TITLE: Long-term outcomes of passive and active restoration approaches following a vessel grounding in Hawaii, USA

RUNNING TITLE: Restoration monitoring of vessel grounding

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Abstract

In February 2010, the cargo vessel *M/V Vogetrader* ran aground on a forereef in Oahu, Hawaii. Baseline surveys documented considerable damage to coral communities. Several restoration actions were implemented in 2013, including active restoration (rubble removal, coral outplanting) and passive restoration (natural recovery), with the goal of returning corals to their pre-disturbance state. In 2022, repeated surveys were conducted across three injury zones that varied in the severity of impact and the restoration actions employed to provide a rare assessment of restoration outcomes a decade post-grounding. We found coral recovery to be contingent on the severity of impact and the quality of the impacted habitat, not the amount of active restoration. Despite rubble removal efforts, present-day rubble cover was significantly higher at the impact sites compared to the reference sites and appeared to constrain recovery in the injury zone where grounding impacts destabilized the reef framework. Outplant efforts did not increase coral density or mean size relative to natural recovery sites, though this may be the result of an ineffective outplant design rather than failed outplanting as a whole. The sites closest to returning to a pre-disturbance state were the passive restoration sites. This, however, likely reflects the low severity of grounding impacts and the marginal (e.g., small and sparse) population of corals at these sites. These findings suggest that the extent of active restoration actions should be carefully and intentionally scaled to the severity and spatial extent of impact (with greater impacted areas receiving greater amounts of restoration), and that with sufficient time, marginal reef habitats with a low impact severity can likely recover from passive restoration alone.

Introduction

Coral reef restoration techniques have grown exponentially in recent decades (reviewed by Lirman and Schopmeyer 2016) in response to the continued global decline in coral reefs (Hughes et al. 2017; Hoegh-Guldberg et al. 2017; Spalding et al. 2017). These techniques include both passive and active approaches. Passive restoration involves removing the underlying disturbance or stressors, such as blast fishing or land-based pollution, and allowing for natural recovery (Rinkevich 2008). Active restoration can involve a range of techniques, such as the direct outplanting of dislodged and/or nursery-reared corals, rubble removal, substrate stabilization, and herbivore management (Lirman and Schopmeyer 2016; Fox et al. 2019; Chung et al. 2019). Many coral restoration projects focus on reefs affected by large-scale disturbances (e.g., heat stress; Anthony et al. 2020; Hein et al. 2021), with fewer published studies on effective restoration techniques for localized incidents of vessel groundings (but see Precht et al. 2001; Schittone 2010). Furthermore, a majority of published restoration studies do not track recovery and overall success past 18-months following intervention (Boström-Einarsson et al. 2020), limiting our ability to test and optimize different restoration techniques for long-term recovery.

The direct impacts of vessel groundings can range from physical damage to chemical pollution. Physical damage may involve relatively minor disturbances, such as the creation of superficial scars on reef framework, to more severe effects, including the dislodgement of coral colonies, the pulverization and flattening of reef habitats, and the loss of sedimentary calcium carbonate (Precht et al. 2001). Due to the inherently slow growth rates of corals (e.g., Hubbard and Scaturo 1985), the full recovery from vessel groundings in the absence of human intervention may take decades or more (Precht et al. 2001, Schroeder et al. 2008). This is particularly true for vessel-impacted reefs with rubble-dominated substrates that can limit coral recruitment (Raymundo et al. 2018). Vessel groundings and the resulting pollution from debris have resulted in localized phase-shifts from coral to corallimorph-dominated benthic communities (Work et al. 2008) as well as cyanobacteria blooms that persist over a decade after the impact (Schroeder et al. 2008). Thus, the few case studies incorporating long-term monitoring of vessel groundings have demonstrated the need for active restoration, such as coral outplanting, rubble removal, and debris removal to accelerate the natural recovery of the impact sites (Precht et al. 2001; Rinkevich 1995).

In February 2010, the cargo vessel *M/V Vogetrader* ran aground on a shallow forereef in Oahu, Hawaii (Figure 1). The *Vogetrader* was removed on the same day as the grounding and an impact assessment of the vessel grounding site was conducted in October 2010 (Kolinski 2010). Six distinct injury zones were described at the grounding site that varied in impact type, impact severity, habitat type, and associated benthic community: (1) sediment burial area (berm of pulverized reef framework and sediment), (2) inner channel slope habitats (scattered scars on the slope of the shipping channel), (3) main scar (area where the vessel hull grounded), (4) partial injury fragments (patches of reef flat impacted by vessel grounding), (5) chain halo (area chronically affected by the chain movement of a marker buoy), and (6) southern reef flat scars (superficial scar fragments located on the deeper portion of the grounding site outside of the channel). Kolinski (2010) measured and identified surviving coral colonies in replicate quadrat subsamples (though the number and area of the quadrat replicates varied among zones) across the impact zones and nearby unimpacted reference sites that were assumed to have similar coral communities prior to the vessel grounding. This assessment revealed substantial damage to coral communities across all impact zones, with injuries to 3,478 m^2 of coral reef habitat and an estimated loss of over 100,000 coral colonies.

Based on these findings, resource trustee agencies developed a restoration plan and conducted restoration at the grounding site with the goal of restoring reefs to their pre-disturbance state (NOAA DARP 2017). In 2013, active restoration interventions were implemented, including the removal of 354 m^3 of rubble from the inner channel zone and the outplanting of 643 coral colonies on the reef flat habitat at the main scar zone (Figure 1). The outplants consisted of nearby corals of opportunity dislodged after a storm and included: *Porites lobata* (401 colonies), *Pocillopora meandrina* (212 colonies), *Pocillopora grandis* (14 colonies), and *Montipora* spp. (16 colonies). The outplants were attached to concrete bases (~0.5 m x 0.1 m; diameter x height) separated by ~1 m, with the number of outplants attached to a given concrete base ranging from one to four colonies. The remainder of the zones were unmanipulated, with the expectation that natural recovery (e.g., passive restoration) would return the reefs to pre-disturbance conditions (NOAA DARP 2017).

This study provides a long-term assessment of coral reef communities damaged by the *Vogetrader* 12-years post grounding and 9-years after active restoration interventions. Using data from the 2010 impact assessment (Kolinski 2010), and repeat surveys in 2022, we employed a Before-After-Control-Impact (BACI; Stewart-Oaten et al. 1986) study design in an attempt to examine the effects of passive and active restoration actions (rather than the impact of the vessel grounding) to meet the restoration goal. To address this goal, we (1) assessed temporal change in three indicators: coral community composition, colony density, and colony size at impact and reference sites, (2) quantified rubble persistence at the impact sites and its effects on juvenile coral abundance, and (3) evaluated coral outplant survivorship and change in size over time. We hypothesized that reef sites where active restoration measures were implemented would display more effective recovery trajectories than those designated for passive restoration alone. While we were able to evaluate coral temporal recovery in the three impact zones to determine the overall success of the *Vogetrader* restoration goal, we were hindered by insufficient restoration baseline data that constrained our conclusions drawn about the active restoration actions performed. Thus, our hypothesis remains unanswered. Nevertheless, the lessons learned in this case study offer a unique opportunity to document the challenges encountered when assessing long-term restoration outcomes on coral reefs and we conclude this paper with recommendations we hope will be applied in future grounding incidents.

Methods

The *Vogetrader* grounding area is located on the southern channel entrance to Barbers Point Harbor on southwest Oahu, Hawaii (21.3158 N, -158.1278 W; Figure 1). In November 2022, we resurveyed three of the six injury zones described by Kolinski (2010): inner channel, main scar, and southern scar fragments (Figure 1). The initial damage assessment reported that the main scar extended to the inshore boundary of the vessel grounding area and was separated by the inner channel (to the southwest side) and the berm pile (to the northeast side). Since the berm pile boundary was no longer present in the 2022 surveys, the inshore portion of the main scar was excluded from our study to avoid sampling areas originally designated as the berm pile. With the exception of the southern scar fragments, which consisted of gradually sloping reef flat (13-14 m), the inner channel and main scar zones were further classified into habitat types (originally defined by Kolinski 2010) that varied in depth, topography, and proximity to the shipping channel. The inner channel included lower slope (10–12 m) and upper slope (9–10 m) reef habitats. The main scar consisted of lower slope (12–13 m), upper slope (11–12 m), and reef flat (10–11 m) habitats. Nearby reference sites were established at comparable depths and habitat types outside of the vessel grounding impact area for each of the three zones. Reference sites established in 2022 may have differed from those used by Kolinski (2010) as GPS coordinates were not available for the 2010 reference sites, although the methods used to select the reference sites were consistent (i.e., within a distance no greater than the length of the impact zone). The inner channel and main scar reference sites were located approximately 50–100 m inshore of the impact zones, while the southern scar fragments reference sites were approximately 25–50 m away from the impact sites (Figure 1).

Coral Communities and Rubble:

For the 2022 surveys, ten replicate 1 m^2 quadrat plots were haphazardly placed within each of the impact and reference habitat types in the inner channel, main scar, and southern scar fragments zones (hereafter referred to as 'sites'). For each quadrat, divers recorded coral species and maximum diameter of colonies > 1 cm whose center point fell within the quadrat. Juvenile corals (< 5 cm diameter; Harrison and Wallace 1990; Bak and Engel 1979) found on unattached rubble substrate were denoted to enable comparison of juvenile densities on attached versus unattached substrate. We used photoquadrats to evaluate the percent cover of rubble at the impact and reference sites. Two transect lines were laid bisecting the width and length of each reef habitat within the sites. Photoquad images were then taken every meter along the two transect lines, one meter above the substrate (1 m^2 area/image). Ten images were randomly selected per site, overlaid with ten random points, and classified as either rubble or non-rubble substrate. Percent cover of rubble was calculated for each image then averaged for each site.

Coral Outplants:

To evaluate outplant survivorship and change in size, colony diameter and partial mortality were assessed on each outplant attached in reef flat habitat of the main scar (Figure 1). Though coral outplants were not tagged in 2013, the concrete bases with outplants were conspicuous and easily identifiable in 2022. Given an annual radial growth rate of 11.6 mm yr^{-1} (Grigg 2006) across the target taxa, we used a minimum coral diameter of 10 cm as a size threshold to differentiate between coral outplants and corals that had recruited to the concrete bases since 2013. For each outplant, divers recorded the taxon, maximum diameter, and extent of tissue mortality as a percentage of the colony surface area. Concrete bases devoid of outplants were enumerated, but obscurity caused by turf overgrowth in the absence of identifiable outplants prevented a complete census of empty bases.

Statistical Analysis:

A Before-After-Control-Impact (BACI) survey design was used to identify trends in coral density, mean size, and community composition at the *Vogetrader* impact area, with the interaction of the factors time (B-A; 2010-2022) and treatment (C-I; reference-impact) used to distinguish restoration-induced change from natural variation. Since our BACI assessment of the vessel grounding area did not conform to a classic BACI model (i.e., 'before' conditions of impact and reference sites varied) (Underwood 1992; Underwood 1994; Smokorowski and Randall 2017), additional analyses were needed to fully capture the nature of temporal change. For coral density and colony size, separate three-way ANOVAs were performed for the inner channel and main scar zones, with treatment (i.e., impact, reference), time (i.e., 2010, 2022), and habitat type as fixed effects. All data were evaluated for normality and equal variance, and were square-root transformed to meet these assumptions. If a significant interaction was observed between the three factors, subsequent two-way crossed ANOVAs were conducted for each habitat type. In the case of the southern scar fragments, for which distinct habitat types were not present, a two-way crossed ANOVA was run with treatment and time as fixed effects. If a significant interaction between the main effects (time and treatment) was identified, we conducted one-factor t-tests within levels of the interacting factor (e.g., Before-Reference versus After-Reference; Before-Impact versus After-Impact; After-Reference versus After-Impact) to evaluate post-restoration convergence of impact and reference sites, as well as significance of temporal change for each treatment group.

Three-way PERMANOVA (factors: treatment, time, and habitat) of Bray-Curtis distances were used to compare coral community composition for the inner channel and main scar. For the southern scar fragments, a two-way PERMANOVA test (999 permutations) was performed, with treatment and time used as factors. As with the univariate ANOVAs, significant interactions between treatment and time were followed by single factor PERMANOVAs within each treatment group to evaluate temporal change. All factors were evaluated for multivariate homogeneous dispersion using the betadisper function in the R package 'vegan: Community Ecology Package' (Oksanen et al. 2022). SIMPER analyses were used to identify which taxa led to observed differences in the community composition. Coral community composition was visualized for each zone using non-metric multidimensional scaling (nMDS).

As no benthic cover data were collected in 2010 or 2013, an analysis of juvenile coral density and percent rubble cover was limited to a comparison between impact and references sites using 2022 data only. Separate two-way crossed ANOVAs (factors: treatment, habitat type) were used for the inner channel and main scar zones to evaluate differences in percent rubble cover, juvenile density (ind m^{-2}), and the proportion of juveniles settled on rubble. All data were square-root transformed prior to analyses to meet assumptions of normality and equal variance.

Analyses of coral outplant success included a comparison of mean size between 2013 and 2022 using a non-parametric Kruskal-Wallis one-way analysis of variance for each of the outplanted taxa (as data failed to meet parametric assumptions). To evaluate whether the extent of partial mortality differed across the four outplanted taxa, a one-way ANOVA was performed on the square-root transformed data, with taxon included as a fixed effect. A post-hoc Tukey's test was then used to determine significant differences between taxa.

Calculations and statistical analyses were completed using R (R Core Team 2022) with the R Studio extension (RStudio Team 2022).

Results

For the inner channel, treatment and time had a significant effect and interaction on coral density, whereas habitat did not (3-factor ANOVA, Table 1). This temporal change was primarily driven by a fourfold decrease in coral density (-48.5 ind m^{-2}; mean change) at the reference sites from 2010 to 2022 (Figure 2a; t- test: $t_{27} = 9.50$, $p = 0.001$), accompanied by a nearly twofold increase in coral density (+6.9 ind m^{-2}) at the impact sites over the same time period, though the latter was not significant (t-test: $t_{31} = -0.98$, $p = 0.334$). This resulted in statistically similar present-day colony densities at the inner channel impact and reference sites (t- test: $t_{27} = 1.36$, $p = 0.185$). At the main scar, treatment, time, and habitat had a significant effect on coral density (Table 1). A significant interaction was found between treatment, time, and habitat, requiring us to investigate the interaction of treatment and time for each habitat separately (Table 1). These subsequent two-factor ANOVAs for each habitat type revealed a significant interaction between time and treatment for all three main scar habitats: lower slope (LS), upper slope (US), and reef flat (RF) (Table S1), though the magnitude of this change varied by habitat. Coral density decreased at the main scar reference sites between 2010 and 2022 (t-test: $t_8 = 6.32$, $p = 0.001$), in particular for the US where a fivefold decline (-85.3 ind m^{-2}) was observed (t-test: $t_2 = 5.14$, $p = 0.026$) (Figure 2c). In contrast, coral density increased across all three impact habitat types (t-test: LS, $t_{11} = -6.54$, $p = 0.001$; US, $t_9 = -15.49$, $p = 0.001$; RF, $t_{11} = -14.76$, $p = 0.001$), with the present-day coral density at the US impact site surpassing that of the reference site (t-test: $t_{16} = -3.55$, $p = 0.003$; Table S2). The overall gains in absolute coral density at the main scar were the highest of the three impact zones. Colony density at the southern scar fragments varied by treatment, but not time (Table 1). As in the other zones, a significant interactive effect was detected for treatment and time (Table 1). This response was predominantly driven by a sixfold increase in coral density (+12.3 ind m^{-2}) at the impact sites from 2010 to 2022 (t-test: $t_{17} = -5.26$, $p = 0.001$), though coral density also declined significantly (t-test: $t_{30} = 2.50$, $p = 0.018$), but to a lesser degree (-6.5 ind m^{-2}), at the reference sites (Figure 2e). While present-day coral densities measured at the impact sites have converged to similar values as the reference sites (14.7 ± 2.5 and 13.6 ± 2.6 ind m^{-2}, respectively; mean $\pm$ SEM; t-test: $t_{37} = -0.51$, $p = 0.614$), they still fall short of 2010 reference levels (20.1 ± 2.4 ind m^{-2}; t-test: $t_{30} = 2.13$, $p = 0.041$). The southern scar fragments had the lowest coral densities among all zones.

With respect to colony size, inner channel corals were significantly larger at the reference sites compared to the impact sites but remained stable over time with no interaction between time and treatment (3-factor ANOVA, Table 1, Figure 2b). The main scar was the only zone that experienced a significant interaction between time and treatment (Table 1, Figure 2d) such that colony size increased significantly over time at the impact sites (+4.3 cm; mean change; t-test: $t_{21} = -8.50$, $p = 0.001$) but remained stable at the reference sites (-0.2 cm; t-test: $t_{26} = 0.73$, $p = 0.475$). Despite this increase in colony size, corals at the main scar impact sites were still significantly smaller than the reference sites (4.8 ± 0.5 cm and 7.0 ± 0.5 cm, respectively; mean $\pm$ SEM) in 2022 (t-test: $t_{56} = 2.73$, $p = 0.008$). Coral size did not vary among habitats at either the inner channel or main scar zones (Table 1). For the southern scar fragments, coral size did not vary among treatment, time, or their interaction, indicating that the vessel grounding did not impact size at all in this zone (Table 1, Figure 2f). In contrast to density, present-day mean coral size was greater at impact sites in the southern scar fragments zone than the impact sites of the other two zones. Mean coral densities and sizes for all impact zones and reference areas are provided in Table S2.

Regardless of habitat type, community composition of the inner channel was significantly different across the factors of time and treatment, as well as their interaction, indicating impact and reference sites changed differently over time (PERMANOVA, Table 2). Indeed, coral communities at the inner channel impact sites were unchanged since the vessel grounding (one-way PERMANOVA: $F_{1,32} = 1.23$, $p = 0.289$), while the reference sites changed significantly since the grounding ($F_{1,32} = 16.76$, $p = 0.001$). The change at the reference sites was driven by losses of *M. patula*, *M. capitata*, and *P. lobata* colonies (SIMPER) and greater dispersion in the coral community in 2022 (betadisper: $F_{1,32} = 8.75$, $p = 0.006$; Figure 3a). Regardless of the community shifts observed at the inner channel reference sites, present-day communities were statistically different across treatments (one-way PERMANOVA: $F_{1,39} = 4.94$, $p = 0.009$). In the main scar, coral communities differed by time, treatment, and habitat (Table 2). Again, a significant interaction between time and treatment was found, with significant temporal differences in coral communities at both the impact sites (one-way PERMANOVA: $F_{1,42} = 115.31$, $p = 0.001$) and reference sites ($F_{1,34} = 7.43$, $p = 0.002$), yet driven by different taxa. At the reference sites, *M. patula*, *M. capitata*, and *P. lobata* declined, while at the vessel grounding impact sites community composition was primarily driven by increases in *M. capitata*, and *P. lobata* (SIMPER, Figure 3b). Present-day coral communities were not statistically different for the main scar impact and reference sites (one-way PERMANOVA: $F_{1,58} = 2.21$, $p = 0.072$). In the southern scar fragments zone, time, treatment, and their interaction were significant (Table 2). Both the impact and reference sites differed over time (one-way PERMANOVA: $F_{1,26} = 12.50$, $p = 0.001$, $F_{1,30} = 8.32$, $p = 0.003$, respectively). At the impact sites immediately after the grounding, the coral community was more homogenous than in 2022 (betadisper: $F_{1,26} = 13.25$, $p = 0.001$) as densities of *M. capitata* and *P. lobata* colonies recovered. In contrast, declines in *M. capitata*, *P. lobata*, *M. patula*, and *P. meandrina* densities drove the temporal differences at the reference sites, coupled with increased dispersion (betadisper: $F_{1,30} = 9.13$, $p = 0.005$; Figure 3c). Present-day coral communities were statistically similar between the impact and reference sites of the southern scar fragments zone (one-way PERMANOVA: $F_{1,38} = 1.62$, $p = 0.204$).

In 2022, rubble cover ranged from 8% to 15% at the reference sites (12 ± 2%; mean ± SEM) and 15% to 44% at the impact sites (31 ± 4%), with significantly more rubble at both the inner channel (ANOVA: $F_{1,36} = 19.52$, $p < 0.001$) and main scar impact sites ($F_{1,54} = 6.31$, $p = 0.015$) than in their corresponding reference sites. Despite these differences in rubble cover, total juvenile coral densities did not differ between the impact and reference sites in the inner channel (17.3 ± 2.4 and 14.6 ± 1.9 ind m^{-2}, respectively; $F_{1,36} = 0.53$, $p = 0.470$) or the main scar (24.3 ± 3.3 and 17.8 ± 1.6 ind m^{-2}, respectively; $F_{1,53} = 1.84$, $p = 0.181$). Across all impact and reference sites, an average of 29 ± 4% (mean ± SEM) and 22 ± 4% of juveniles were observed on rubble, respectively. The proportion of juveniles on rubble was significantly higher at the US impact site of the inner channel than the corresponding reference site ($F_{1,18} = 14.47$, $p = 0.001$; Figure 4). This was also the site with the greatest amount of rubble cover (44 ± 4%).

Of the 643 colonies outplanted in 2013, 290 colonies with live tissue remained in 2022; 33% of *P. lobata*, 64% of *P. meandrina*, 44% of *Montipora* spp., and 100% of *P. grandis* colonies. Colonies not relocated in 2022 were assumed to be dead or dislodged, as 28 concrete bases were observed at the site devoid of any live corals. The average outplant size was significantly larger in 2022 compared to 2013 for *P. lobata* (+10.5 cm; mean change), *P. meandrina* (+8.7 cm), and *Montipora* spp. (+11.5 cm), but not for *P. grandis* (+6.9 cm) (Table 3). *P. lobata* and *P. meandrina* colonies had significantly higher partial mortality (60 ± 3% and 56 ± 3%, respectively; mean ± SEM) compared to *P. grandis* (21 ± 9%), while *Montipora* spp. had intermediate values (32 ± 15%) of partial mortality (ANOVA: $F_{3,288} = 8.40$, $p < 0.001$; Figure 5).

Discussion

Status of the Vogetrader Impact Area:

Our results indicate that none of the three impact zones recovered to pre-disturbance reference levels for all three indicators of coral communities. Rather, recovery trajectories were distinct across the impacted zones, likely driven by differences in the severity of impact in each zone and the habitat quality (e.g., the number and size of coral present in the zone) prior to impact. Minimal recovery was observed at the inner channel impact sites, which experienced no significant change in coral community metrics since the grounding despite rubble removal and passive restoration efforts. The absence of any measurable recovery could be attributed to the severity of damage when the *Vogetrader*'s bow excavated the reef slope along the edge of the shipping channel, gouging a large portion of the reef framework and creating a large rubble berm along the inner channel zone (NOAA DARP 2017). As a result, the reef framework in this zone was destabilized and semi-consolidated (NOAA DARP 2017) and remains so 12 years after the impact, likely precluding coral community recovery. These findings are consistent with a prior study that reported a shift in coral communities that were far different from pre-injury communities when damaged reef framework was left unrestored (Precht et al. 2001).

In contrast, the main scar impact sites had a more favorable recovery trajectory, with moderate increases in both coral density and size, although colonies remained smaller than the temporally stable reference sites. Coral densities were equal between the main scar RF where active outplanting occurred and the main scar US where only passive restoration took place. This similarity indicates that the density of outplants (643 relatively large colonies; 16 cm average starting diameter) and/or the outplant design (widely spaced veneer of outplants) was not sufficient to influence a response metric like coral density a decade later. Admittedly, differences in habitat between restoration type, as well as the lack of baseline data performed with respect to the active restoration actions, limits our ability to definitively state whether the active restoration in the main scar increased colony density. The *Vogetrader* hull scraped across the substrate resulting in complete removal of all corals in this zone; yet the reef framework was not severely destabilized compared to the inner channel (NOAA DARP 2017; Kolinski 2010). Our results suggest a positive recovery trajectory is possible for scraping impacts where the reef framework remains largely intact.

The southern scar fragments (SSF) sites demonstrated the most successful recovery of the three zones, with coral density and community composition nearing 2010 reference levels, and colony size showing no impact from the grounding. The improved recovery may reflect less severe vessel impacts in the SSF zone than the other two zones, consisting primarily of superficial scraping and scouring of the benthos (NOAA DARP 2017), and the SSF coral communities in the 2010 reference sites were marginal—corals were small and scarce (Kolinski 2010). These long-term findings (12 years) contrast with a previous study using passive restoration in Panama, which reported no recovery on marginal reefs two years after a physical disturbance (Schloder et al. 2013). The recovery trajectory in the SSF suggests that given sufficient recovery time, passive restoration may be effective for marginal reef habitats where grounding impacts are minimal.

Despite the limitations in design, assessing the level of recovery from the *Vogetrader* restoration efforts hinges on whether we consider the restoration target to be the reference sites in 2010 at the time of the grounding, or reference sites in the present day. Due to the effects of climatic and environmental stressors (Hoegh-Guldberg et al. 2017, Hughes et al. 2017; Bruno and Selig 2007), over the long-term, corals at reference sites are likely to exhibit temporal declines that are unrelated to vessel groundings (Viehman et al. 2009). Thus, the likelihood of recovery to 2010 pre-grounding levels at the main scar may be impractical under contemporary reef conditions. We can speculate about why coral densities at the main scar (and to a lesser extent, at the inner channel and southern scar fragments) reference sites declined over time. This decline may reflect a difference in reference sites surveyed in 2010 and 2022, as GPS coordinates for the 2010 reference sites were unavailable. Alternatively, a temporal decline may have occurred in West Oahu coral populations due to bleaching-induced mortality during thermal stress events in the Hawaiian Islands in 2014, 2015, and 2019 (Bahr et al. 2017; Winston et al. 2022). However, the bleaching extent on West Oahu was relatively minor compared to the rest of the Hawaiian Islands (Winston et al. 2022), and therefore likely not responsible for the large changes in density observed at the reference sites. The decline at reference sites may instead reflect large swell events that cause coral damage and dislocation, as was documented along this coastline in 2013 (NOAA DARP 2017).

Regardless of the mechanism driving the decline in coral density at the reference sites, our study demonstrates how evaluations of restoration success on coral reefs can deviate from the classic BACI framework assumptions. A classic BACI model relies on the interaction of treatment and time to demonstrate that an environmental perturbation (i.e., in our case, a restoration action) has affected the study system, and assumes that impact and reference sites were similar prior to the environmental perturbation and that change over time is larger at the impact sites than the reference sites (Underwood 1992; Underwood 1994; Smokorowski and Randall 2017). In design, assessments of restoration success following a vessel grounding do not conform to a classic BACI model. Selected reference sites are chosen to represent pre-grounding conditions, with 'before' conditions varying between grounding and reference sites. Therefore, the restoration goal is for grounding sites to become similar to the reference sites after a given period, which is regarded as a 'backwards BACI analysis' (sensu Chevalier et al. 2019). Interpreting BACI studies after disturbances on coral reefs is further complicated by potential temporal change in coral communities at reference sites (i.e., shifting baselines) that violates a primary BACI assumption of stasis at control sites over time. Thus, significant interactions between treatment and time can arise for different reasons—such as larger changes at the reference sites over time, as seen with our *Vogetrader* monitoring efforts. Therefore, additional statistics beyond a BACI interactive effect (such as the post-hoc statistics used in this study) are useful to full capture the nature of temporal change when assumptions of the classic BACI model are not met (Chevalier et al. 2019).

Influence of Rubble at the Vogetrader Impact Area:

Our study highlights the persistence of unconsolidated rubble following severe damage from a vessel grounding. For the main scar and inner channel zones, rubble cover averaged 30.4% at the impact sites compared to 11.8% at the reference sites, indicating that elevated rubble cover can persist 12 years after a vessel grounding. The longevity of unconsolidated rubble on reefs has been documented following acute storm disturbances and physical damage (e.g., dynamite fishing, vessel groundings) (Dollar and Tribble 1993; Riegl 2001), which is in support of the decade-long rubble persistence reported for the *Vogetrader* grounding area. In 2013, restoration practitioners considered additional substrate stabilization using cement, but chose not to implement it because they hypothesized that CCA, coral, and other benthic organisms would naturally stabilize the reef framework (NOAA DARP 2017). Our study indicates that there has been minimal natural consolidation as rubble remains abundant. Unlike its tropical counterparts, subtropical reefs such as those around Oahu are especially prone to rubble persistence due to the lower abundance of benthic organisms that naturally stabilize the reef framework (Huntington et al. 2022). The 2013 active restoration objective to reduce rubble cover to 20% at the impact sites (Parry 2013) was a good target, as the reference sites all contained < 20% rubble in 2022. Unfortunately, the majority of the *Vogetrader* impact area did not meet this target; only the main scar RF was < 20% rubble. However, the lack of initial rubble cover data from 2010 and 2013 limits our ability to definitively assess the efficacy of the rubble removal actions, as additional reef erosion 9-years post active restoration could have contributed to the high present-day rubble cover at the impact sites.

Despite more rubble at the impact sites, our study provided mixed evidence for unconsolidated rubble negatively affecting early life stages of corals at the *Vogetrader* impact area. We observed similar coral juvenile densities across the impact and reference sites, suggesting that elevated rubble cover at the impact sites has not meaningfully affected total juvenile abundance across the *Vogetrader* grounding area. Several studies have reported a negative relationship between rubble cover and the survivorship of juvenile coral recruits (Fox and Caldwell 2006; Viehman et al. 2018). While we did not track juvenile survival as part of this study, these seemingly counterintuitive findings may be the result of conflicting effects of rubble, whereby the presence of biofilms and cryptic habitat in rubble fields increases larval settlement (Harrington et al. 2004; Webster et al. 2004), but the movement of the rubble reduces post-settlement survival (Kenyon et al. 2020; Fox et al. 2003). We did, however, identify subtle differences in the proportion of juvenile coral settled on rubble. The inner channel, where rubble covered an average of 36% of the benthos, had the greatest proportion of juvenile coral on rubble (47 ± 7%; mean ± SEM among all habitats) compared to the proportion in the main scar (16 ± 4%), where rubble cover averaged 27%. While the mechanisms driving the difference in the proportion of juveniles on rubble between the inner channel and main scar are unknown, these results suggest that the damaged reef framework and the abundance of rubble in the inner channel may have constrained recovery across this zone.

Efficacy of Coral Outplanting:

In 2014, survivorship of coral outplants was high — 89% one year after outplanting (NOAA DARP 2017). Yet by 2022, only 45% of the outplants had survived. This falls well short of the 64% average survival rate described in a recent review of 94 coral transplantation studies (Boström-Einarsson et al. 2020). However, only three studies incorporated a monitoring period longer than nine years, with outplant survivorship ranging from 9% (Garrison and Ward 2012) to > 85% (Hudson et al. 1989; Rodgers et al. 2017). These studies demonstrate that long-term outplant survival rates are highly variable and that the rate we observed at the *Vogetrader* impact area falls near the midpoint of those reported, albeit with a wide range, nearly a decade after their attachment. Furthermore, our results show that while survival within the first year may be high, outplant survivorship can still decline substantially over time.

Of additional concern was the high extent of partial mortality—particularly for *P. lobata* and *P. meandrina*—indicating that while some outplants were able to persist over time, many outplants are not thriving under present-day conditions. High partial mortality may be an unavoidable limitation involved in using corals of opportunity for restoration. By definition, corals of opportunity have been dislodged from the reef and typically have higher pre-existing partial mortality prior to use in restoration relative to nursery-reared corals. The criteria for a viable outplant at the *Vogetrader* site was less than 50% partial mortality (Parry 2013). Mean partial mortality in outplants was comparable to levels observed in wild colonies of similar size at the main scar impact sites for *P. lobata* and *Montipora* spp. (mean partial mortality of wild and outplant populations; *P. lobata* = 46% and 60%, *Montipora* spp. = 32% and 32%), but was threefold higher for *P. meandrina* (17% and 56%). Despite the high partial mortality and low survivorship reported for coral outplants, the use of corals of opportunity for reef restoration can still be a viable strategy as it is a cost-effective method for re-establishing reproductively mature colonies and 3-dimensional complexity compared to nursery-reared corals (Edwards and Clark 1999; Bayraktarov et al. 2019).

The *Vogetrader* outplant design is particularly valuable as it consisted of branching, massive, and encrusting coral taxa, and thus can be used to understand morphological differences in outplant survival and size change. Prior restoration projects have predominantly focused on fast-growing/weedy branching corals (e.g., *Acropora* spp.), whereas slow-growing/stress-tolerant corals such as *Porites* spp., and competitive taxa such as *Montipora* spp., are often overlooked despite their importance in reef building (Montero-Serra et al. 2018; Edwards and Clark 1999; Loya et al. 2001). Our findings, however, indicate contrasting results to these described morphological trends. In our study, *P. lobata* (massive) and *Montipora* spp. (encrusting) experienced the greatest increase in size over time, yet the lowest survivorship of the four outplanted taxa. In comparison, the two branching corals (*P. meandrina* and *P. grandis*) had the smallest mean increase in size but the highest survivorship. Size differences across outplants may have played a role in outplant survivorship. In general, survivorship typically increases with outplant size (Smith and Hughes 1999; Becker and Mueller 2001; van Woesik et al. 2021). Thus, the correspondingly high survival rates we observed for *P. meandrina* and *P. grandis* may reflect that >80% of the outplanted *Pocillopora* colonies had a starting diameter > 15 cm. Regardless of size, the high survival of the *Pocillopora* spp. is noteworthy as this genus is often regarded as less resilient to stress and exhibits high mortality after injury (Loya et al. 2001; van Woesik et al. 2011; Pisapia et al. 2015). In comparison, fewer colonies of *P. lobata* and *Montipora* spp. exceeded a starting size of 15 cm (62% and 56%, respectively), which could be linked to the lower survivorship we encountered for those taxa.

Our results revealed that the scale of coral outplanting implemented at the *Vogetrader* grounding area did not accelerate recovery of coral density at the outplant site (main scar: reef flat) relative to adjacent passive restoration sites (main scar: upper slope). These results support findings from another study that suggest that outplanting corals did not yield significant improvements in coral cover, natural recruitment, or juvenile coral abundance (Roper et al. 2022). This, however, does not indicate that coral outplanting is an ineffective restoration strategy to increase coral density. Rather, these results instead reflect a flawed outplant design implemented at the *Vogetrader* impact area. The initial outplanting of 643 colonies to an area of 843 m^2 (0.76 ind m^{-2}) is unlikely to have a meaningful effect on coral density after nine years at a site where passive restoration (e.g., natural recruitment) yielded a density of 37.5 ind m^{-2}. This becomes even more pronounced when considering that the 45% outplant survivorship signifies a present-day outplant density of 0.34 ind m^{-2}. Thus, the density of outplants at the *Vogetrader* impact area was not sufficient (widely spaced, thin veneer of large colony outplants) to influence a response metric like coral density relative to the level of natural recovery. The indirect effects of the outplant efforts on coral reef ecosystem function were evident at the *Vogetrader* impact area. For example, *P. grandis* outplants added complex reef structure to the site that attracted numerous reef fish (Figure S1). These findings underscore the importance of including clearly defined goals when using outplants in restoration. If the restoration design is aimed at increasing coral density at a reef site, the best approach may be to outplant a high number of smaller sized colonies. However, if the restoration objectives are designed to bolster reef structure and habitat complexity, adding clusters of large coral outplants could be a viable approach, and metrics of habitat complexity or fish habitat provisioning would be a more appropriate response variable than colony density to evaluate this objective.

Conclusions / Recommendations

Monitoring of coral restoration efficacy past the first few years is uncommon and published studies of restoration outcomes following vessel groundings are even rarer. This has led to an inadequate understanding of which restoration strategies, if any, are effective for restoring reefs following vessel groundings. While our findings are specific to the restoration design and localized conditions at the *Vogetrader* impact area, we have gleaned several important messages from this study.

1. Future monitoring efforts should consider tailoring the restoration actions to the severity of the impact within a given area. Our results suggest that marginal habitats with a low impact severity, such as the southern scar fragments, will likely recover to pre-grounding reference conditions from passive restoration alone given sufficient recovery time. In contrast, reefs with greater physical damage and a higher starting coral density (main scar and inner channel) require additional active restoration strategies and/or recovery time to reach pre-impact reference conditions.

2. We observed longevity of rubble at the *Vogetrader* impact sites, with rubble persisting over a decade post-grounding despite relatively high wave energy during summer months and some active removal efforts. Thus, we suggest that active rubble removal be spatially extensive and thorough to be effective, especially in areas where rates of natural consolidation are expected to be low. In addition, we propose that restoration at vessel grounding sites should also incorporate substrate stabilization techniques where feasible, such as the deployment of mesh, frames, or large boulder structures over rubble beds (Ceccarelli et al. 2020)—particularly in areas where the vessel impacted the stability of the reef framework. These structures can facilitate recovery by preventing the mobilization of unconsolidated reef framework, providing solid settlement substrates for coral recruits, and establishing 3-dimensional structure for fish.

3. The outplanted taxa at the *Vogetrader* impact area have been infrequently used in reef restoration (Boström-Einarsson et al. 2020). Thus, our data on long-term survivorship, change in size, and extent of partial mortality inform restoration management of which Pacific taxa are best suited for outplanting using a corals of opportunity approach. Though the sample size was low ($n = 14$), our results indicate that *P. grandis* should be considered for future restoration efforts for its high survivorship and habitat provisioning attributes, but a larger sample size is needed to definitively make this recommendation.

4. These findings underscore the importance of defining the scale to which outplants should be deployed during the restoration planning phase—informed by natural recruitment and survival rates from the area—in order to achieve a measurable effect on coral density.

5. Given the divergence from a classic BACI study design inherent to assessing restoration success from acute impacts in coral reef systems, we advocate for using additional statistics beyond a BACI interactive effect (such as the post-hoc statistics used in this study) to fully capture the natural temporal change in complex reef systems and improve conclusions of reef recovery.

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Data Availability

The data that support the findings are available at NOAA National Centers for Environmental Information (Accession Number 0282702).

Additional Information

Competing Interests: The authors declare no competing interests.

Tables

Table 1 | Three-way crossed ANOVAs results for coral density (ind m^{-2}) and coral diameter (cm) in the inner channel, main scar, and two-way crossed ANOVA results for the southern scar fragments. All factors were treated as fixed effects. Data were square-root transformed. P-values < 0.05 are indicated by *.

Zone		**Coral Density**			**Coral Diameter**		
Effect	**df**	**SS**	**F**	**P**	**SS**	**F**	**P**
Inner Channel							
Treatment	1	130.794	47.696	< 0.001*	23.675	35.788	< 0.001*
Time	1	54.538	19.888	< 0.001*	2.091	3.162	0.081
Habitat	1	9.414	3.433	0.069	0.045	0.069	0.795
Treatment:Time	1	87.854	32.037	< 0.001*	0.095	0.143	0.706
Treatment:Habitat	1	12.545	4.575	0.037*	0.401	0.607	0.439
Time:Habitat	1	11.909	4.343	0.042*	0.916	1.385	0.244
Treatment:Time:Habitat	1	7.496	2.734	0.104	1.600	2.418	0.125
Residuals	59	161.792			39.031		
Main Scar							
Treatment	1	29.553	21.968	< 0.001*	14.419	35.966	< 0.001*
Time	1	56.682	42.135	< 0.001*	21.138	52.725	< 0.001*
Habitat	2	40.933	15.214	< 0.001*	0.608	0.759	0.472
Treatment:Time	1	302.802	225.089	< 0.001*	11.537	28.778	< 0.001*
Treatment:Habitat	2	5.217	1.939	0.152	2.135	2.663	0.077
Time:Habitat	2	13.010	4.836	0.011*	0.057	0.071	0.932
Treatment:Time:Habitat	2	11.376	4.228	0.019*	0.197	0.246	0.783
Residuals	68	91.477			27.262		
Southern Scar Fragments							
Treatment	1	11.425	7.183	0.010*	0.098	0.231	0.633
Time	1	2.638	1.659	0.203	0.703	1.657	0.203
Treatment:Time	1	37.628	23.658	< 0.001*	1.498	3.533	0.065
Residuals	56	89.067			23.752		

Table 2 | PERMANOVA results for coral community composition of the inner channel, main scar, and southern scar fragments zones. All factors were treated as fixed effects. Data were square-root transformed. P-values < 0.05 are indicated by *.

Zone Effect	**df**	**SS**	**Pseudo-F**	**P (permutation)**
Inner Channel				
Time	1	0.526	5.857	0.004*
Treatment	1	1.617	17.989	0.001*
Habitat	1	0.203	2.262	0.096
Time:Treatment	1	0.586	6.522	0.002*
Time:Habitat	1	0.185	2.059	0.135
Treatment:Habitat	1	0.227	2.529	0.064
Time:Treatment:Habitat	1	0.204	2.266	0.107
Residuals	59	3.007		
Main Scar				
Time	1	3.189	68.495	0.001*
Treatment	1	0.755	16.220	0.001*
Habitat	2	0.594	6.380	0.001*
Time:Treatment	1	2.427	53.099	0.001*
Time:Habitat	2	0.243	2.610	0.051
Treatment:Habitat	2	0.234	2.516	0.047*
Time:Treatment:Habitat	2	0.073	0.787	0.489
Residuals	68	1.938		
Southern Scar Fragments				
Time	1	0.196	3.491	0.023*
Treatment	1	0.263	4.685	0.005*
Time:Treatment	1	1.012	18.038	0.001*
Residuals	56	2.734		

Table 3 | Mean outplant diameter (SEM) by year, followed by Kruskal-Wallis (K-W) test results evaluating significant differences in mean colony diameter across years. P-values < 0.05 are indicated by *.

Taxon	Outplant Diameter (cm)		K-W: 2013 - 2022	
	2013	**2022**	χ^2	**P**
***Montipora* spp.**	13.75 (2.21)	25.29 (2.59)	8.37	0.004*
P. lobata	14.48 (0.44)	24.97 (0.93)	96.19	< 0.001*
P. meandrina	18.63 (0.58)	27.29 (0.81)	77.63	< 0.001*
P. grandis	25.00 (2.09)	31.88 (4.56)	0.01	0.940

Figures

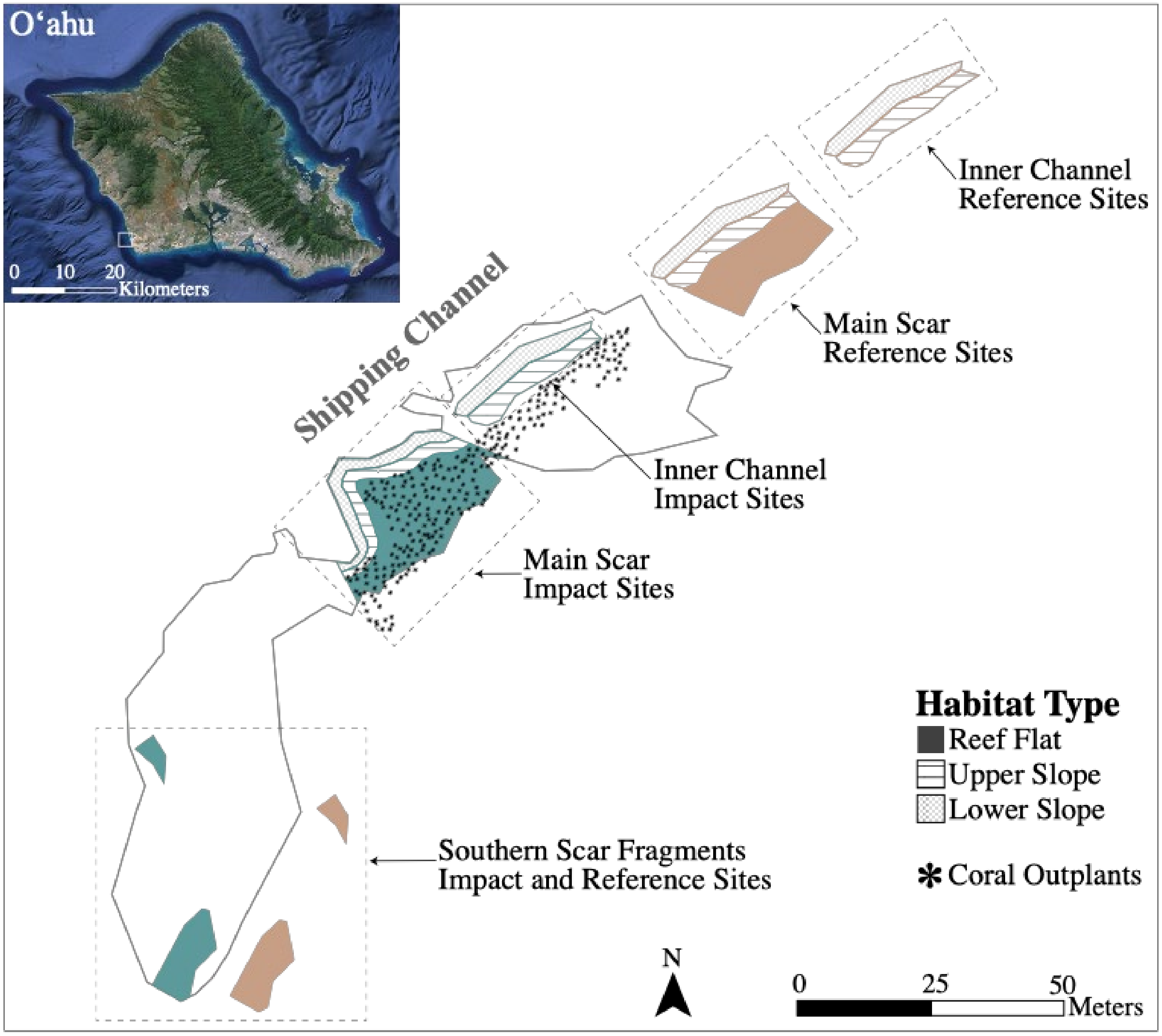

Figure 1 | The *Vogetrader* grounding area (gray polygon) classified into three distinct impact zones, including inner channel, main scar, and southern scar fragments. Boundaries of the impact (turquoise) and reference (tan) sites are indicated. Habitat types are characterized as reef flat (solid color), upper slope (horizontal lines), and lower slope (dashes). Locations of the coral outplants are marked by asterisks. Reference sites are located outside the grounding area. Note, reference sites are not to scale to be able to visualize on this map, but were located within a distance no greater than the length of the impact zone as defined by the initial impact assessment (Kolinski 2010).

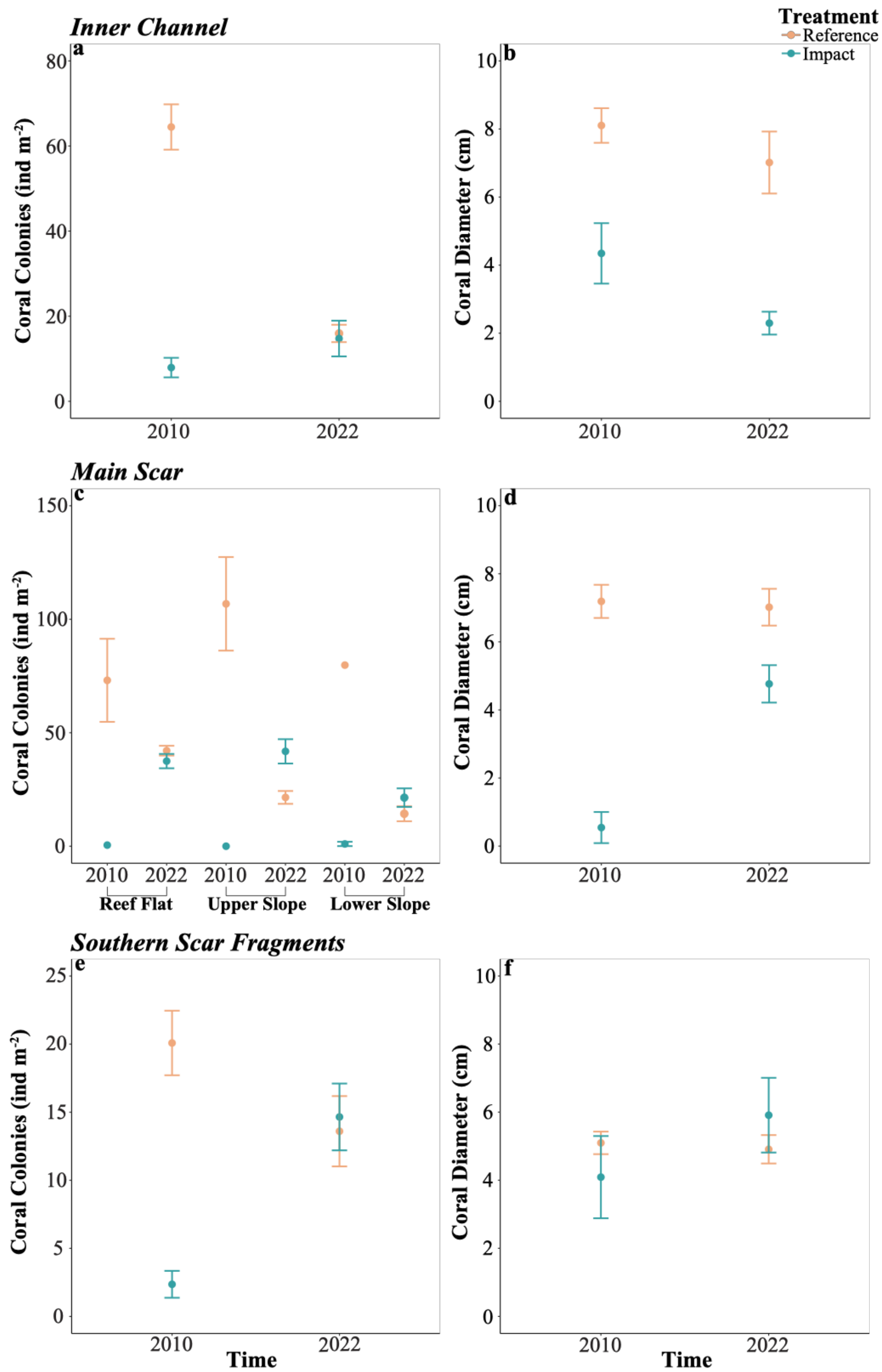

Figure 2 | Comparison of (a, c, e) mean coral density (ind m^{-2}) and (b, d, f) mean coral diameter (cm) by time and treatment for the inner channel, main scar, and southern scar fragments zones. Habitat is presented for coral density in the main scar zone given the significant interaction of habitat, treatment, and time (Table 1). Otherwise, density and coral diameter values were pooled across habitat types. Treatment is characterized as reference (tan) or impact (turquoise) sites. Error bars represent SEM. Data are untransformed. Y-axis scale varies for coral density (panels a, c, e).

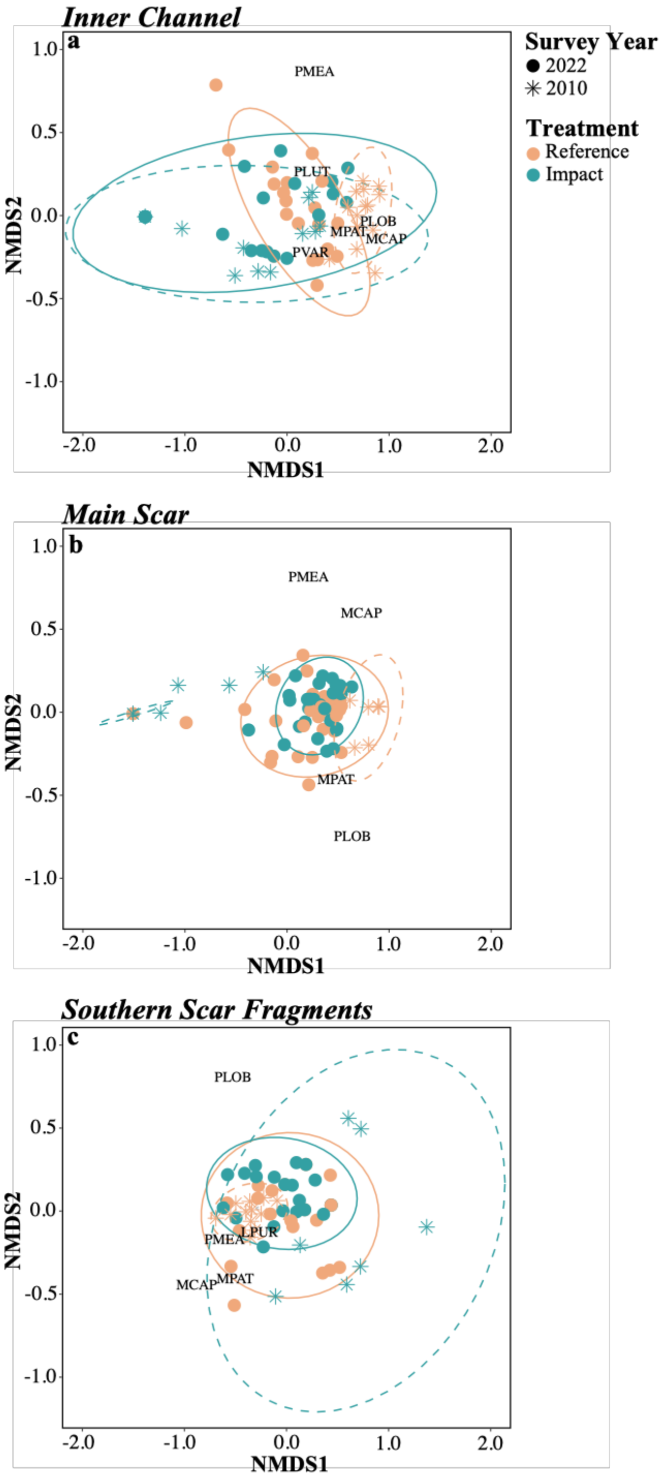

Figure 3 | Nonmetric multidimensional scaling (nMDS) of coral community composition for reference (tan) and impact (turquoise) sites between the two survey years in the (a) inner channel, (b) main scar, and (c) southern scar fragments zones. Ellipses represent 95% confidence-intervals for 2010 (dashed) and 2022 (solid). Significant coral scores ($p < 0.05$) are indicated in grey. Data were square-root transformed.

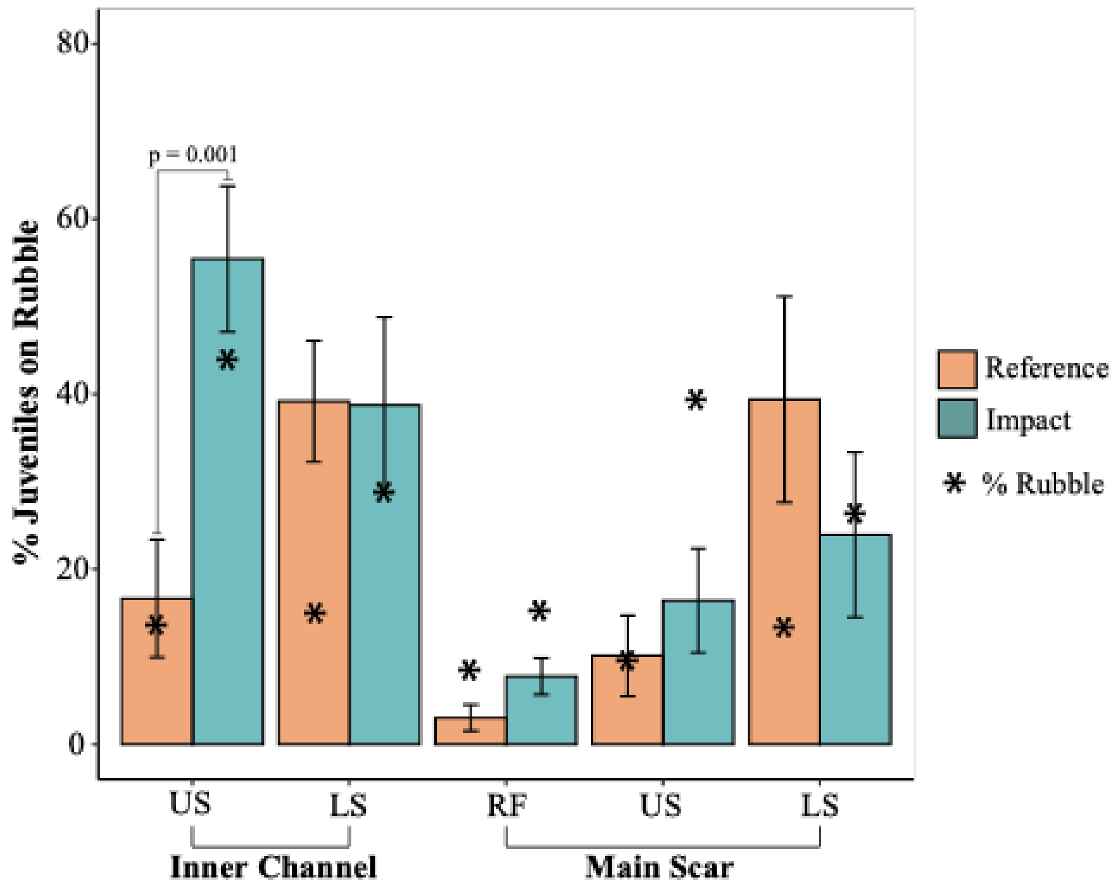

Figure 4 | Mean (± SEM) percent juveniles on unattached rubble across reference (tan) and impact (turquoise) treatments. The inner channel zone includes upper slope (US) and lower slope (LS) habitat types. The main scar zone includes reef flat (RF), US, and LS habitat types. Data are untransformed. Significant p-values (post-hoc Tukey's test) across impact and reference sites are indicated. Star symbol denotes mean rubble cover (%) for each site.

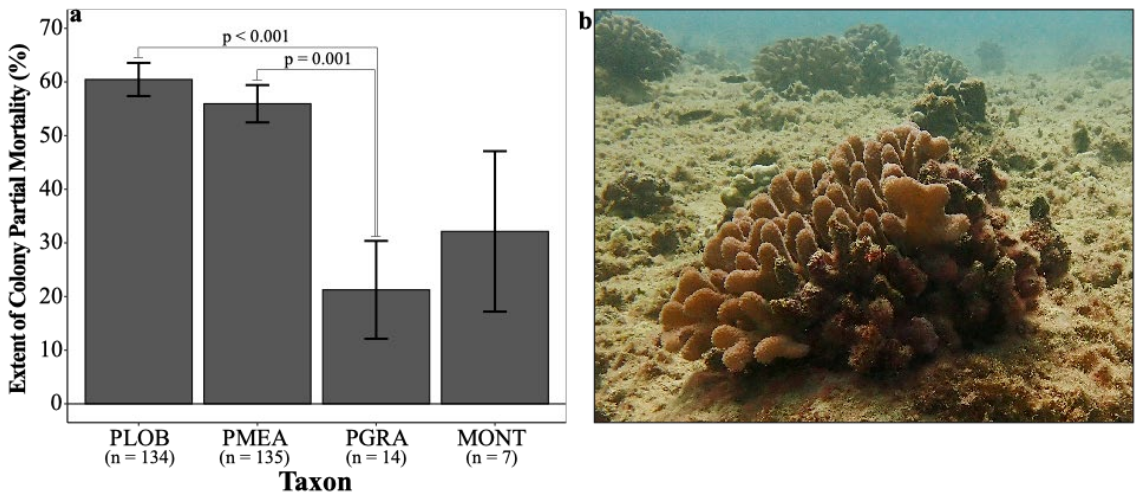

Figure 5 | (a) Mean (± SEM) extent of partial mortality (%) of coral outplants across taxon: *Porites lobata* (PLOB), *Pocillopora meandrina* (PMEA), *Pocillopora grandis* (PGRA), *Montipora* spp. (MONT). Data are untransformed. Significant p-values (post-hoc Tukey's test) between taxon are indicated. (b) Image of *P. meandrina* outplant taken during the 2022 monitoring surveys.

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