational Multivariate Analysis Of Variance (PERMANOVA) was run (using square root transformed MaxN values for each species, in this case, as random factors cannot be included in PERMANOVA models the subsite was nested within habitat) to assess the difference in fish assemblages between habitat types, with respect to the interaction of habitat type and year (using the Vegan package in R, Bray Curtis similarity and 9999 permutations - Oksanen et al. (2023). Multiple comparisons were obtained from the pairwiseAdonis package in R (Martinez 2017; but effect sizes (provided by the R^2 values in the multiple comparisons), rather than significant differences were extracted for comparisons between habitats in the same year. Principal Coordinate Analysis (PCoA) was used to illustrate variation between habitat types and identify how fish communities change between habitats and years. Comparisons of different artificial reef ages over successive years were also tested using PERMANOVA, but by recoding the independent variable to include the reef age and year, hence each level had only two replicates based on the two recording periods within a monitoring season. Again, results were visualised using PCoA.

Analysis was also conducted based on functional categories to better understand if the use of each habitat type varied between functional groups. To avoid analysing rarely occurring species (many of which did not have trophic group information available), only commonly occurring fish species were included within the analysis. Fish species were included when their total number of individuals' made up >0.5% of total number of individuals' of all fish within the dataset. Based on this, 23 fish species were included, which made up 72.8% of total number of individuals of all fish within the dataset. Using the trophic guild classifications from Parravicini et al. (2020), these fish were categorized by six functional groups: (1) herbivores, (2) herbivores/detritivores, (3) omnivores, (4) omnivores/planktivores, (5) planktivores and (6) predators. PERMANOVA was performed to assess whether there was a significant interaction between habitat type and year for functional groups, using the exact same model that was run for species.

Table 1 Average fish species richness and total of number of observed species per habitat type

Habitat type	Average species richness per recording over the five year period	Total number of species observed over the five year period
Artificial reef	37.4 (SD: 8.3)	209
Coral reef	42.1 (SD: 10.1)	249
Sand flat	3.9 (SD: 2.1)	49

Results and discussion

Table 1 highlighted that CRs showed the significantly highest species richness (5 year mean: 42.1 fish species per recording, SD: 10.1), followed by a slight, but significant reduction on the artificial reefs (5 year mean: 37.4 fish species per recording, SD: 8.3), then a significant drop on the SF (5 year average: 3.9 fish species per recording, SD: 2.1). Yearly species richness was highlighted by Fig. 2. There was no significant difference in number of species in each subsite ($p = 0.240$), nested within the habitat grouping (Table 2). The number of species did not show a significant interaction between habitat or year ($p = 0.792$), but both fixed factors (species and year) were significant ($p < 0.001$ in both cases).

It is difficult to establish how fish assemblages on artificial reefs develop over time, as what appear to be either localised changes, or other stochastic effects affect all three communities studied. Community structure (addressed with two-way PERMAVOVA) showed significant interactions between habitat and year (PERMANOVA: $F_{8,69}$=2.35, $p < 0.001$), as well as a significant difference between subsites within habitats ($F_{6,69}$ = 1.62, $p = 0.033$). PCoA showed that similar communities existed on coral reefs and artificial reefs, notably in terms of Scaridae, Labridae and Acanthuridae. The greatest similarity was between 2023/24 on the artificial reef and 2021/22 on the coral reefs, with the coral reef community shifting downwards on the second axis of the PCoA in 2024 (Fig. 3). None the less, from 2020 to 2021 the AR to CR effect size dropped by 0.054, followed by fluctuations of less than 0.013 between 2021 and 2024, indicating greater stability in community between these habitats in the later 4 years of study (also highlighted visually in Fig. 4).

As highlighted by Table 2, comparison of the communities found on artificial reefs at different ages of reef over successive years also demonstrated significant differences (PERMANOVA $F_{14,29}$ = 2.36, $p < 0.001$), but with the biggest differences being from samples from 2020, compared to the other years. The age of the AR appeared to be less important in determining community structure than the year (Fig. 5). As such, it appears that 2020 was considerably different from other years in both the artificial reef and coral reef fish communities. In particular, neon damselfish (*Pomacentrus coelestis*), strongly associated with artificial reefs in 2020 (Fig. 5), were often replaced by bicolour chromis damselfish (*Stegastes partitus*) in later years. The later species becoming more closely associated with the reef structures, compared to the neon damselfish, which were found higher in the water column (Figure S2).

In terms of functional group analysis, we found a significant interaction between habitat type and fish functional group composition (F=2.49, d.f. = 6 and 69, $p < 0.001$).

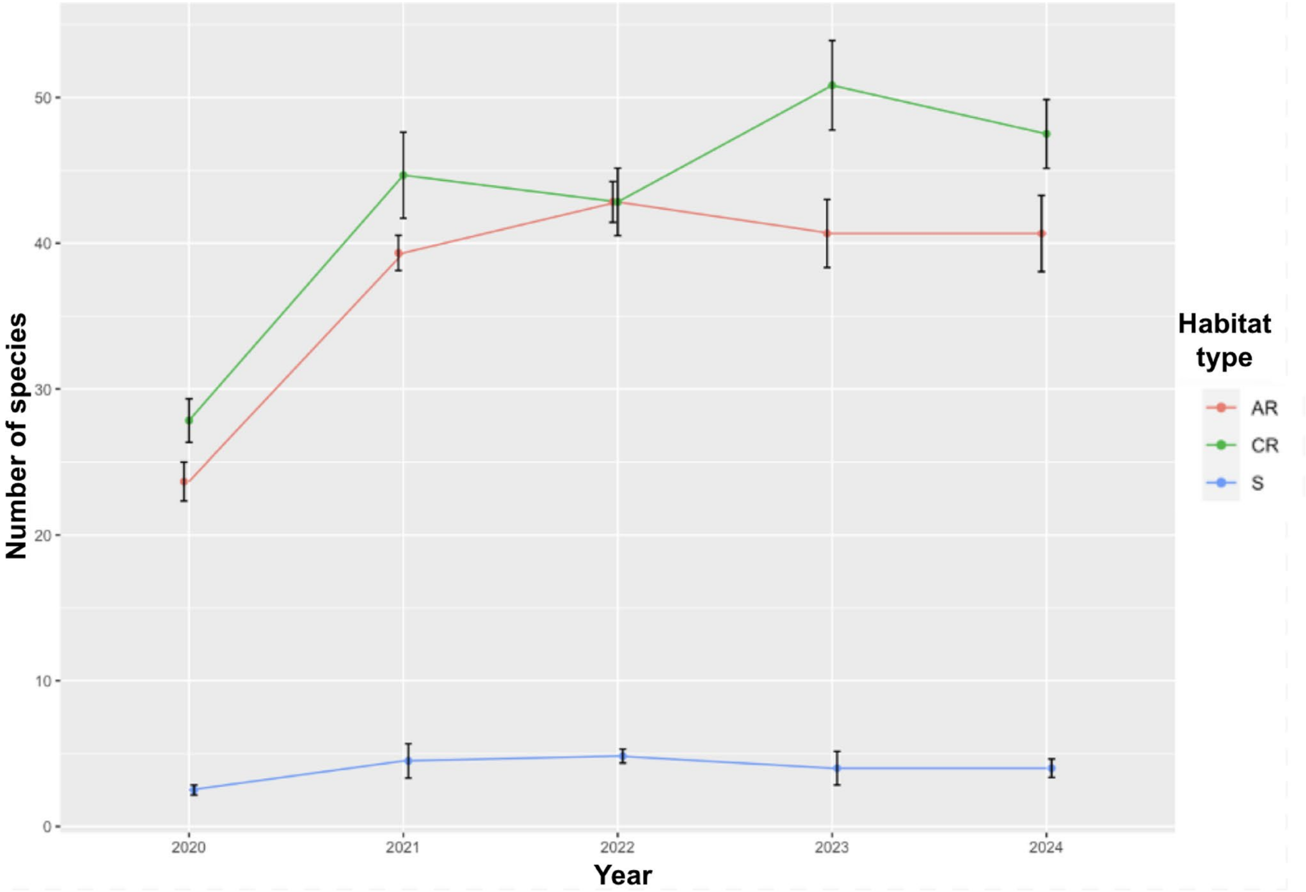

Fig. 2 Interaction plot (means +/-SE) of number of species in each habitat Artificial reef (AR), Coral reef (CR), degraded sand flat (SF) over the 5 year sampling period (2020–2024)

Table 2 (a) Two-way ANOVA (with Poisson error distribution) of the species richness of different habitats over time. (b) post-hoc tests for habitat, (c) post-hoc tests for year – note, the column 'Difference' in tables b and c is based on the Poisson transformed differences between factors, rather than data shown in Fig. 2

(a)	npar	Sum Sq	Mean Sq	F value	*p*
Habitat	2	616.62	308.31	308.31	< 0.001
Year	4	81.22	20.30	20.31	< 0.001
Habitat*Year	8	4.66	0.58	0.58	0.792
(b)	**Difference**	*p*			
AR ->CR	−0.13	0.0014			
AR ->S	2.24	< 0.001			
CR ->S	2.38	< 0.001			
(c)	**Difference**	*p*			
2020->2021	−0.523	< 0.001			
2020->2022	−0.561	< 0.001			
2020->2023	−0.538	< 0.001			
2020->2024	−0.515	< 0.001			
2021->2022	−0.038	0.698			
2021->2023	−0.015	0.884			
2021->2024	0.008	0.940			
2022->2023	0.023	0.817			
2022->2024	0.046	0.649			
2023->2024	0.023	0.829			

Functional groups were not evenly distributed across habitat types (Fig. 6), and there were notable difference (and significant) differences between the AR and CR, with predators (e.g. *Lutjanus rufolineatus*) being more abundant on the AR, and planktivores (e.g. *Chromis margaritifer*) and herbivores (e.g. *Siganus guttatus*) more abundant on the CR. Nonetheless, some overlaps did exist between the AR and CR, with relatively high numbers of planktivorous damselfish (e.g. *Abudefduf vaigiensis*) and predatory triggerfish (e.g. *Odonus niger*) in both habitat types. The sand habitats had limited fish communities other than *Heteroconger hassi* (planktivore), making the functional group analysis somewhat inconclusive for this habitat.

Here we demonstrate that artificial reef restoration of degraded habitats increases fish biodiversity and creates mobile fish communities similar, although statistically different to natural coral reefs. While many studies have identified enhanced biodiversity associated with artificial reefs, compared to areas lacking structural complexity (Leitao et al. 2008a), in this study we use data from five years of continuous monitoring, which demonstrates that: (1) artificial reefs can support similar communities to coral reefs

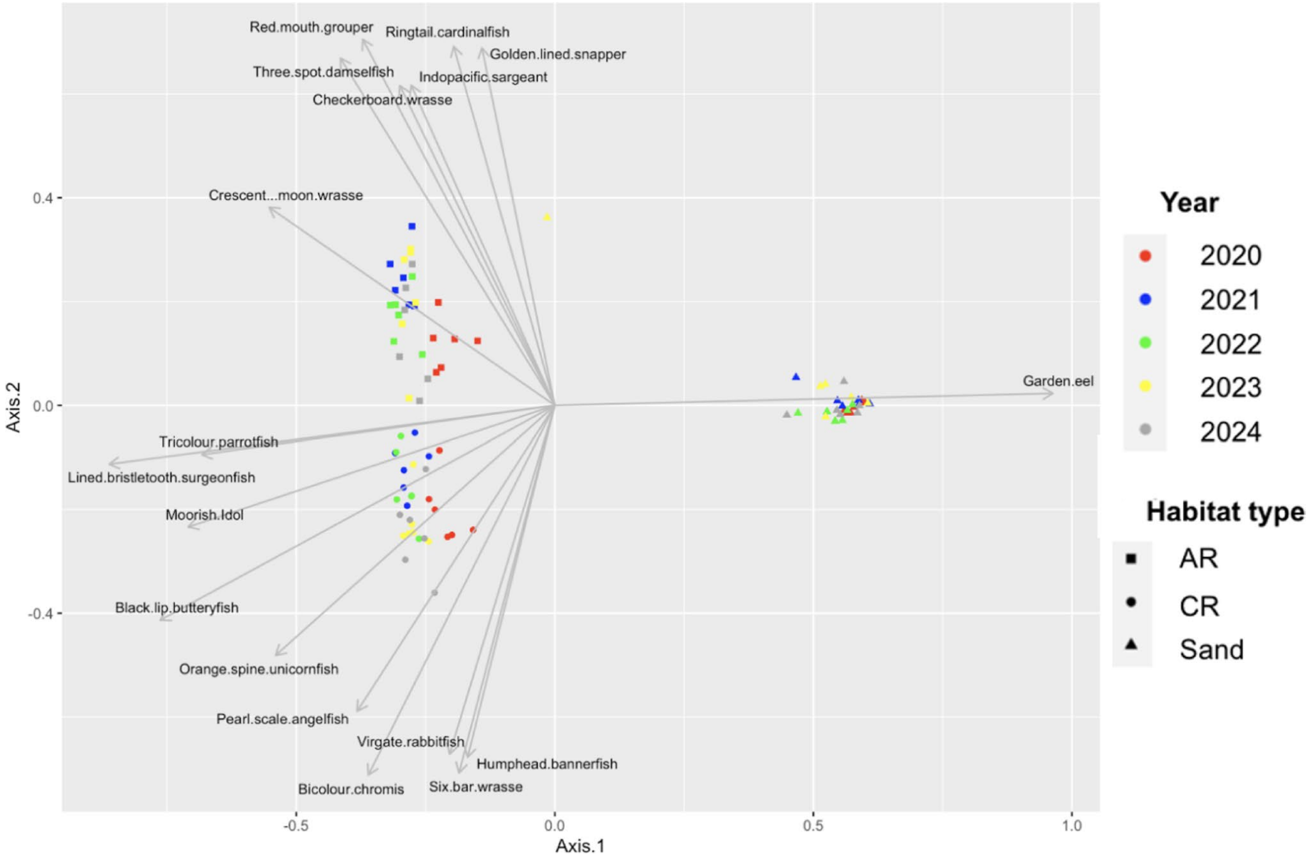

Fig. 3 Principal Coordinate Analysis (PCoA) plot for fish community structure between the artificial reef (AR), coral reef (CR) and degraded sand flat (SF) over a 5 year sampling period (with overlaid arrows based on $r > 0.45$ and $p < 0.001$)

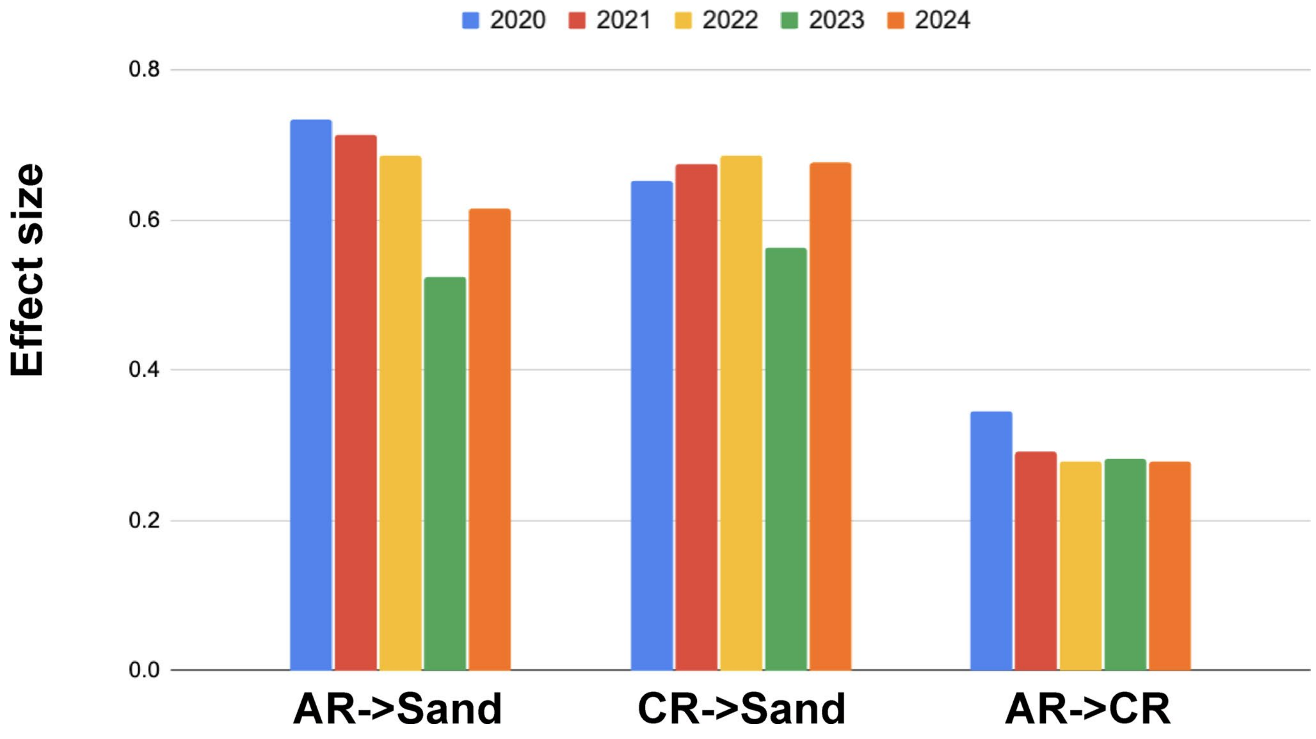

Fig. 4 Effect size (provided by the R^2 values in the multiple comparisons test) of difference in fish community structure over a 5 year sampling period. Note "AR" = artificial, reef, "Sand" = sand flat, and and "CR" = coral reef

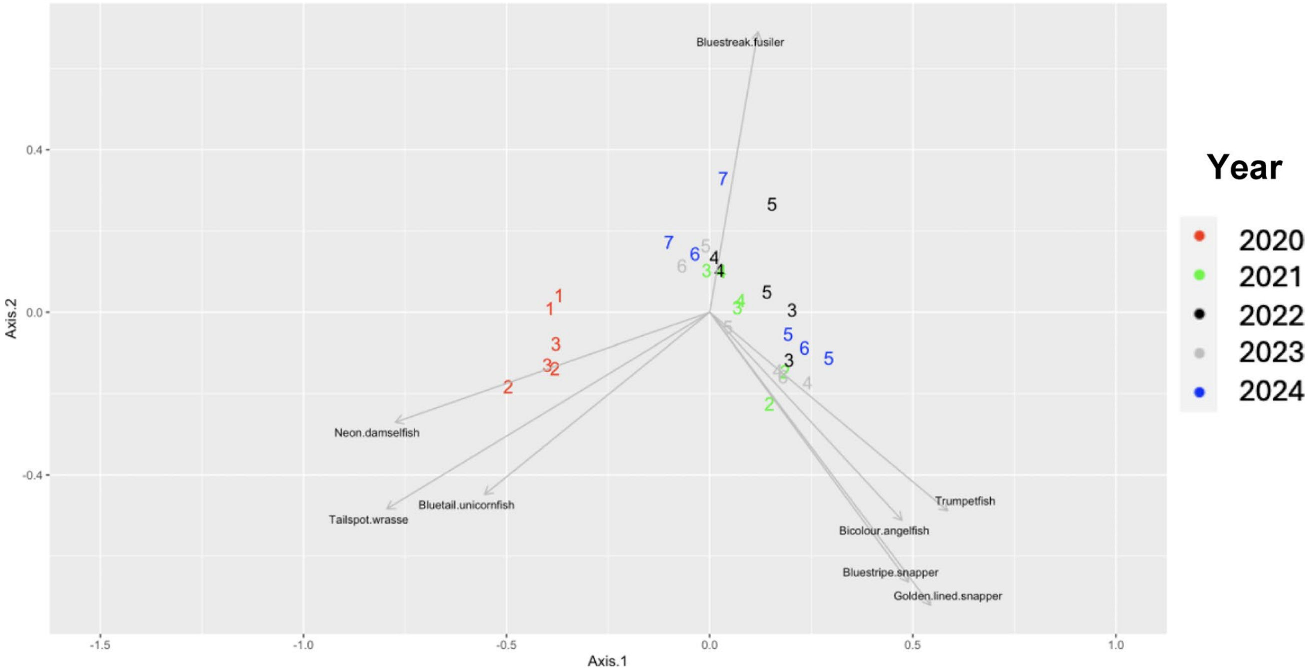

Fig. 5 Principal Coordinate Analysis (PCoA) plot highlighting the differences in fish communities in the artificial reef over a 5 year sampling period (with overlaid arrows based on $r > 0.45$ and $p < 0.001$). Note: point labels refer to AR age

even after a short time (~1–3 years after deployment), (2) coral and artificial reef fish communities and trophic groups do not fully converge over a 5 year period, with the AR consistently supporting greater populations of predatory fish and the coral reef supporting greater populations of planktivorous and herbivorous fish, (3) both communities on the coral reef and artificial reef responded in a similar manner to localised environmental and ecological changes (e.g. recruitment, predator-prey interaction, environmental factors, unusual weather patterns) and/or external factors (such as fishing), and (4) the five-year dataset allows us to draw different conclusions to what we would have after a three-year dataset, much more commonly associated with research funding timescales.

We found that in the first year of monitoring, the ARs (which ranged between 1 and 3 years after deployment) were already supporting a similar (although significantly lower) level of diversity and community structure to the adjacent natural coral reef. These findings add to a growing body of literature showing that, in terms of some ecosystem services such as tourism or fisheries – which are both dependant on healthy reef fish assemblages (Blanco-Pimentel et al. 2022; Martin et al. 2025), well-managed coral reef restoration in the tropics can generate near-immediate benefits for local communities who have lost reefs in the past (e.g. Boakes et al. 2023b; Peixoto et al. 2024). The species richness AR values in our study species were particularly high compared to many ARs around the world (e.g. Carr and Hixon (1997); Chou (1997), so it may be said that the value of the ARs in this study area are comparatively high.

Less widely discussed in the literature however, is the long-term convergence of AR and CR communities, with very few studies assessing the ability of ARs to mimic natural reefs in terms of the community structure they can support. Over the five year monitoring period, we observed that, despite showing some similarities and overlaps, community structure on the AR remained distinct and statistically different from the CR. We also found that species richness was typically slightly higher on the CR than the AR. The observed differences in fish assemblages and richness are likely explained by the differences in benthic communities between the two habitat types, with the CR having a higher species richness and abundance of specific species and/or colonies of benthic organisms, which had not yet established on the AR (Boakes et al. 2023a), such as certain species of wrasse (e.g. *Thalassoma hardwicke*) and rabbitfish (e.g. *Siganus virgatus*). These established benthic communities likely make it better suited to providing habitats to specialist fish species with more specific niches and/or benthic requirements. In the final two years of monitoring, communities on the CR and AR appeared to not be further converging, perhaps as a result of the benthic complexity. These findings are supported by previous studies which have highlighted that although fish assemblages on ARs can be abundant and biodiverse, they rarely mimic nearby natural reef communities (e.g. Hackradt et al. (2011), Folpp et al. (2013),Lowry et al. (2014). Artificial reefs also vary

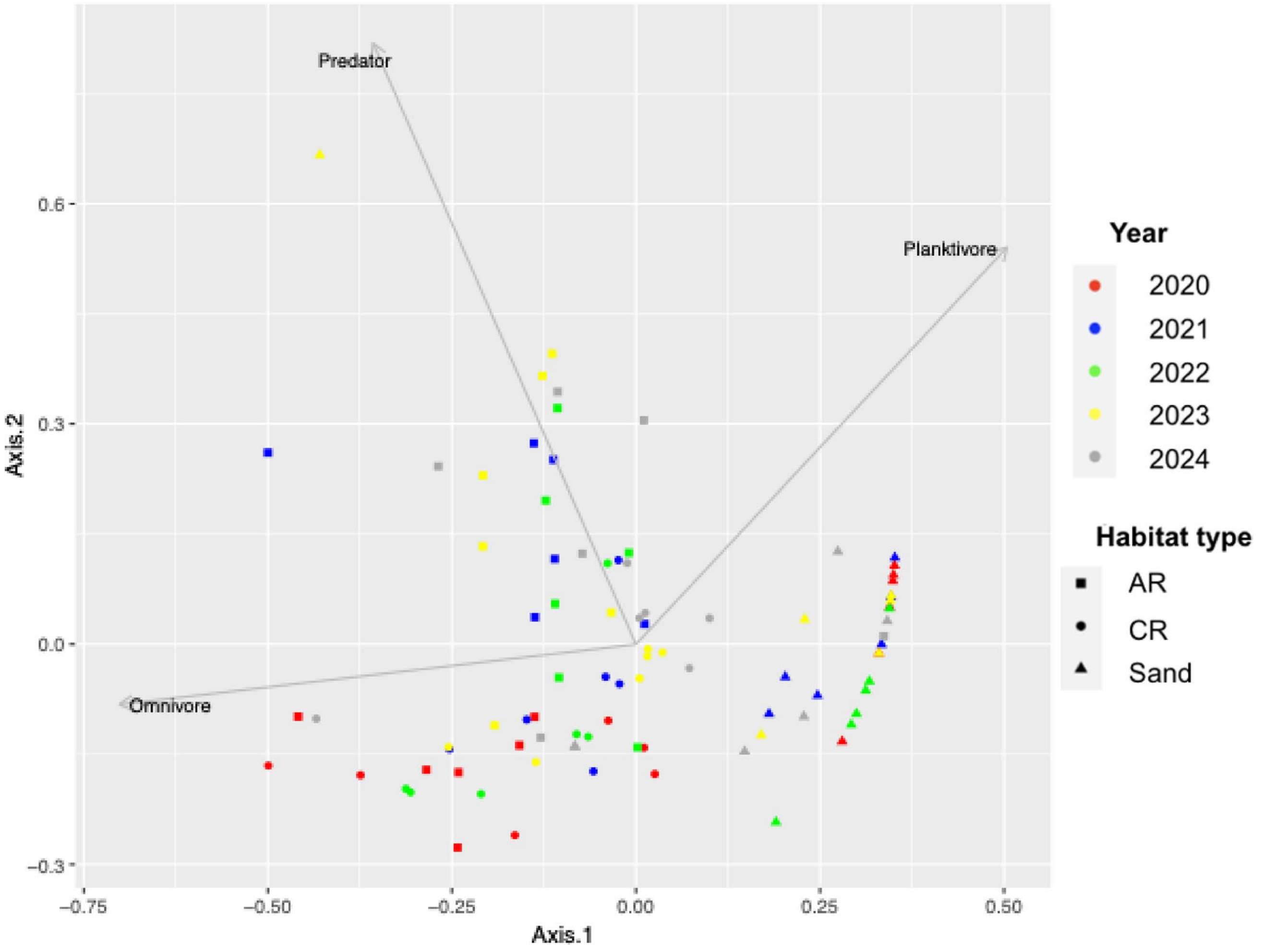

Fig. 6 Principal Coordinate Analysis (PCoA) plot for fish trophic groups between the artificial reef (AR), coral reef (CR) and degraded sand flat (SF) over a 5 year sampling period (with overlaid arrows based on $r > 0.45$ and $p < 0.001$)

greatly in design and their ability to replicate particular features of natural reefs, which has been demonstrated to also influence the extent to which they can support the same fish assemblages as natural reefs (Vivier et al. 2021).Longer-term data sets (10–20 years) are needed to confirm how successful ARs are at mimicking natural reefs, as communities may begin to converge further as benthic communities on the AR become more established (as demonstrated by Perkol-Finkel and Benayahu (2005) and Redman and Szedlmayer (2009).

Our data showed that there were several species which were present in all three habitats, such as *Lutjanus fulviflamma*. The habitats in this study were in relatively close proximity to each other (approximately 250 m apart) and it is possible certain individuals could have been moving around between all three habitats during the RUV recordings. We wish to acknowledge how the close proximity between habitat types does have potential to confound the results. Despite this, the results do show significant differences in community structure between each habitat type. It may be that any non-independence of species (due to movement from another habitat type) may have weakened the magnitude of difference between sites, however all habitat types were found to be significantly different from one another, so we expect that this influence is negligible. Furthermore, a large proportion of AR research projects, conduct their studies by installing reef units with research as their primary aim (such as Schroeter et al. (2015) - allowing for reef unit deployment plans to be designed based on an experimental perspective. Whilst these projects may have a solid experimental design, they are often relatively small scale (as research budgets often don’t allow for large scale AR deployment). Our study design was different, because we conducted our research on a ‘real-life’ coral restoration programme which deployed ARs on a large scale (one of the largest in Indonesia). This meant that there were experimental limitations in terms of local geography (the close proximity between habitat types), but did give this study the advantage of being able to collect data on a large scale AR deployment programme.

Leichter et al. (2003) discussed the influence that the deployment of artificial substrate can impact the supply of nutrients and larvae to nearby natural coral reefs. We found that in the final year of monitoring, the CR experienced a slight reduction in average species richness, whilst the AR remained the same. The predominant local current at the study location was from west to east, and therefore would have passed through the AR first. Therefore, the supply of nutrients and larvae would have been predominantly received (and potentially used-up) first by the AR, which could have influenced supply to the CR. This provides one possible explanation for the observed reduction in CR species richness in the final sampling year, and may be considered a potential negative impact caused by the deployment of artificial reefs.

The ability of ARs to mimic natural reefs in terms of the communities they support, are likely driven by the extent to which they can perform the same key functional processes as natural reefs (Boakes et al. 2023a). Whilst a list of coral reef functions can be found in Brandl et al. (2019), at present, very little is known about the extent to which ARs can support a similar level of functioning as natural reefs. Predation is a functional process which is important in structuring reef fish communities (Stewart and Jones 2001), with piscivorous fish playing an important role in controlling populations of low trophic level fish on ARs (Leitao et al. 2008b). Our functional analysis showed a significant difference between functional groups between the AR and CR, and found that predators were more abundant on the AR, while planktivores and herbivores were more abundant on the CR. The difference in functional groups between habitat types would indicate that the depth and breadth of functional processes which carried are out by fish (notably predation and herbivory) likely vary between the two habitat types. The greater amount of predators on the AR suggest that it may be able better support fisheries due to the high numbers of commercial fish supported (Díaz-Pérez et al. 2016), but the AR may be lacking in its ability to support other functional processes such as herbivory (McClanahan et al. 2002). We also found that neon damselfish (planktivore) were initially abundant on the AR, but were absent by year three. In contrast, bicolor chromis (also planktivore) damselfishes were initially absent on the AR, but were abundant by year three. The observed changes in population assemblages on the AR were likely driven by foraging behaviours of the damselfish, as well as the feeding preferences of their predators (Beukers-Stewart and Jones 2004), which over time likely contributed to the initial convergence of the communities between these two habitat types. Better understanding these functional processes, and the extent to which ARs can perform them to the same levels as natural reefs, will be key to managing and advancing coral reef restoration in future.

Our results showed that for all three habitats, fish species richness and community structure in the first year of sampling was significantly different to all other years. On the AR, this may be explained by increases in colonisation due to the provision of new habitats (as per Lowry et al. (2014), although some structures had been in place several years already. The same explanation cannot be given for the CR, which existed as a habitats long before data collection for this study began and still showed a large increase in species richness between 2020 and 2021. We propose that diversity and community structure was potentially driven by essentially stochastic localised effects (e.g. recruitment, predator-prey interaction, environmental factors, unusual weather patterns) and/or external factors (such as fishing), which may explain the outlying results in year 1. These effects, which appeared to influence communities on in all habitats, likely played an important role in determining artificial reef community structure and make it difficult to establish how fish assemblages on artificial reefs develop over time. This is supported by literature showing that fish assemblages on artificial reefs are influenced by environmental factors (such as unusual weather patterns - Godoy et al. (2002) and localised ecological processes (such as predator-prey interactions – Leitao et al. (2008c).

The method used in this study, Remote Underwater Video (RUV), is one of many options to assess fish assemblages. This specific method was chosen because it is a relatively cost-effective and safe option, compared to many of the others available (King et al. 2018), and is a suitable option for monitoring by restoration programmes with limited budgets and experience. RUV was selected due to the relatively low-level of disturbance caused, compared to roving diver surveys. This is because the stationary nature of the camera and frame minimally interferes with marine organisms, thereby enabling the observation of normal fish behaviour in the absence of diver-induced disturbances(Emslie et al. 2018). However, the use of RUV also has limitations; it was noticed that there were several small, cryptobenthic fish (e.g. Apogonidae) found on the AR and CR which resided within the protective space provided by the substrata, and these were not clearly identifiable from the RUV recordings. It is therefore likely that the abundance of certain species of small cryptobenthics were likely underestimated in our analysis as discussed in Boakes et al. (2023a), and future studies should consider additional options to survey these types of fish. Another limitation of RUV is the substantial amount of time taken to analyse the video footage, with one 20 min taking us up to 4 h to analyse. The time-consuming nature of its associated analysis, may discourage restoration programmes from using RUV, especially if budgets and staff time is limited. Currently, the use of artificial intelligence to speed-up RUV analysis is a developing field (Sheaves et al. 2020). While, to the best of the authors' knowledge, current technology remains inadequate for accurately analysing RUV videos, future advancements are likely to make such analysis more accessible to programmes with limited time.

Long term, robust data sets on AR fish communities continue to be rare (Reis et al. 2021b), and from those that do exist, there appears to be a range of results in terms of how long it takes for colonisation rates to stabilise, and how ARs can support similar communities and functions to nearby natural reefs. For example, in certain systems, ARs have been shown to support equivalent assemblages of fish as natural reefs (Lemoine et al. 2019), and in some cases have higher abundances of economically important fish species compared to natural reefs (Arena et al. 2007). In contrast, other studies have highlighted that AR communities will not converge with natural reefs, even over long time scales, unless they possess structural features similar to those of the natural surroundings (Perkol-Finkel et al. 2006). By comparing the results of this five year study with data collected at the three year interval, we would draw different conclusions. Both the species richness difference and community difference showed a higher degree of convergence in 2022, yet then demonstrated divergence in 2023 and 2024. Given typical research funding occurs over a 3 year period, this study clearly demonstrates the benefits of longer-term data. Ultimately, longer-term data sets (10–20 years +) are needed to make stronger conclusions, especially as we observe that external (or non-measured) factors, other than habitat type, are important in determining community structure. It is clear that benthic communities play an important role in shaping mobile assemblages and driving reef functioning on artificial reefs. Given the vulnerability of these communities to anthropogenic threats (notably climate changed-induced coral bleaching), it is likely that overall performance of tropical artificial reefs will continue to be influenced by external factors. As investment in artificial reef construction continues to increase globally, it is important that we continue to monitor and better understand these changes.

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Data and code availability The authors declare that the data supporting the findings of this study are available within the supplementary material (S3). Should the R code be required, it is available from the corresponding author upon reasonable request.

Declarations

Conflict of interest The authors declare no financial or non-financial conflict of interest.

Ethical approval This research for this article did not involve human participants. Nonetheless, ethical approval was obtained from Indonesia's National Research and Innovation Agency (BRIN). Ethics approval number: 08122024000004. All applicable international, national and/or institutional guidelines for sampling, care and experimental use of organisms for the study have been followed, and all necessary approvals were obtained. A foreign research permit was obtained from Indonesia's National Research and Innovation Agency (BRIN). Foreign research permit numbers: 548/SIP/IV/FR/7/2024 and 920/SIP/IV/FR/12/2024.

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