Journal of Oceanology and Limnology
Vol. 44, No. 3, pp. 1074-1086, 2026
https://doi.org/10.1007/s00343-025-5104-2

Research Paper

From rubble to reef: ecological transformation in volcanic rock-assisted coral restoration*

Xiangbo LIU[1, 3], Ruimei CHEN[2], Xiaoyu LIN[2, 3], He ZHAO[1, 3], Junling ZHANG[2, 3], Wentao ZHU[2, 3], Yinyin ZHOU[2, 3], Aimin WANG[2, 3], Xiubao LI[2, 3, **]

[1] *International Joint Research Center for Coral Reef Ecology of Hainan Province, School of Ecology, Hainan University, Haikou 570228, China*

[2] *Key Laboratory of Tropical Hydrobiology and Biotechnology of Hainan Province, School of Marine Biology and Fisheries, Hainan University, Haikou 570228, China*

[3] *Wenchang Advanced Fisheries Research Institute, Hainan University, Wenchang 571300, China*

Received Apr. 14, 2025; accepted in principle Jul. 12, 2025; accepted for publication Aug. 5, 2025

Abstract Coral reefs are increasingly degraded due to anthropogenic disturbance and climate change, transforming them into structurally unstable rubble fields that impede natural recovery and resilience. Under these conditions, the likelihood of coral reef natural recovery declines significantly in the absence of human intervention. In 2017, we launched a restoration project by introducing 8 t of natural volcanic rock into a rubble field near Wuzhizhou Island, Sanya, Hainan Island, China. Comprehensive surveys were conducted in 2019 and 2023 to assess the long-term impacts of this intervention and verify the effectiveness of volcanic rocks in rubble field restoration. By 2023, the coral cover had increased to 27.4% in the restoration area, compared to just 10.2% in the control area. Additionally, the recruitment density of juvenile corals reached 8.33 inds./m^2, which was nearly 7 inds./m^2 higher than in the control area. The restoration area supported 28 fish species in density of 135.37 inds./100 m^2, while the control area supported only 9 fish species in density of 20.5 inds./100 m^2. Similarly, macroinvertebrate density was higher in the restoration area (0.59 inds./m^2) than in the control area (0.29 inds./m^2). These findings suggest that volcanic rocks offer an effective in situ substrate for coral larvae recruitment, enhancing the habitat structural complexity of rubble fields and promoting the aggregation of fish and macroinvertebrates.

Keyword: coral reef; coral restoration; coral community; volcanic rock

1 INTRODUCTION

Coral reefs are among the most biologically diverse ecosystems on the Earth. Often described as the "tropical rainforests of the sea", coral reefs occupy less than 0.1% of the marine area but support nearly 25% of all marine species (Fisher et al., 2015; Boakes et al., 2022). In addition, coral reefs serve not only as natural coastal barriers protecting shorelines from waves and storms, but also contribute over \$1 trillion to the global economy through tourism and related industries (Moberg and Folke, 1999). In recent decades, climate change and anthropogenic disturbance have been regarded as dominant factors in coral degradation, with global coral cover showing a consistent annual decline (Hughes et al., 2017). The increasing frequency and intensity of these pressures make natural recovery increasingly difficult, as degraded reefs often transform into unstable rubble fields (Emslie et al., 2024). This situation underscores the urgent need for human intervention to restore this critical ecosystem and ensure the long-term survival of coral reefs and the services they provide. (Rinkevich, 2008).

In this context, coral reef restoration research has gained increasing attention, with various techniques

* Supported by the Innovative Talent Foundation of Hainan Province (No. KJRC2023C39) and the National Natural Science Foundation of China (Nos. 42161144006 or 3511/21, 42306124)

** Corresponding author: xiubaoli@hainanu.edu.cn

significantly improving the survival, growth, and three-dimensional structure development of transplanted corals. Current coral restoration research focuses on coral transplantation techniques, cost-effectiveness, asexual propagation of reef-building corals, coral nurseries (Mbije et al., 2010; Rachmilovitz and Rinkevich, 2017), outplanting methodologies (Yang et al., 2024), artificial reef design (Perkol-Finkel and Benayahu, 2004; Zheng et al., 2021), the use of heat-resistant coral genotypes (Bowden-Kerby, 2023), and habitat modification of degraded coral reefs (Rinkevich, 2019). The concept of coral gardening has introduced innovative approaches for cultivating coral colonies (Rinkevich, 2021b), and these methods have become increasingly widespread. Advances in coral nursery design and the widespread application of grid plates have improved the growth rates of transplanted corals (Xia et al., 2022; Liu et al., 2023). Outplanting, a key technique in the second phase of coral gardening, is now widely applied. However, fragmented substrates in degraded areas limit the applicability of some restoration methods. For instance, transplantation methods relying on nails are limited to hard substrates and are unsuitable for rubble fields (Huang et al., 2023).

Therefore, restoration techniques based on ecological engineering principles have been employed to modify substrate types in degraded reef areas, thereby facilitating coral reef recovery. These approaches are designed to promote ecological processes and restore or enhance ecosystem functioning, particularly through the stabilization and rehabilitation of unconsolidated rubble fields (Rinkevich, 2021a). For instance, the Mars Assisted Reef Restoration System (MARRS) is widely used in Indonesia to restore rubble fields damaged by dynamite fishing (Williams et al., 2019). Similarly, framed reef modules (FRMs) have been deployed to rehabilitate fragmented reef areas in the northern Wuzhizhou Island, China (Liu et al., 2024). A case study in Fiji reported that V-frames provided habitats for various invertebrates and served as substrates for recruitment in rubble fields (Bowden-Kerby, 2023). Another study demonstrated that natural rocks can also be used as restoration tools to stabilize rubble fields in Indonesia, attracting coral larvae and facilitating the recruitment of coralline algae (Yanovski and Abelson, 2019). In the Great Barrier Reef, a biodegradable material called the “reef bag” was utilized to stabilize loose rubble. After four years, the rubble piles become stable, primarily due to the natural binding effects of colonizing sponges and algae, which enhanced substrate cohesion (Kenyon et al., 2025). The application of these ecological engineering restoration tools has increased the proportion of hard substrates in rubble fields, promoting coral larval settlement, facilitating the formation of dense coral patches that provide fish habitats, and enhancing genetic diversity (Edwards et al., 2015; Cruz and Harrison, 2017; Fox et al., 2019).

The feasibility of using ecological engineering restoration methods to create stable substrates for long-term restoration has been well demonstrated. However, many restoration studies have focused primarily on indicators such as coral cover, growth rate, and three-dimensional structure, while paying limited attention to the long-term dynamics of biological community structure and biodiversity following restoration. Future coral restoration strategies should focus beyond coral survival rates to enhancing recruitment and establishment of reef-associated fauna at restoration sites (Horoszowski-Fridman et al., 2024). This shift will not only improve the ecological stability of degraded areas but also provide valuable insights for long-term ecological restoration efforts.

Volcanic rocks, abundant on Hainan Island, China, possess porous surfaces, low density and light weight, making them an ideal material for coral reef restoration. In 2017, natural volcanic rocks were employed as a substrate to cover the rubble fields, aiming to accelerate coral reef recovery in the northern area of Wuzhizhou Island and promote long-term ecological sustainability (Xia et al., 2020). This study focused on comparing changes in biodiversity and coral community composition in rubble fields undergoing volcanic rock restoration after six years. By analyzing data on substrate types, coral cover, juvenile coral recruitment, and community composition in both restored and control areas, we aimed to evaluate the long-term stability and ecological transformation of coral communities restored with volcanic rocks. The results may provide scientific evidence and valuable data for understanding shifts in biological communities within coral restoration efforts, potentially contributing to the refinement of future restoration strategies.

2 MATERIAL AND METHOD

2.1 Study site

Wuzhizhou Island, located in Haitang Bay, Sanya, Hainan Province, South China, is a well-known tourist destination famous for its coral reef resources. However, over the past few decades, coral reefs have suffered significant degradation due to coastal engineering and climate change. Coral cover sharply declined from approximately 80.0% in 2007 to 19.1% in 2017, resulting in the formation of rubble fields in the northern part of the island (Huang et al., 2020a). The substrate in these rubble fields mainly consists of gravel, sand, and dead coral skeletons overgrown with turf algae. These loose materials, affected by water flow, can cause coral larvae to suffocate and die upon settlement, thereby severely hindering their attachment (Kenyon et al., 2023).

In recent years, increased emphasis on environmental management and marine resource conservation by tourism companies and government agencies has led to more coral reef restoration projects. In August 2017, we deployed 8 t of volcanic rock (costing $2.5×10^4/hm^2$) in a rubble field in the northern area of Wuzhizhou Island (18.316 8°N, 109.761 8°E). The rocks were released from a boat and arranged underwater by scuba divers. The project covered approximately 232.7 m^2 at a water depth of 4–5 m (Fig.1). A control area was established at the same depth, approximately 10 m from the restoration site. A 50-m fixed transect was laid, with 1 m on each side designated as the control zone, totaling 100 m^2. The substrate in the control zone was identical to that in the restoration area before the volcanic rock deployment. Qualitative observations revealed that both study areas consisted of rubble fields with nearly no macroinvertebrate and fish populations.

Our previous research determined that the volcanic rocks used in the study ranged in diagonal length from 20 cm (minimum) to 70 cm (maximum) on any face, with a mean of 38.2 cm (Xia et al., 2020). These volcanic rocks were systematically arranged in a raised grid pattern within the designated restoration area (Fig 2). This study incorporated data from Xia et al. (2020) collected in 2019. The September 2023 survey focused on the changes in the coral reef community within the volcanic rock area and assessed the restoration effects on the rubble field.

2.2 Substrate-type investigation and temperature monitoring

Since 2019, a HOBO® temperature data logger (UA-002-08) has been deployed adjacent to the volcanic rock restoration area at the same depth to enable long-term, continuous temperature monitoring,

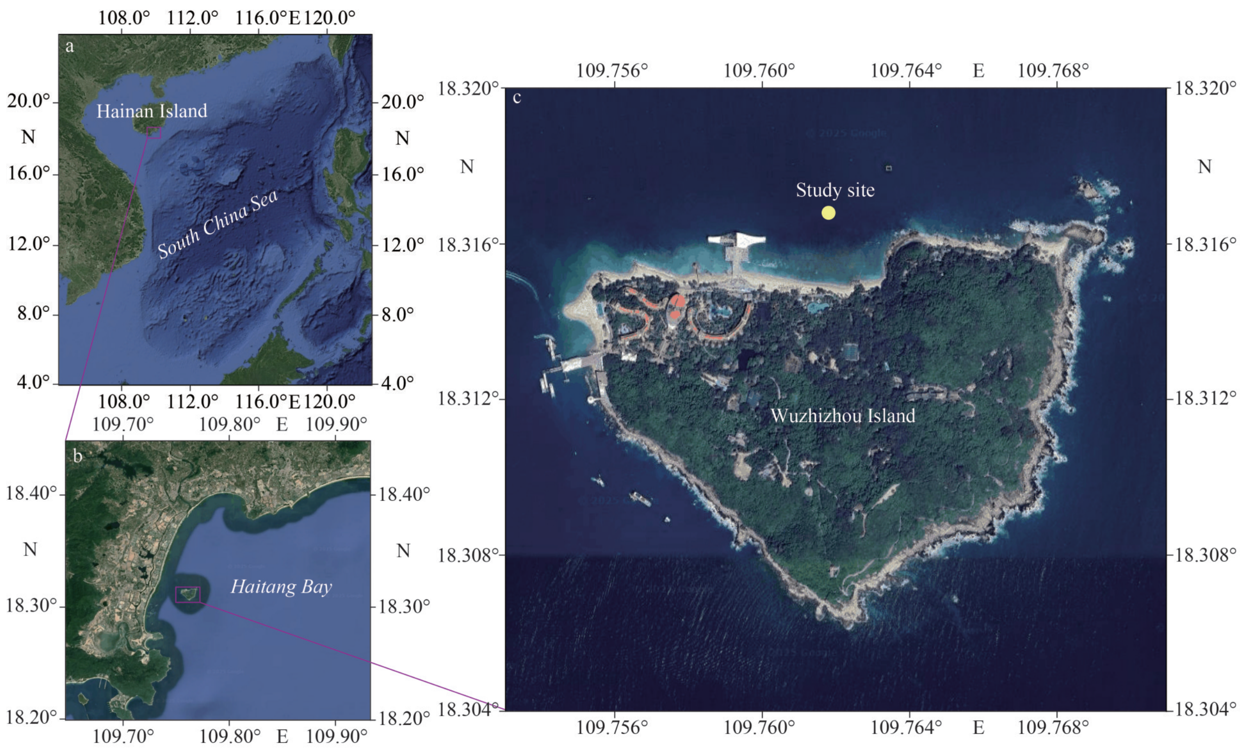

Fig.1 The study site in Hainan Island (a) and the close-ups of Haitang Bay (b) and Wuzhizhou Island (c)

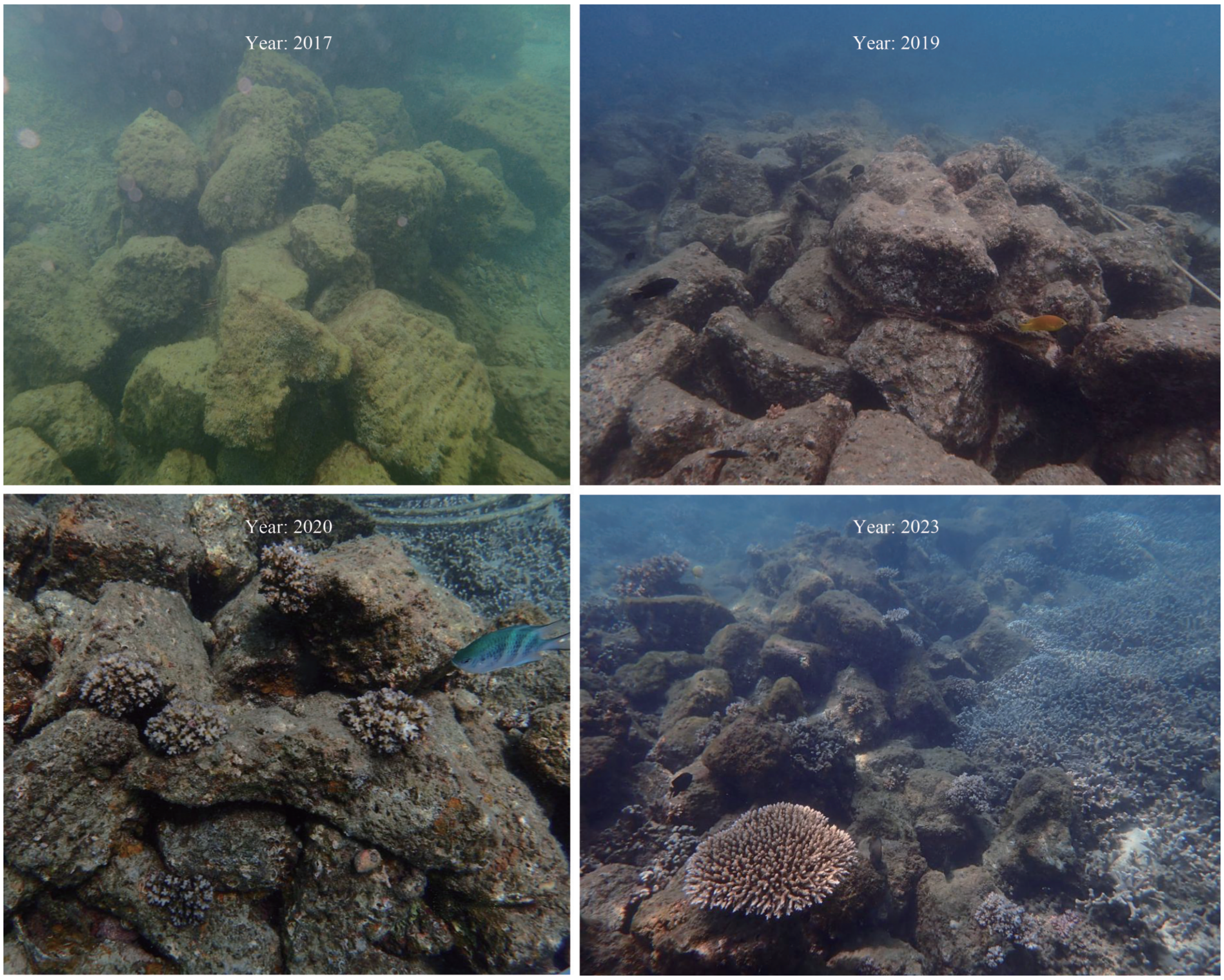

Fig.2 Status of volcanic rock substrate in 2017, 2019, 2020, and 2023

with data recorded every 15 min. In 2023, the line intercept transect method was employed to record the substrate types in the volcanic rock restoration area. Three transects were established, oriented approximately 120° apart to encompass the volcanic rock restoration area. Divers used GoPro 10 (GoPro Inc., USA) cameras to record the substrate types under the transects, and the video data were subsequently analyzed in the laboratory. Substrate types were documented at every 10-cm mark along the belt transect and categorized as Rock (>15 cm), Stony coral, Soft coral, Rubble, Zoanthids, Sand, Macroalgae, and others (Huang et al., 2020a). Coral species were identified based on Dai and Cheng (2020), Huang (2018), and the Corals of the World website (Veron et al., 2024), supplemented by photographs and field observations. Coral classification followed the standards outlined in the 2020 edition of "Classification and Identification of Reef-Building Corals in China" (Huang et al., 2020b).

2.3 Community composition and juvenile coral

2.3.1 Juvenile coral recruitment

Juvenile coral density (diameter 0.4–5 cm) was quantified in at least 48 randomly placed quadrats (40 cm×70 cm) across both the restoration and control areas. Juvenile coral density was calculated as the ratio of the total number of juveniles to the total quadrat area.

2.3.2 Community composition survey

Surveys of macroinvertebrate and fish community composition were conducted using video footage captured by divers with a GoPro 10 (GoPro Inc., USA) underwater camera (240 frames at 2.7-k resolution). Divers swam at approximately 5 m/min in a single direction within the volcanic rock restoration area, recording all observed macroinvertebrates and reef fish until returning to the starting point to prevent duplicate observations.

Due to the intricate three-dimensional structure of the volcanic rocks, only macroinvertebrates on the surface and within crevices were recorded to prevent disturbing the substrate and attached corals. The survey methodology for macroinvertebrates and fish followed Xia et al. (2022). To ensure data accuracy, photos and videos were validated in the laboratory for species identification and community composition analysis.

In the control area, divers swam above the belt transect using a GoPro10 underwater camera (240 frames at 2.7-k resolution, GoPro 10 (GoPro Inc., USA)) to record macroinvertebrates and coral reef fish that appeared within 1 m on either side of the transect. Subsequent video analysis in the laboratory documented the species and numbers of macroinvertebrates and reef fish, ensuring accurate and detailed data collection. Macroinvertebrates were categorized as sea cucumbers, sea urchins, sea snails, crinoids, and starfish, followed Huang and Lin (2012). Reef fish identification followed Fu (2013).

2.3.3 Diversity index

Dominant species in the survey area were identified using the dominance index (Y). Species diversity was evaluated using the species richness index (Margalef D), diversity index (Shannon-Wiener H), and evenness index (Pielou J), with calculation formulas following Xia et al. (2020).

2.4 Statistical analysis

The Before-After-Control-Impact (BACI) framework compares the "before" (pre-restoration) and "after" (post-restoration) conditions of the study area, as well as the "control" (control area) and "impact" (restoration site) areas during the same time periods. A two-way analysis ANOVA was conducted to test for variation in coral cover and juvenile coral density across periods (before, after) and treatments (impact vs. control). This approach follows the methodology of Adjeroud et al. (2016) and Osenberg et al. (1994). Results are presented as the mean±SD. Data analysis and visualization were performed using SPSS 4.0, Origin 2024, and Adobe Illustrator.

3 RESULT

3.1 Temperature monitoring and thermal disturbance

Wuzhizhou Island is located within the Qiongdong upwelling region, identified in studies as a potential thermal refuge for corals (Zhu et al., 2022). However, the exact duration of Qiongdong upwelling events remains unpredictable. Between 2020 and 2022, these upwelling events occurred earlier and were shorter than anticipated. During this period, two anomalous high-temperature events were recorded in the volcanic rock restoration area from August to October in both 2020 and 2022, with seawater temperatures exceeding 30 °C (Fig.3), surpassing the thermal tolerance of corals. Despite these events, no coral bleaching was observed in the volcanic rock restoration area. However, several studies have reported that these high-temperature events caused coral bleaching in parts of reef areas across the northern South China Sea and around Hainan Island (Feng et al., 2022; Lyu et al., 2022; Zhao et al., 2023).

3.2 Benthic composition survey

The benthic composition survey indicated that the substrate of the control and restoration area (before the volcanic rocks were deployed) were dominated by sand and rubble (>98%) in 2017, the coral cover in both the restoration and control areas was 0.22%. Six years after volcanic rock deployment, coral cover increased to 27.4% (Fig.4a), significantly higher than the 10.2% observed in the control area. In the restoration area, 21 species of stony corals and 3 species of soft corals were recorded (Supplementary Table S1). The dominant coral species was *Porites cylindrica*, covering 17.1%. In contrast, only 4 species of stony corals were

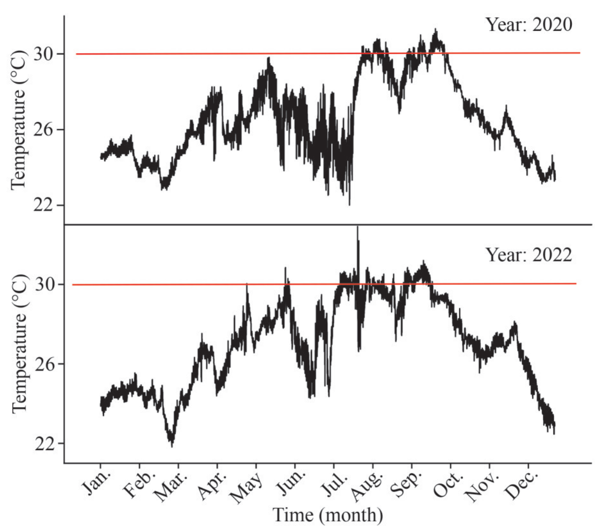

Fig.3 Two abnormally high temperatures in the restored area

recorded in the control area, *P. cylindrica* also being the dominant species, covering 8%. In both 2019 and 2023, only one species of macroalgae, *Hypnea pannosa*, was observed in both the restoration and control areas. In 2019, macroalgal cover was highest in the restoration area, reaching 1.67%, while it was 0.57% in the control area. By 2023, macroalgal cover decreased to 0.21% in the restoration area, compared to 0.6% in the control area. Two hydrocoral species (*Millepora platyphylla* and *Millepora dichotoma*) were documented in 2023. The benthic coverage of these hydrocorals was 2.17% in the restoration area versus 2.6% in the control area. In the restoration area, the overall substrate composition shifted from rubble and sand to predominantly stony coral and rock (79%) (Fig.4b). Additionally, soft coral coverage was 0.4%. While coral cover increased to 10.2% in the control area, the substrate remained predominantly rubble and sand, accounting for 86% of the total composition. Significant differences in coral cover were detected for period, control/impact stations, and their interaction (period×control/impact) in the two-way analysis of variance (Table 1), indicating that the intervention significantly altered coral cover in the restoration area.

3.3 Juvenile coral density

In 2017, before volcanic rock deployment, the density of juvenile corals was 0.15 inds./m^2 in the restoration area and 0.22 inds./m^2 in the control areas (Fig.5a). In 2023, six years after the deployment of volcanic rocks, the density of juvenile corals increased to 8.33 inds./m^2, significantly higher than the 1.34 inds./m^2 in the control area. The juvenile

Table 1 Results of the two-way analysis of variance (ANOVA) conducted to test for variation in the coral cover of among periods (before-2017, after-2023) and categories of stations (impact vs. control)

Parameter	df	*F*	*P*
Period	1	50.964	0.000
Control/impact	1	10.865	0.011
Period×control/impact	1	11.148	0.010

coral density of Pocilloporidae species decreased in 2023 compared to 2019 (Fig.5b). However, the total density of juvenile coral in the restoration area increased in 2023, with new recruits from the Poritidae, Euphylliidae, and Merulinidae families (Fig.6e–f). Soft coral recruitment was observed on volcanic rock (Fig.6c–d). Poritidae and Pocilloporidae comprising 86% of the total juvenile corals (Fig.6a–b). In particular, the Poritidae family exhibited the most significant increase, reaching a density of 3.03 inds./m^2. In contrast, the control area maintained low recruitment density and species quantity, predominantly consisting of *P. cylindrica* species. Additionally, significant differences were observed in period, control/impact stations, and period× control/impact for juvenile coral density in the two-way analysis of variance (Table 2).

3.4 Community investigation

Six years after the deployment, volcanic rock positively influenced the reef fish community in the restoration area. A total of 28 species of reef fish were recorded in the restoration area (Supplementary Table S2), with a density of 135.37 inds./100 m^2

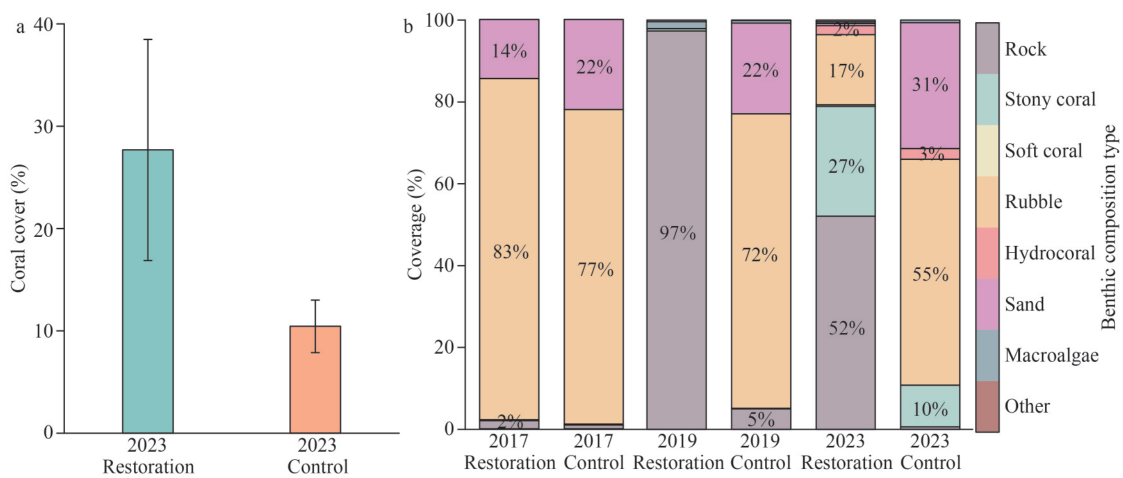

Fig.4 Coral cover difference and benthic composition coverage

a. coral coverage in 2023; b. composition of benthic composition. Error bars indicate ±SD in (a).

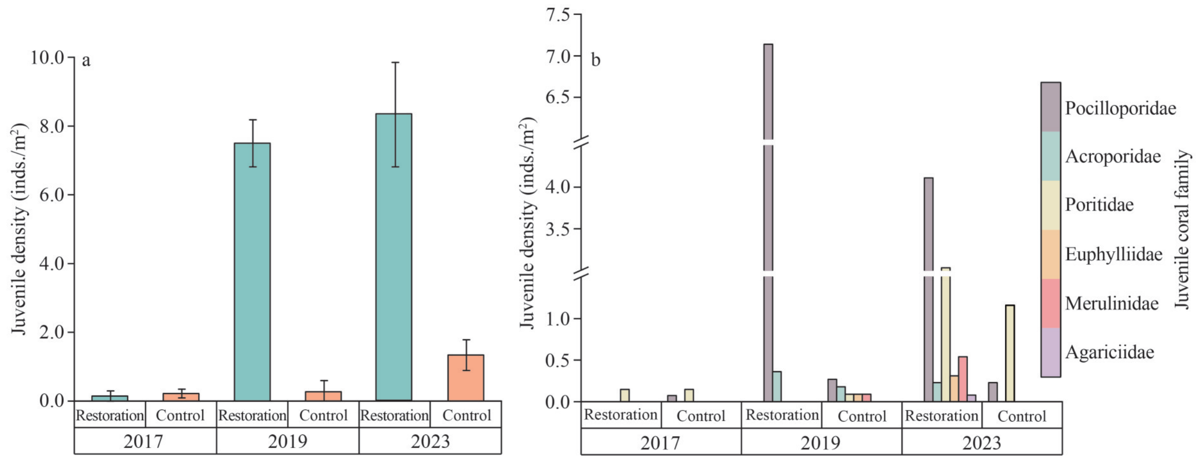

Fig.5 The density of juvenile coral recruitment

Error bars indicate ±SD in (a).

Table 2 Results of the two-way analysis of variance (ANOVA) conducted to test for variation in the juvenile coral density of among periods (before-2017, after-2023) and categories of stations (impact vs. control)

Parameter	df	*F*	*P*
Period	1	68.331	0.000
Control/impact	1	36.747	0.000
Period×control/impact	1	38.436	0.000

(Fig.7a), representing an increase of 14.61 inds./100 m^2 compared to 2019. In the control area, only 9 reef fish species were observed at a density of 20.5 inds./100 m^2. However, *Dascyllus reticulatus* (Pomacentridae) dominated the restoration area in 2023, accounting for 89.8% of the total fish and representing a 42.5% increase in Pomacentridae density since 2019. Other fish families, such as Labridae, experienced a decline in density.

Six years post-deployment, macroinvertebrate density in the restoration area (0.59 inds./m^2; Fig.7b) was higher than in the control area (0.29 inds./m^2). The dominant species included sea urchins and crinoids, with the sea urchin (*Diadema setosum*) comprising 60.6% of the total macroinvertebrate population (Supplementary Table S3), and *Comanthus gisleni* being the predominant crinoid species. Compared to 2019, a marked decline in macroinvertebrate density was observed in the restoration area, particularly among crinoids and sea urchins, which saw reductions of approximately 1.64 and 1.33 inds./m^2, respectively. However, an increase in species quantity was recorded, marked by the emergence of sea cucumbers (primarily, *Synapta maculata*), sea snails (primarily, *Turbo chrysostomus*), and starfish (primarily, *Echinaster luzonicus*). In contrast, the control area was primarily dominated by the sea cucumber (primarily, *Holothuria edulis*).

3.5 Dominance and diversity index

In 2023, *P. cylindrica* remained the dominant coral species in both the restoration area (*Y*: 0.108) and the control area (*Y*: 0.099). Coral biodiversity indices (Shannon-Wiener *H*, Margalef *D*, Pielou *J*) were higher in the restoration site compared to the control area (Fig.8a). For reef fish, species richness (*D*: 3.15) was significantly higher in the restoration area than in the control (*D*: 1.49). However, both diversity (*H*: 2.29) and evenness (*J*: 0.48) were lower in the restoration area than in the control (*H*: 2.86, *J*: 0.90) in 2023. For macroinvertebrates, the restoration area showed higher values in all biodiversity indices (*H*: 2.13, *J*: 0.59, *D*: 1.60) compared to the control area (*H*: 1.07, *J*: 0.54, *D*: 0.62) in 2023. Compared to 2019, species richness increased in both the restoration and control areas, although evenness declined.

4 DISCUSSION

The degradation of the coral reef's three-dimensional framework and substrate fragmentation, driven by anthropogenic impacts and climate change, poses significant challenges to coral restoration efforts (Loke et al., 2015). This study employed natural volcanic rocks to stabilize rubble fields and support the long-term coral restoration. The coarse and porous surfaces of volcanic rocks offer a suitable substrate for the settlement of coral

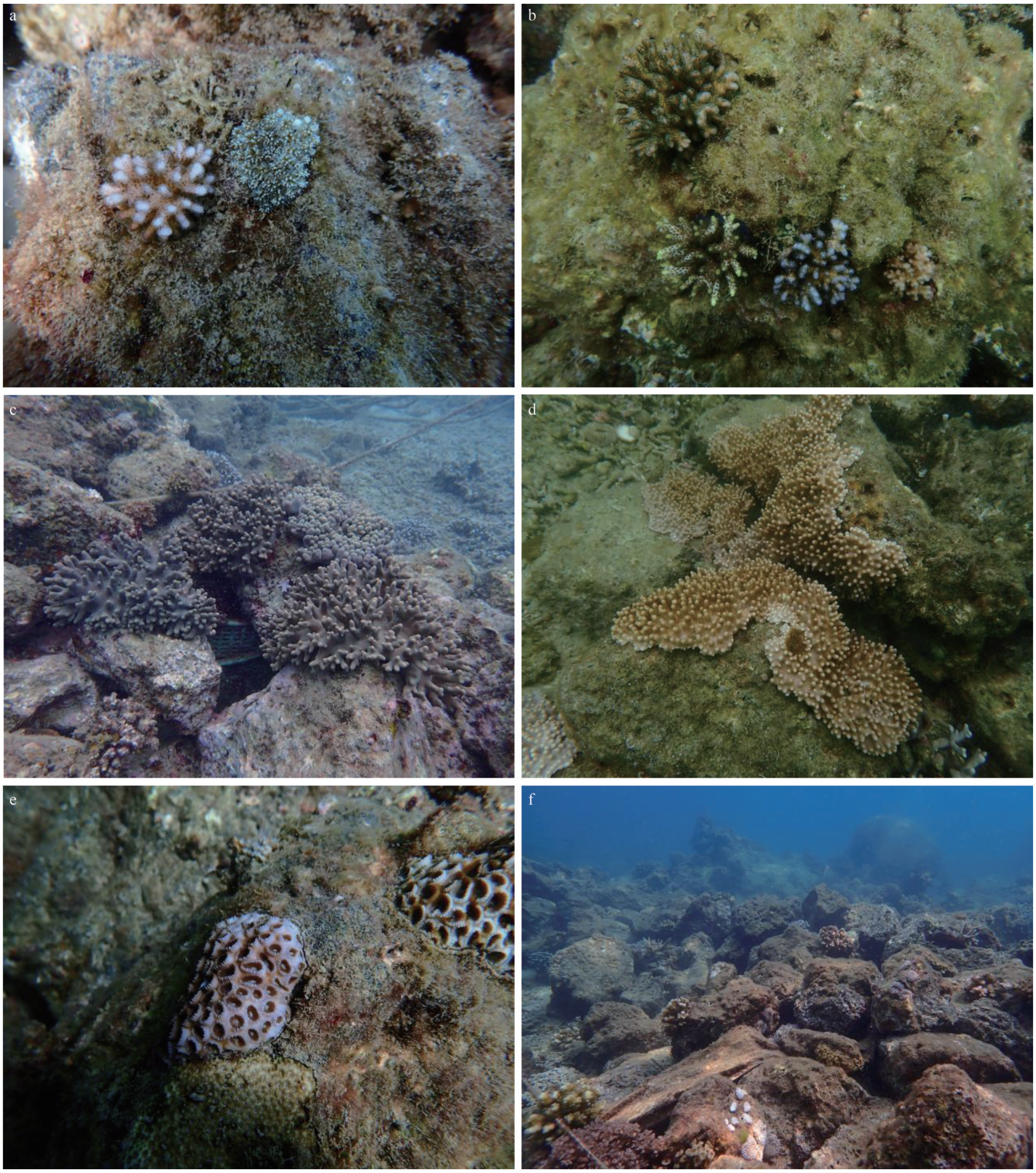

Fig.6 Coral recruitment in volcanic restoration area

a. juveniles of *Pocillopora damicornis* and *Galaxea fascicularis*; b. juveniles of *P. damicornis* and *Acropora*; c–d. soft corals attached; e. adults of Merulinidae; f. juveniles that attached to volcanic rock.

larvae. In addition to facilitating larval settlement, this restoration approach enhances habitat complexity, attracting reef fish and macroinvertebrates to colonize the degraded area. This illustrates that volcanic rocks serve as a simple yet effective long-term rehabilitation strategy for transforming ecological communities in rubble fields.

Six years after the deployment of volcanic rocks, stony coral cover increased to 27.4%, primarily due to coral larval recruitment and sustained growth across the entire restoration area. This corresponds to an average annual increase of approximately 4.6%. Although this result is slightly lower than the 6% annual growth rate of coral cover reported by

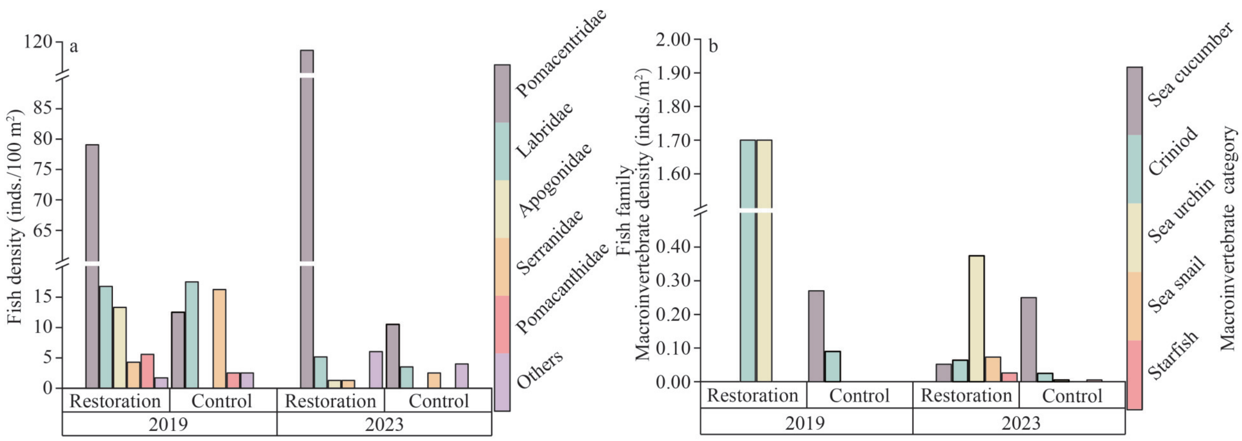

Fig.7 Fish density (a) and macroinvertebrate density (b)

Fox et al. (2019). Notably, a longer experimental duration can significantly increase the average annual coral cover, as both newly settled and previously attached coral larvae continue to grow and expand over time. Compared to restoration studies involving coral transplantation, natural coral recruitment in volcanic rock restoration areas was slower due to the longer sexual reproductive cycle of corals (Randall et al., 2020). However, the rate of coral cover increase in volcanic rock restoration area was approaching that of healthy coral reef. A coral transplantation study in the central Pacific of Mexico found that restoring coral cover from 5% to 20% after a thermal bleaching event took 20 years (Martínez-Castillo et al., 2023). Compared to the degradation of coral reefs caused by climate change, blast fishing activities have caused greater damage to the three-dimensional structure of coral reefs. Following blast fishing, restoring coral cover to 30%–60% in degraded areas primarily depends on deploying artificial reefs combined with coral transplantation, a process that typically requires at least 3 years (Zheng et al., 2021; Lange et al., 2024). Therefore, combining volcanic rocks and coral transplantation may increase the rate of coral cover expansion. However, this approach also involves higher restoration costs, necessitating a balance between ecological benefits and financial feasibility.

The effectiveness of the volcanic rock restoration method relies on a sufficient supply of coral larvae in the surrounding marine environments. Coral larval settlement is often constrained by fragmented and unstable substrates, which can result in settlement failure and hinder subsequent growth. To ensure the persistence and growth of corals in the restoration area, the coral larvae recruitment rate must match or exceed the coral mortality rate (van Woesik et al., 2014). Compared to the control area, volcanic rock surfaces significantly increased juvenile coral density, reaching 7.50 inds./m^2 after 2 years and 8.33 inds./m^2 after 6 years. These values exceed the natural larval recruitment rate of 3.75 inds./m^2 reported for Wuzhizhou Island and surpass the typical recruitment levels of <7 inds./m^2 observed in most coastal reefs around Hainan Island, including Sanya and Wenchang (Huang et al., 2020a; Lyu et al., 2024). These results demonstrate that volcanic rocks provide a favorable substrate for the settlement and growth of sexually produced coral larvae. Coral species such as *Porites cylindrica* and *Pocillopora damicornis* accounted for the majority of coral recruits, possibly due to their shorter reproductive cycles (Harii et al., 2002). Additionally, sexual reproduction allows coral larvae to acquire novel genotypes (Ritson-Williams et al., 2010), thereby enhancing genetic diversity and ultimately increasing the resilience of coral communities (Linden and Rinkevich, 2011).

Macroinvertebrates play a crucial role in sustaining the structure and functionality of coral reef ecosystems, and their density is often correlated with algal cover, biomass, and diversity in coral reefs (Burkepile and Hay, 2008). Volcanic rocks offered feeding and habitat areas for snails and sea urchins, which promoted their aggregation and reproduction. These macroinvertebrates help regulate the proliferation of algae (Villanueva et al., 2010), and this behavioral activity also provides space for coral larvae to settle (Edmunds and Carpenter, 2001). However, we observed a decrease in sea urchin density in 2023 (0.37 inds./m^2) compared to 2019 (1.7 inds./m^2). Nonetheless, sea urchin density in the volcanic rock area remained higher than the average density of 0.13 inds./m^2 on Wuzhizhou

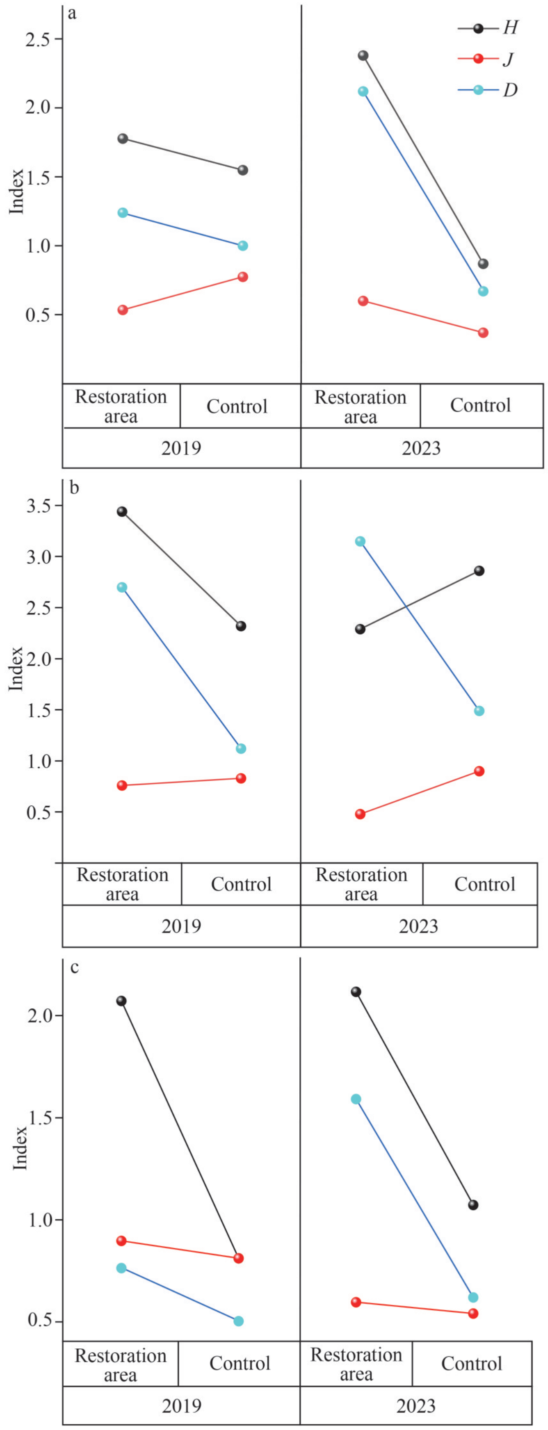

Fig.8 The diversity indices

a. coral diversity index; b. fish diversity index; c. macroinvertebrate diversity index. *H*: Shannon-Wiener index; *D*: Margalef index; *J*: Pielou index.

Island (Huang et al., 2020c). Although our results demonstrate the aggregation effect of volcanic rocks on large invertebrates, the accuracy of this finding is limited by monitoring methods, and we are unable to conduct a census of the organisms hiding in the depths and bottom of the crevices in the volcanic rocks. The complex three-dimensional structure of volcanic rocks creates favorable refuges and habitats for benthic organisms, with certain macroinvertebrates preferring dark, sheltered environments (Shiell and Knott, 2008). Currently, research on the influence of crinoids on coral reef ecosystems remains limited. Previous studies confirmed that crinoids filter plankton and suspended particulate matter from seawater, potentially reducing particle concentrations in restoration areas, although they are not considered a primary factor in ecosystem regulation (Birkeland, 1989). Sea cucumbers contribute to sediment cleaning and may help inhibit microbial pathogens in the restoration area. Research indicates that sea cucumbers can ingest toxic bacteria in sediments (Clements et al., 2024), reducing the prevalence of coral reef diseases and supporting reef ecosystem resilience (Grayson et al., 2022).

The introduction of volcanic rocks enhanced the structural complexity of the rubble field, facilitating coral larvae settlement and growth, and contributing to the development of a three-dimensional reef framework. This resulted in a significant increase in both the density and species richness of reef-associated fish (Richardson et al., 2018). Surveys conducted in 2019 and 2023 revealed that fish populations in the restoration area were predominantly dominated by the Pomacentridae species, with a significant imbalance compared to other fish species such as Labridae and Acanthuridae. This disparity may stem from the territorial dominance of Pomacentridae species. As highly territorial and aggressive competitors, these fish actively exclude other herbivorous fish and invertebrates to defend their territories, exhibiting strong survival strategies (Eurich et al., 2018). This territorial behavior likely contributes to the skewed fish community structure within the restoration area, reducing species evenness and overall fish biodiversity. However, the territorial behavior of Pomacentridae fish can also benefit coral communities, particularly in volcanic rock habitats, by safeguarding both adult and juvenile corals, promoting increased coral diversity (Schopmeyer and Lirman, 2015). Studies have demonstrated that Pomacentridae fish protect corals within their territories from coral-eating fish such as parrotfish thus promoting coral survival and growth (Eurich et al., 2018).

In coral reef ecological restoration projects, local threat factors of degraded coral reefs and site-specific restoration materials should be comprehensively evaluated to ensure practical ecological benefits and cost-effectiveness (Shumway et al., 2025). The use of volcanic rocks from Hainan Island costs only

\$2.5×10^4/hm^2, significantly lower than some artificial reef structures (>\$2.0×10^5/hm^2) and even cheaper than elevated frames constructed with reinforced steel (\$1.0×10^5–\$1.5×10^5/hm^2) (Williams et al., 2019; Yang et al., 2024; Patterson Edward et al., 2025). This cost-effectiveness is partly due to the method's lack of reliance on large-scale coral transplantation, reducing the need for costly professional divers. Additionally, volcanic rocks are inexpensive and widely available on some islands. Although coral recovery by the volcanic rock method is relatively slow, at least our research provides a simple and low-cost restoration method and proves that it effectively promotes community ecological transformation in the degraded area. Additionally, following long-term ecological transformation, volcanic rock communities may offer substantial conservation value. These restored sites could function as alternative diving locations, helping to meet tourism demands while alleviating anthropogenic pressure on adjacent healthy coral reef ecosystems (Firth et al., 2023).

5 CONCLUSION

In this study, we demonstrated that deploying locally sourced volcanic rocks to restore rubble fields is an effective and cost-effective approach for improving substrate stability, enhancing habitat complexity, and facilitating coral recruitment. Furthermore, it played a vital role in transitioning rubble fields to more stable and resilient coral reef ecosystems. The findings underscore the potential of volcanic rock restoration to improve coral diversity and overall ecosystem health. Future research should focus on evaluating the integration of volcanic rock restoration with other coral restoration techniques to optimize outcomes. This research can serve as a valuable reference for coral reef management and inform policy decisions aimed at preserving these ecosystems. In addition, long-term monitoring of volcanic rock restoration areas is essential for assessing their ecological and economic benefits to coral communities, which are often visible and appealing to stakeholders, particularly in regions or enterprises relying on revenues from ecotourism, such as diving tourism, and recreational fisheries.

6 DATA AVAILABILITY STATEMENT

The research data associated with this article can be obtained by contacting the corresponding author through formal request.

References

Adjeroud M, Gilbert A, Facon M et al. 2016. Localised and limited impact of a dredging operation on coral cover in the northwestern lagoon of New Caledonia. *Marine Pollution Bulletin*, **105**(1): 208-214, https://doi.org/10.1016/j.marpolbul.2016.02.028.

Birkeland C. 1989. The influence of echinoderms on coral-reef communities. CRC Press, America. p.1-79.

Boakes Z, Hall A E, Ampou E E et al. 2022. Coral reef conservation in Bali in light of international best practice, a literature review. *Journal for Nature Conservation*, **67**: 126190, https://doi.org/10.1016/j.jnc.2022.126190.

Bowden-Kerby A. 2023. Coral-focused climate change adaptation and restoration based on accelerating natural processes: launching the "Reefs of Hope" Paradigm. *Oceans*, **4**(1): 13-26, https://doi.org/10.3390/oceans4010002.

Burkepile D E, Hay M E. 2008. Herbivore species richness and feeding complementarity affect community structure and function on a coral reef. *Proceedings of the National Academy of Sciences of the United States of America*, **105**(42): 16201-16206, https://doi.org/10.1073/pnas.0801946105.

Clements C S, Pratte Z A, Stewart F J et al. 2024. Removal of detritivore sea cucumbers from reefs increases coral disease. *Nature Communications*, **15**: 1338, https://doi.org/10.1038/s41467-024-45730-0.

Cruz D W D, Harrison P L. 2017. Enhanced larval supply and recruitment can replenish reef corals on degraded reefs. *Scientific Reports*, **7**: 13985, https://doi.org/10.1038/s41598-017-14546-y.

Dai C F, Cheng Y R. 2020. Corals of Taiwan Vol. 1: Scleractinia fauna. Owl Press, Taibei, China. (in Chinese)

Edmunds P J, Carpenter R C. 2001. Recovery of *Diadema antillarum* reduces macroalgal cover and increases abundance of juvenile corals on a Caribbean reef. *Proceedings of the National Academy of Sciences of the United States of America*, **98**(9): 5067-5071, https://doi.org/10.1073/pnas.071524598.

Edwards A J, Guest J R, Heyward A J et al. 2015. Direct seeding of mass-cultured coral larvae is not an effective option for reef rehabilitation. *Marine Ecology Progress Series*, **525**: 105-116, https://doi.org/10.3354/meps11171.

Emslie M J, Logan M, Bray P et al. 2024. Increasing disturbance frequency undermines coral reef recovery. *Ecological Monographs*, **94**(3): e1619, https://doi.org/10.1002/ecm.1619.

Eurich J G, McCormick M I, Jones G P. 2018. Habitat selection and aggression as determinants of fine-scale partitioning of coral reef zones in a guild of territorial damselfishes. *Marine Ecology Progress Series*, **587**: 201-215, https://doi.org/10.3354/meps12458.

Feng Y T, Bethel B J, Dong C M et al. 2022. Marine heatwave events near Weizhou Island, Beibu Gulf in 2020 and their possible relations to coral bleaching. *Science of the Total Environment*, **823**: 153414, https://doi.org/10.1016/j.scitotenv.2022.153414.

Firth L B, Farnworth M, Fraser K P P et al. 2023. Make a difference: choose artificial reefs over natural reefs to compensate for the environmental impacts of dive tourism. *Science of the Total Environment*, **901**; 165488, https://doi.org/10.1016/j.scitotenv.2023.165488.

Fisher R, O Leary R A, Low-Choy S et al. 2015. Species richness on coral reefs and the pursuit of convergent global estimates. *Current Biology*, **25**(4): 500-505, https://doi.org/10.1016/j.cub.2014.12.022.

Fox H E, Harris J L, Darling E S et al. 2019. Rebuilding coral reefs: success (and failure) 16 years after low-cost, low-tech restoration. *Restoration Ecology*, **27**(4): 862-869, https://doi.org/10.1111/rec.12935.

Fu L. 2013. Coral Reef Fishes of the South China Sea: The Xisha, Nansha and Zhongsha Islands. China CITIC Press, Beijing, China. (in Chinese)

Grayson N, Clements C S, Towner A A et al. 2022. Did the historic overharvesting of sea cucumbers make coral more susceptible to pathogens? *Coral Reefs*, **41**(2): 447-453, https://doi.org/10.1007/s00338-022-02227-w.

Harii S, Kayanne H, Takigawa H et al. 2002. Larval survivorship, competency periods and settlement of two brooding corals, *Heliopora coerulea* and *Pocillopora damicornis*. *Marine Biology*, **141**(1): 39-46, https://doi.org/10.1007/s00227-002-0812-y.

Horoszowski-Fridman Y B, Izhaki I, Katz S M et al. 2024. Shifting reef restoration focus from coral survivorship to biodiversity using Reef Carpets. *Communications Biology*, **7**(1): 141, https://doi.org/10.1038/s42003-024-05831-4.

Huang D J, Xu Q, Li X B et al. 2020c. The community structure of echinoderms in sandy coral reef area in Wuzhizhou Island, Sanya, China. *Oceanologia et Limnologia Sinica*, **51**(1): 103-113, https://doi.org/10.11693/hyhz20190900174. (in Chinese with English abstract)

Huang H. 2018. Coral Reef Atlas of Xisha Islands. Science Press, Beijing, China. (in Chinese)

Huang J Z, Wang F X, Zhao H W et al. 2020a. Reef benthic composition and coral communities at the Wuzhizhou Island in the South China Sea: the impacts of anthropogenic disturbance. *Estuarine*, *Coastal and Shelf Science*, **243**: 106863, https://doi.org/10.1016/j.ecss.2020.106863.

Huang L T, Huang H, Jiang L. 2020b. A revised taxonomy for Chinese hermatypic corals. *Biodiversity Science*, **28**(4): 515-523. (in Chinese with English abstract)

Huang L T, Yu X L, Liu C Y et al. 2023. Actions to achieve rapid coral self-attachment: insights from outplanting nails, coral orientation, and substrate biological condition. *Restoration Ecology*, **31**(7): e13958, https://doi.org/10.1111/rec.13958.

Huang Z, Lin M. 2012. An Illustrated Guide to Species in China's Seas. Vol. 5. Animalia. China Ocean Press, Beijing, China. (in Chinese)

Hughes T P, Kerry J T, Álvarez-Noriega M et al. 2017. Global warming and recurrent mass bleaching of corals. *Nature*, **543**(7645): 373-377, https://doi.org/10.1038/nature21707.

Kenyon T M, Harris D, Baldock T et al. 2023. Mobilisation thresholds for coral rubble and consequences for windows of reef recovery. *Biogeosciences*, **20**(20): 4339-4357, https://doi.org/10.5194/bg-20-4339-2023.

Kenyon T M, Jones C, Rissik D et al. 2025. Bio-degradable 'reef bags' used for rubble stabilisation and their impact on rubble stability, binding, coral recruitment and fish occupancy. *Ecological Engineering*, **210**: 107433, https://doi.org/10.1016/j.ecoleng.2024.107433.

Lange I D, Razak T B, Perry C T et al. 2024. Coral restoration can drive rapid reef carbonate budget recovery. *Current Biology*, **34**(6): 1341-1348. e3, https://doi.org/10.1016/j.cub.2024.02.009.

Linden B, Rinkevich B. 2011. Creating stocks of young colonies from brooding coral larvae, amenable to active reef restoration. *Journal of Experimental Marine Biology and Ecology*, **398**(1-2): 40-46, https://doi.org/10.1016/j.jembe.2010.12.002.

Liu X B, Zhu W T, Chen R M et al. 2024. Framed reef modules: a new and cost-effective tool for coral restoration. *Restoration Ecology*, **32**(1): e13997, https://doi.org/10.1111/rec.13997.

Liu X B, Zhu W T, Xia J Q et al. 2023. Evaluation of the transplantation effect of artificial substrates with different apertures on *Acropora microphthalma*. *Journal of Tropical Biology*, **14**(5): 536-544, https://doi.org/10.15886/j.cnki.rdswxb.20220007. (in Chinese with English abstract)

Loke L H L, Ladle R J, Bouma T J et al. 2015. Creating complex habitats for restoration and reconciliation. *Ecological Engineering*, **77**: 307-313, https://doi.org/10.1016/j.ecoleng.2015.01.037.

Lyu Y H, Wang W N, Zhou Z H et al. 2024. Evaluation of the ecological status of shallow-water coral reefs in China using a novel method and identification of environmental factors for coral decline. *Marine Pollution Bulletin*, **201**: 116227, https://doi.org/10.1016/j.marpolbul.2024.116227.

Lyu Y H, Zhou Z H, Zhang Y M et al. 2022. The mass coral bleaching event of inshore corals form South China Sea witnessed in 2020: insight into the causes, process and consequence. *Coral Reefs*, **41**(5): 1351-1364, https://doi.org/10.1007/s00338-022-02284-1.

Martínez-Castillo V, Rodríguez-Troncoso A P, Tortolero-Langarica J A et al. 2023. Active restoration efforts in the Central Mexican Pacific as a strategy for coral reef recovery. *Revista de Biología Tropical*, **71**(S1): e54795, https://doi.org/10.15517/rev.biol.trop..v71is1.54795.

Mbije N E J, Spanier E, Rinkevich B. 2010. Testing the first phase of the 'gardening concept' as an applicable tool in restoring denuded reefs in Tanzania. *Ecological Engineering*, **36**(5): 713-721, https://doi.org/10.1016/j.ecoleng.2009.12.018.

Moberg F, Folke C. 1999. Ecological goods and services of coral reef ecosystems. *Ecological Economics*, **29**(2): 215-233, https://doi.org/10.1016/S0921-8009(99)00009-9.

Osenberg C W, Schmitt R J, Holbrook S J et al. 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. *Ecological Applications*, **4**(1): 16-30, https://doi.org/10.2307/1942111.

Patterson Edward J K, Mathews G, Diraviya Raj K et al. 2025. Long term coral restoration efforts to mitigate anthropogenic and climatic impacts in Gulf of Mannar, India: lessons learnt, success, challenges and prospects.

Journal of Environmental Management, **391**: 126377, https://doi.org/10.1016/j.jenvman.2025.126377.

Perkol-Finkel S, Benayahu Y. 2004. Community structure of stony and soft corals on vertical unplanned artificial reefs in Eilat (Red Sea): comparison to natural reefs. *Coral Reefs*, **23**(2): 195-205, https://doi.org/10.1007/s00338-004-0384-z.

Rachmilovitz E N, Rinkevich B. 2017. Tiling the reef—exploring the first step of an ecological engineering tool that may promote phase-shift reversals in coral reefs. *Ecological Engineering*, **105**: 150-161, https://doi.org/10.1016/j.ecoleng.2017.04.038.

Randall C J, Negri A P, Quigley K M et al. 2020. Sexual production of corals for reef restoration in the Anthropocene. *Marine Ecology Progress Series*, **635**: 203-232, https://doi.org/10.3354/meps13206.

Richardson L E, Graham N A J, Pratchett M S et al. 2018. Mass coral bleaching causes biotic homogenization of reef fish assemblages. *Global Change Biology*, **24**(7): 3117-3129, https://doi.org/10.1111/gcb.14119.

Rinkevich B. 2008. Management of coral reefs: we have gone wrong when neglecting active reef restoration. *Marine Pollution Bulletin*, **56**(11): 1821-1824, https://doi.org/10.1016/j.marpolbul.2008.08.014.

Rinkevich B. 2019. The active reef restoration toolbox is a vehicle for coral resilience and adaptation in a changing world. *Journal of Marine Science and Engineering*, **7**(7): 201, https://doi.org/10.3390/jmse7070201.

Rinkevich B. 2021a. Ecological engineering approaches in coral reef restoration. *ICES Journal of Marine Science*, **78**(1): 410-420, https://doi.org/10.1093/icesjms/fsaa022.

Rinkevich B. 2021b. Augmenting coral adaptation to climate change via coral gardening (the nursery phase). *Journal of Environmental Management*, **291**: 112727, https://doi.org/10.1016/j.jenvman.2021.112727.

Ritson-Williams R, Paul V J, Arnold S N et al. 2010. Larval settlement preferences and post-settlement survival of the threatened Caribbean corals *Acropora palmata* and *A. cervicornis*. *Coral Reefs*, **29**(1): 71-81, https://doi.org/10.1007/s00338-009-0555-z.

Schopmeyer S A, Lirman D. 2015. Occupation dynamics and impacts of damselfish territoriality on recovering populations of the threatened staghorn coral, *Acropora cervicornis*. *PLoS One*, **10**(11): e0141302, https://doi.org/10.1371/journal.pone.0141302.

Shiell G R, Knott B. 2008. Diurnal observations of sheltering behaviour in the coral reef sea cucumber *Holothuria whitmaei*. *Fisheries Research*, **91**(1): 112-117, https://doi.org/10.1016/j.fishres.2007.12.010.

Shumway N, Foster R, Fidelman P. 2025. The governance of marine and coral reef restoration, lessons and paths forward for novel interventions. *Environmental Science & Policy*, **164**: 103999, https://doi.org/10.1016/j.envsci.2025.103999.

van Woesik R, Scott W J, Aronson R B. 2014. Lost opportunities: coral recruitment does not translate to reef recovery in the Florida Keys. *Marine Pollution Bulletin*, **88**(1-2): 110-117, https://doi.org/10.1016/j.marpolbul.2014.09.017.

Veron J E N, Stafford-Smith M G, E T et al. 2024. Corals of the World. Version 0.01 Beta, http://coralsoftheworld.org/. Accessed on 2024-03-20.

Villanueva R D, Edwards A J, Bell J D. 2010. Enhancement of grazing gastropod populations as a coral reef restoration tool: predation effects and related applied implications. *Restoration Ecology*, **18**(6): 803-809, https://doi.org/10.1111/j.1526-100X.2010.00742.x.

Williams S L, Sur C, Janetski N et al. 2019. Large-scale coral reef rehabilitation after blast fishing in Indonesia. *Restoration Ecology*, **27**(2): 447-456, https://doi.org/10.1111/rec.12866.

Xia J Q, Jia Z Y, Zhang G H et al. 2020. Study on effect of basalt on restoration of damaged coral reef. *Journal of Zhejiang Ocean University* (*Natural Science*), **39**(3): 237-244, https://doi.org/10.3969/j.issn.1008-830X.2020.03.007. (in Chinese with English abstract)

Xia J Q, Zhu W T, Liu X B et al. 2022. The effect of two types of grid transplantation on coral growth and the in-situ ecological restoration in a fragmented reef of the South China Sea. *Ecological Engineering*, **177**: 106558, https://doi.org/10.1016/j.ecoleng.2022.106558.

Yang B, Zheng H N, Cui Z P et al. 2024. Restoring degraded coral colony using two coral transplantation techniques: a case study from Dapeng Bay, Shenzhen, China. *Regional Studies in Marine Science*, **69**: 103289, https://doi.org/10.1016/j.rsma.2023.103289.

Yanovski R, Abelson A. 2019. Structural complexity enhancement as a potential coral-reef restoration tool. *Ecological Engineering*, **132**: 87-93, https://doi.org/10.1016/j.ecoleng.2019.04.007.

Zhao Y, Chen M R, Chung T H et al. 2023. The 2022 summer marine heatwaves and coral bleaching in China's Greater Bay Area. *Marine Environmental Research*, **189**: 106044, https://doi.org/10.1016/j.marenvres.2023.106044.

Zheng X Q, Li Y C, Liang J L et al. 2021. Performance of ecological restoration in an impaired coral reef in the Wuzhizhou Island, Sanya, China. *Journal of Oceanology and Limnology*, **39**(1): 135-147, https://doi.org/10.1007/s00343-020-9253-z.

Zhu W T, Ren Y X, Liu X B et al. 2022. The impact of coastal upwelling on coral reef ecosystem under anthropogenic influence: coral reef community and its response to environmental factors. *Frontiers in Marine Science*, **9**: 888888, https://doi.org/10.3389/fmars.2022.888888.

Electronic supplementary material

Supplementary material (Supplementary Tables S1–S3) is available in the online version of this article at https://doi.org/10.1007/s00343-025-5104-2.