Status and Trends Assessments for Land-based Sources of Pollution Impacts on Benthic Reef Communities in Faga'alu Bay, American Samoa

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Cover: NOAA Diver Morgan Winston conducts coral community surveys along the Faga'alu southern shallow forereef. Photo credit: NOAA PIFSC Ecosystem Sciences Division/Brittany Huntington.

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Executive Summary

This report provides a summary of key findings for work completed between 2012 and 2020 to assess the effectiveness of management actions conducted at the Samoa Maritime quarry in Faga'alu, American Samoa. Collectively, these data offer a contrast between the 2012/2013 pre-intervention and 2015–2020 post-intervention status, and examine how benthic and coral community response variables differed across factors of year (2012/2013, 2015, 2020) and reef stratum (backreef north, backreef south, forereef north, and forereef south).

Prominent findings of this study include:

1. There was a significant ~12% reef wide reduction in macroalgal cover which was first noticed on the backreef north in 2015 becoming evident reef wide by 2020. Macroalgal communities are spatial and temporally variable and can quickly respond to changes in nutrient concentrations.

2. There was a significant, nearly three-fold increase in juvenile coral density from 2012/2013 to 2020, indicative of higher recruitment and post-settlement survival of corals.

3. Although coral cover decreased between 2012 and 2015, it recovered to the pre-intervention levels by 2020. We speculate this decline was driven by the 2015 bleaching event. The return to pre-intervention coral cover levels by 2020 underpins resilience and fast-growing nature of the Faga'alu coral assemblages.

4. Adult coral colony densities exhibited significant spatial differences but not temporal differences, indicating that the Faga'alu reef has measurably distinct coral communities among the different reef strata, and that those differences have remained unchanged over the survey years.

5. With the exception of the coral genus *Psammocora,* mean colony size of the most abundant taxa varied significantly over time; *Acropora* and *Pocillopora* decreased driven by a greater number of juveniles and small-sized colonies indicating active and successful recruitment. *Montipora*, *Pavona*, and *Porites* colony sizes increased; larger colonies contribute importantly to recovery due to inherent size-based differences in reproductive potential, and the capacity to withstand further disturbances.

6. The inherent high levels of within- and between-strata variability make difficult detection and attribution of temporal change. We propose that future monitoring efforts incorporate:

    - Site-specific, quantifiable performance measures to evaluate management outcomes and implementation of permanent representative stratum-specific sites/transects/plots

    - A selection of unequivocally identifiable 'sentinel' species to control within stratum coral community metrics variability (e.g., adult densities, colony sizes, partial mortality)
    - Novel image-analysis techniques to reduce variability and aid following the temporal trajectories of Faga'alu reef community.

Introduction

Despite the attention that climate change and ocean acidification have received in the last decade, land-based sources of pollution, namely sedimentation, turbidity, nitrification, and contamination remain the main direct cause of coral reef degradation worldwide (Rogers 1990; Fabricius 2005; Bégin et al. 2016). Currently, sediment runoff and agricultural pollution threaten approximately 25% of the total global reef area, with further increases in sediment and nutrient fluxes projected to occur over the next 50 years (Burke 2011; Kroon et al. 2014). Chronic sedimentation stress (i.e., the combination of suspended, re-suspended, and deposited sediment) disrupts the coral host vital processes, including alimentation, reproduction, and growth, leading to impoverished and ecologically dysfunctional coral communities (Philipp and Fabricius 2003; Risk and Edinger 2011).The occurrence of partial mortality, bleaching, and all forms of coral health impairment increase with nearshore anthropogenic processes, particularly sedimentation stress (Nugues and Roberts 2003; Smith et al. 2008). Evidence suggests that within a certain threshold, a range of turbidity levels can moderate coral bleaching during thermal-stress events (Sully and van Woesik 2020). However, these indirect benefits of shading to reduce impacts from thermal stress do not ameliorate direct impacts of the sedimentation itself on coral health. Today more than ever, under global climate change conditions, building resilient coral reef communities is a top priority. The survival and persistence of many reefs worldwide hinges on timely, sustained, and efficient efforts to eliminate and mitigate local anthropogenic stressors (12th ICRS consensus statement). As coral reef restoration activities continue to surge, the success of these interventions in achieving their primary goals depends on the removal of the initial causes of coral population decline (Baums 2008). Despite the vast body of literature documenting coral reef degradation driven by sediment stress (Erftemeijer et al. 2012; Restrepo et al. 2016), corresponding examples of upland management demonstrating the halting or reversing of coral reef decline are hard to find (Bartley et al. 2014).

Faga'alu Bay, American Samoa, has been historically affected by severe sedimentation stress due to excessive terrigenous runoff resulting from prolonged and destructive land use practices. Land-based sources of pollution impacts to the Faga'alu reef span decades, including the creation of the Matafao landfill in the early 1950s to early 1960s, the establishment and operation of the Samoa Maritime Quarry in the early 1960s, and the extensive reef flat dredging for the development of the Faga'alu recreational park, the boat channel, and launch ramps in the 1970s (U.S. Senate 1972; Latinis et al. 1996; Fenner et al. 2008; Holst-Rice et al. 2016). The main consequence of these activities has been the extensive physical damage, as well as the excess influx and resuspension of silt and clay-type sediments, which have structurally altered and chronically affected the health condition of the adjacent coral reef community (Fenner et al. 2008; Messina and Biggs 2016). Superimposed are distinct geomorphological, depth, light, water circulation, biological and natural disturbance (e.g., the devastating 2009 tsunami) patterns that have shaped each of the reef strata identified in this study.

The Samoa Maritime rock quarry was found to be a major contributor to the periodic flux of excessive sediments onto the adjacent reef in Faga'alu. As a result of the direct quarry impacts, Faga'alu was selected as one of three priority watersheds by the U.S. Coral Reef Task Force (USCRTF) for the Watershed Partnership Initiative (WPI) in August, 2012. This designation fueled discussions between local and federal agencies (The National Fish and Wildlife Foundation, The American Samoa Coral Reef Advisory Group, the American Samoa

Environmental Protection Agency, and NOAA's Coral Reef Conservation Program) to implement corrective actions at the quarry and reduce land-based sources of pollution inputs to the coral reefs in Faga'alu Bay. In September–December 2014, mitigation actions were implemented at the Samoa Maritime Quarry, including the installation of drainage systems, alternative ground cover, and retention ponds (Holst-Rice et al. 2016).

To assess the effectiveness of the management intervention, pre-mitigation status and condition baseline data of the Faga'alu reef, including benthic cover and coral demographics, were collected in 2012–2013 (Holst-Rice et al. 2016). Subsequent, post-intervention monitoring data were collected in 2015 and 2020. The work described in this report was funded by the NOAA Coral Reef Conservation Program (CRCP) through two internal projects entitled: *"Comprehensive Baseline Assessment and Pilot Test of Outcome Performance Measures in Faga'alu Bay, American Samoa,"* awarded to Suzie Holst (NOAA National Ocean Service) and *"Status and Trends Assessment for Land-based Sources of Pollution Impacts on Benthic Reef Communities in Faga'alu Bay and Vatia Bay, American Samoa,"* awarded to the NOAA Ecosystem Sciences Division (ESD). Herein we examine whether the Faga'alu reef has undergone measurable benthic changes, following the mitigation action that may inform local and federal managers on the effectiveness of the management interventions conducted in 2014. More specifically, we aimed to address the following overarching questions:

1. Did benthic cover of the most abundant functional groups (i.e, crustose coralline algae, hard coral, macroalgae, sediment, and turf algae) change over time?

2. Did the coral community composition—i.e., density of adult and juvenile colonies—change through time?

3. Did coral colony size and size frequency distribution change through time?

Methods

Biological surveys implemented a modified stratified random sampling design to assess the survey domain which encompassed the hard-bottom reef habitat from 0 to 18 m in depth within Faga'alu Bay. Based on the geomorphology of the reef, the stratification scheme combined two reef zone categories (backreef and forereef) and cardinal positions (north and south) into four distinct strata: i.e., backreef north, backreef south, forereef north, and forereef south (Figure 1). A digital map of the survey domain was overlaid with a 30 m × 30 m (900 m^2 in area) GIS layer, and grid cells containing hard-bottom reef habitat were designated as the sampling units, hereafter referred to as survey sites. Sampling effort allocation was relative to strata area and sites were randomly assigned within each stratum.
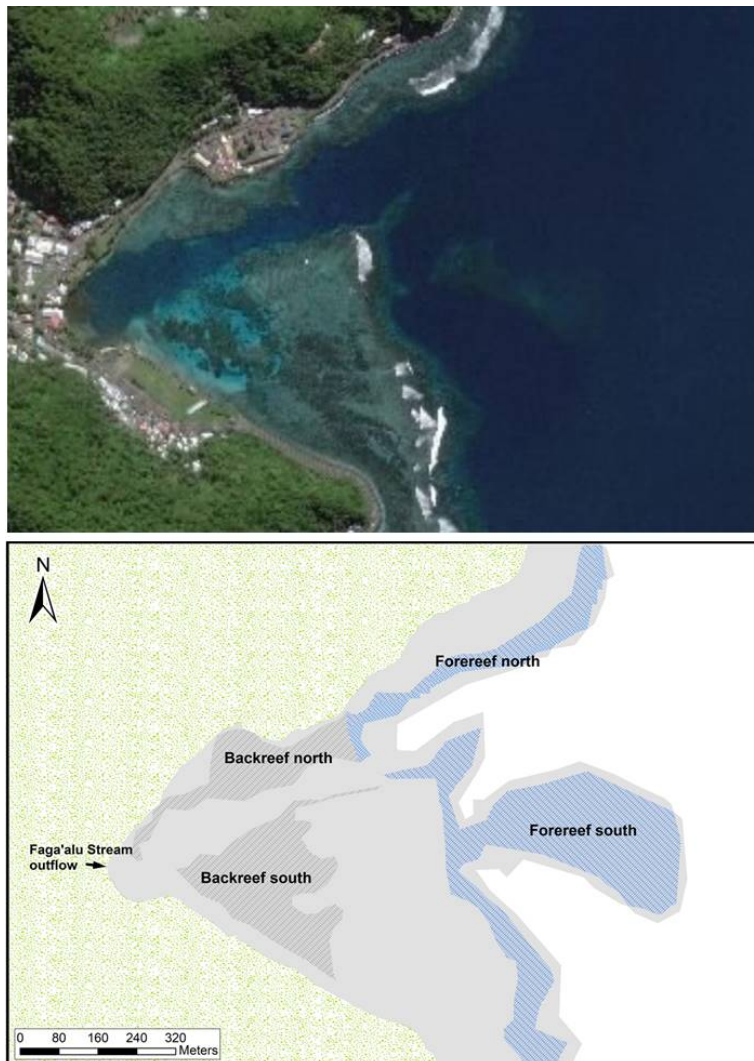
Figure 1. Distribution and spatial coverage of the survey strata in Faga'alu Bay. Backreef strata are depicted in dark gray, while forereef strata are depicted in blue. The boundary of Faga'alu Bay is shown in light gray. A satellite image is included in the upper panel for geographical and visual context.

Benthic surveys were conducted at three time points before and after the quarry mitigation work as follows: pre-intervention: March 2012 and March 2013; and post-intervention in October–November 2015, and January 2020 by staff of NOAA's ESD. The pre-intervention baseline surveys were completed in two separate field missions: March–August 2012 for benthic cover surveys and March 2013 for the coral community composition and size surveys.

Benthic cover, coral community composition, and colony size

Following Swanson et al. (2018) and Winston et al. (2019) belt-transects were the focal point of the biological surveys. Coral community composition was assessed within a number of 1.0 m × 2.5 m segments located at the 0–2.5 m, 5.0–7.5 m, 10–12.5 m, 15–17.5 m, and 20–22.5 m mark along each transect; bottom-time permitting, covering a total area ranging 5–12.5 m^2 per transect. Over time, protocols were adjusted to increase efficiency such that two, 25-m transects were implemented in 2012/2013, two 18-m transects in 2015, and one 18-m transect in 2020. Within segments, all adult coral colonies (≥5 cm maximum diameter) whose center fell within 0.5 m on either side of the transect line were identified to the genus-level (Supplemental Material) and measured for size (maximum diameter to nearest cm). Juvenile coral colonies (<5 cm), distinguished by the presence of a distinct tissue and skeletal boundary (not a fragment of a larger colony), were surveyed within three 1.0 m × 1.0 m segments at the 0–1.0 m, 5.0–6.0 m, and 10.0–11.0 m mark of each transect (covering 3 m^2 per transect). Juvenile colonies were identified to genus and measured for size (maximum diameter to nearest 5 mm).

For the estimation of benthic cover, in 2012 the line-point-intercept (LPI) methodology was implemented at 25-cm intervals (total of 250 points per transect). In 2015 and 2020, still photographs were collected using the photoquadrat method at predetermined points along the transect line with a high-resolution digital camera mounted on a pole (Winston et al. 2019). In 2015 photographs were taken every meter from the 1-m to the 15-m mark on each of two transects per site (n = 30), and in 2020 every meter from the 1-m to the 30-m mark along the extended belt-transect line (n = 30). Photoquadrats were analyzed implementing the computer software CoralNet (Beijbom et al. 2015). Benthic cover of each photograph was assessed by randomly overlaying ten points on each image (total of 300 points per site) and identifying the benthic elements underneath at each point following the 'Tier3b' genus/functional classification scheme outlined in Lozada-Misa et al. (2017). For example, hard corals were differentiated by genus and morphology (e.g. *Acropora* branching, *Acropora* table, *Montipora* encrusting, *Favia*, *Porites* massive), and macroalgae were identified to the genus level (e.g. *Halimeda, Peyssonnelia*, *Lobophora*). To the extent possible 'Tier3b' categories followed the level of detail implemented in the LPI surveys (Supplemental Material).

Statistical analyses

Non-parametric permutation analytical techniques were implemented to evaluate differences in benthic and coral community composition across year and stratum (Table 1). We calculated Bray-Curtis dissimilarity matrices on the untransformed percent cover data for most abundant benthic functional groups [i.e., coral, crustose coralline algae (CCA), fleshy macroalgae (hereafter macroalgae), sediments, turf algae], and most abundant coral taxa (i.e., *Acropora*, *Montipora*, *Pavona*, *Pocillopora*, *Porites*, and *Psammocora*). For adult and juvenile colony densities, only hard coral genera that occurred in at least 10% of sites were included in the analyses (including the hydrocoral *Millepora*) (Table 2) and a Bray-Curtis dissimilarity matrix

was calculated on the fourth root-transformed density data. Subsequent two-factor, permutational multivariate analyses of variance (PERMANOVA) were performed (9999 permutations) to test how response variables of (1) benthic functional group cover, (2) dominant coral taxa cover, (3) adult coral colony density, and (4) juvenile coral colony density, varied across factors of year (i.e., 2012/2013, 2015, and 2020) and stratum (i.e., backreef north, backreef south, forereef north, and forereef south; hereafter: BRN, BFS, FRN, FRS). PERMANOVAs were run using Type II sum of squares due to unbalanced sample sizes among strata (Table 1) and factors were evaluated to meet the assumption of homogeneity of dispersion using the betadisper function. When an interaction between factors was detected, additional stratum-specific PERMANOVAs were computed to identify which combination of year*stratum exhibited significant differences. Pair-wise comparisons were run for each significant factor identified from PERMANOVA outputs and a similarity percentage analysis (SIMPER) were computed to identify which functional or taxonomic group(s) accounted for the differences observed between factor levels. ANOVAs (or the non-parametric equivalents) were performed to detect the taxa-specific differences among survey years. Univariate Kruskal-Wallis and post-hoc Dunn's tests were performed to identify which benthic cover functional groups differed significantly among factor levels. Additional Kruskal-Wallis and post-hoc Dunn's tests were completed to examine differences in mean coral colony sizes among survey years. Lastly, kernel density estimation (KDE) analyses were implemented to assess differences in colony size frequency distributions (SFD) for the most abundant coral taxa among the survey years (following Langlois et al. 2012). KDE analyses were conducted on raw and standardized colony sizes (for all colonies >1 cm in length) to assess whether the SFD shifted in shape (e.g., frequency of size classes) and/or position (e.g., growth). All analyses were run in R (R Core Team 2013) using the *vegan* package 2.5-1v multivariate analyses (Oksanen et al. 2018). Contextual maps illustrating the site-level data are presented in the Supplementary Material.

Table 1. Number of survey sites conducted in Faga'alu Bay, American Samoa, between 2012 and 2020. BRN: backreef north; BRS: backreef south: FRN: forereef north: and FRS: forereef south. * Pre-intervention baseline surveys were completed in two separate field missions: March–August 2012 for benthic cover surveys and March 2013 for the coral colony density and size surveys.

| Years | Strata | | | | |
	BRN	BRS	FRN	FRS	Total
2012/ 2013*	5	13	4	14	36
2015	3	4	4	7	18
2020	8	10	6	11	35
Total	16	27	14	32	89

Table 2. List of coral taxa included in the coral colony density analyses. Only taxa that occurred in more than 10% of the sites were included in the multivariate PERMANOVA analyses.

ACSP: *Acropora*	GASP: *Galaxea*	MISP: *Millepora*
CYPS*: Cyphastrea*	GONS: *Goniastrea*	PAVS: *Pavona*
ECHL: *Echinophyllia*	LEPT: *Leptastrea*	POCS*: Pocillopora*
FASP : *Favia*	LESP: *Leptoseris*	POSP: *Porites*
FUSP: *Fungia*	MOSP: *Montipora*	PSSP: *Psammocora*

Results

Benthic cover

PERMANOVA analysis identified significant differences in percent cover of the reef benthos among years and among strata, with no significant interaction between factors (Figure 2, Table 3). Pairwise post-hoc comparisons indicated that benthic community composition in 2020 was significantly different from both 2012 and 2015 ($p = 0.027$ and $p = 0.040$, respectively). These temporal benthic cover changes reflected: (1) a significant decline in coral cover from 26.3% ± 2.6 SE in 2012 to 14.9% ± 3.1 in 2015, followed by a significant increase back to 24.1 ± 2.7 in 2020 (Kruskal-Wallis); and (2) a significant reduction in macroalgal cover from 26.5% ± 4.6 in 2015 to 14.4 ± 3.3 in 2020 (Kruskal-Wallis) (Fig. 2). Temporal differences in the cover of coralline algae, turf algae, and sediments were not significant. Strata each differed significantly in community composition (Table 3; all pairwise post-hoc analyses among strata of $p < 0.01$), underscoring the existing spatial heterogeneity of benthic cover across Fagaʻalu Bay that persisted through time.
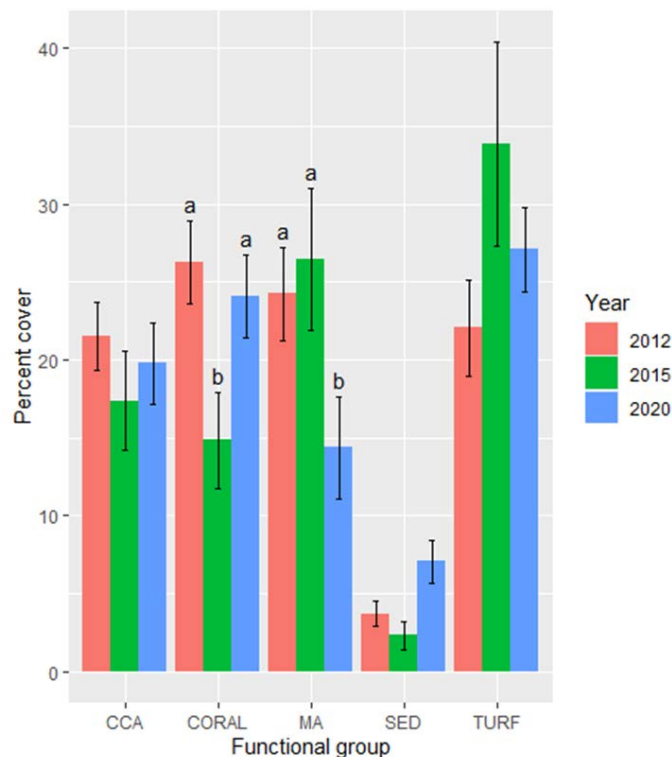
Figure 2. Fagaʻalu reef mean percent cover of most abundant benthic functional groups reef across survey years (CCA: crustose coralline algae; CORAL: hard corals; MALG: macroalgae; SEDI: sediment; and TURF: turf algae). Letter groups above the bars indicate significant differences among years within a functional group, where shared letters indicate no statistical difference between years and differing letters indicate significant difference (Junn's test, $p < 0.05$). If no letters are present for a functional group, the Kruskal-Wallis test was not significant.

Table 3. PERMANOVA results table for benthic cover of most abundant functional groups of coralline algae, hard coral, macroalgae, sediment, and turf algae. df: degrees of freedom; SS: sum of squares; MS: mean sum of squares; Pseudo-F: F value by permutation; P(perm): p-value based on permutations.

Source	df	SS	MS	Pseudo-F	P(perm)
YEAR	2	0.628	0.314	4.446	0.001
STRATUM	3	3.239	1.079	15.281	< 0.001
YEAR*STRATUM	6	0.491	0.081	1.159	0.307
Residuals	77	5.441	0.070		

A PERMANOVA on the cover of the most abundant coral taxa (i.e., *Acropora*, *Montipora*, *Pavona*, *Pocillopora*, *Porites*, and *Psammocora*) revealed significant differences among years, strata, as well as an interaction effect between factors (Table 4a). Subsequent single factor PERMANOVAs (factor = year) per stratum (Table 4b–e) followed by univariate analyses of variance indicated that *Psammocora* cover significantly increased in the BRN from 1.9% ± 0.3 in 2012 to 4.0% ± 1.1 in 2020 (Kruskal-Wallis, H = 8.03; p = 0.018; Dunn's test p = 0.008); and on the FRS *Pavona* cover decreased from 7.2% ± 1.8 in 2012 to 1.0% ± 0.3 in 2020 (Kruskal-Wallis, H = 8.73, p =0.01; Dunn's test, p < 0.01). This cover reduction did not result in measurable stratum-level cover increases of other benthic functional groups (particularly turf or macroalgae), indicating that those changes were inconsequential and highly localized. Although collectively coral cover on the BRN, BRS, and FRS significantly decreased between 2013 and 2015 (PERMANOVA p = 0.01; post-hoc pairwise comparisons p = 0.04), all genus-specific Kruskal-Wallis analyses between survey years yielded p > 0.05, underscoring the existing within stratum heterogeneity. Also, the few significant changes in coral genera over time reflect the high variability in coral cover among sites–even within a given strata (Figure 3). Despite this spatial variability, *Porites* appears to have substantially contributed to the overall reef wide coral cover temporal variability. *Porites* decreased in 2012–2015 and increased in 2020, particularly in the BRN, FRN, and BRS—though high variation in *Porites* cover between sites prevented these trends in abundance across year from being significant within any of the four strata.

Table 4. PERMANOVA results table for cover of most abundant coral taxa (*Acropora, Montipora, Pavona, Pocillopora, Porites*, and *Psammocora*); df: degrees of freedom; SS: sum of squares; MS: mean sum of squares; Pseudo-F: F value by permutation; P(perm): p-value b based on permutations.

a) Coral cover: reef wide					
Source	df	SS	MS	Pseudo-F	P(perm)
YEAR	2	1.610	0.804	4.512	**< 0.001**
STRATUM	3	7.477	2.492	13.970	**< 0.001**
YEAR*STRATUM	6	1.873	0.312	1.750	**0.008**
Residuals	77	13.737	0.178		
b) Coral cover: backreef north					
YEAR	2	0.712	0.356	2.003	**0.035**
Residuals	13	2.311	0.177		

c) Coral cover: backreef south					
YEAR	2	0.771	0.385	2.781	**0.016**
Residuals	24	3.328	0.138		
d) Coral cover: forereef north					
YEAR	2	0.749	0.374	1.677	0.114
Residuals	11	2.455	0.223		
e) Coral cover: forereef south					
YEAR	2	0.786	0.393	2.021	**0.028**
Residuals	29	5.641	0.194		
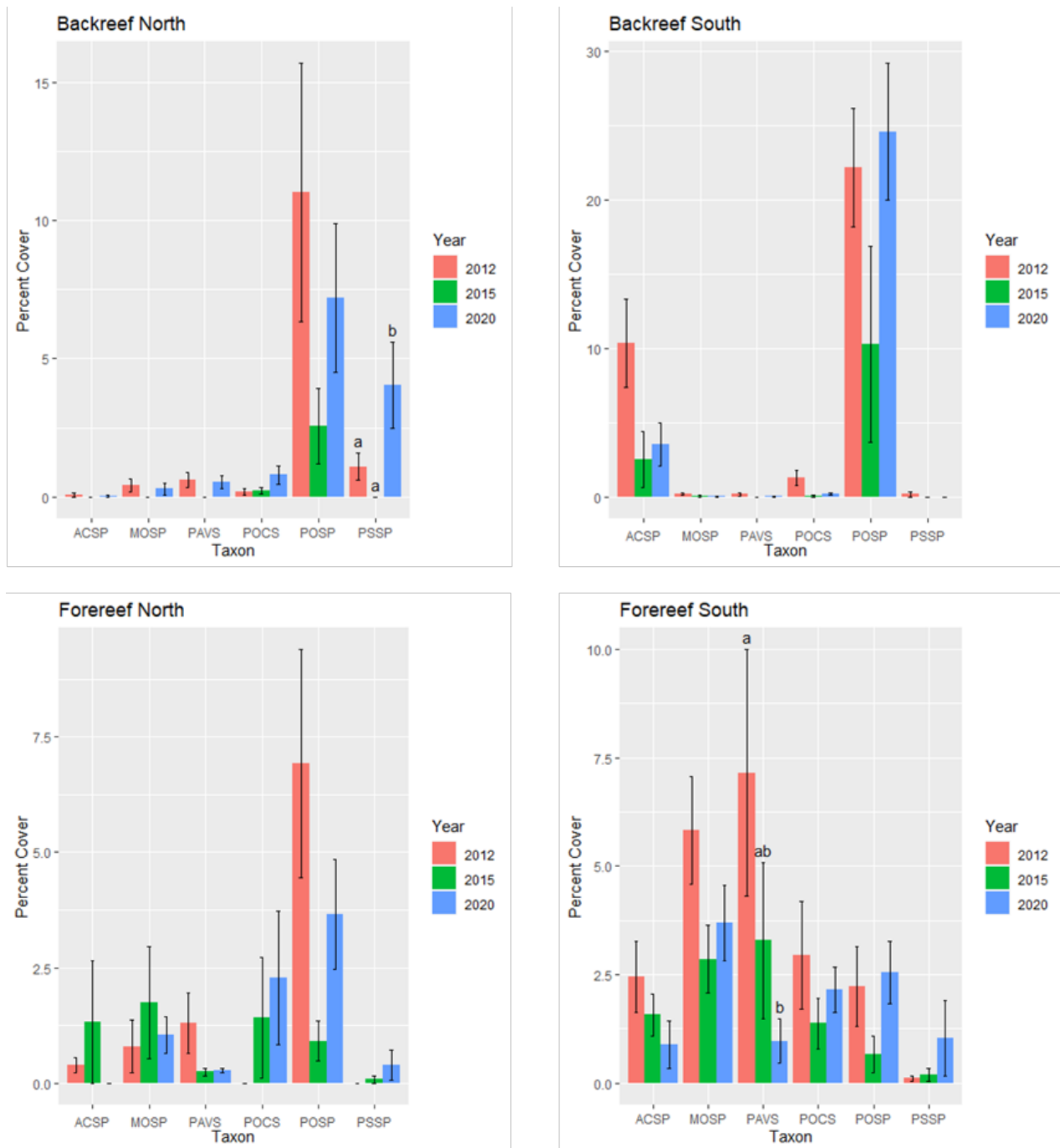
Figure 3. Fagaʻalu reef mean cover (± SE) of the most abundant coral taxa by stratum across survey years (ACSP: *Acropora*, MOSP: *Montipora*, PAVS: *Pavona*, POCS: *Pocillopora*, POSP: *Porites*, and PSSP: *Psammocora*). Letter groups above the bars indicate significant differences among years within taxa, where shared letters indicate no statistical difference between years and differing letters indicate significant difference (Dunn's test, p < 0.05). If no letters are present for a functional group, the Kruskal-Wallis test was not significant.

Adult and juvenile coral community composition

Significant differences in adult colony densities between strata were detected by the PERMANOVA analysis (pairwise post hoc analyses, $p < 0.01$), but none between years or interaction between factors (Table 5). This is graphically illustrated by the non-metric multidimensional scaling plot (nMDS) (Figure 4a), such that sites that are more similar in coral composition appear closer together on the plot. Although *Acropora*, *Montipora*, *Pavona*, *Pocillopora*, *Porites*, and *Psammocora* genera accounted for 60%–65% of the density differences among strata (SIMPER), none varied significantly among the three survey years (Figure 4b, Figure 5). The lack of interactions between the factors of year and stratum indicates that the Faga'alu reef has clear, measurable coral community differences among the different strata, and that those differences have remained unchanged over the duration of these surveys.

Table 5. PERMANOVA results table for adult coral colony densities (colonies/m². df: degrees of freedom; SS: sum of squares; MS: mean sum of squares; Pseudo-F: F value by permutation; P(perm): p-value based on permutations.

Adult colony density					
Source	df	SS	MS	Pseudo-F	P(perm)
YEAR	2	0.263	0.131	1.904	0.060
STRATUM	3	3.757	1.252	18.123	**<0.001**
YEAR*STRATUM	6	0.419	0.069	1.011	0.444
Residuals	77	5.322	0.069		
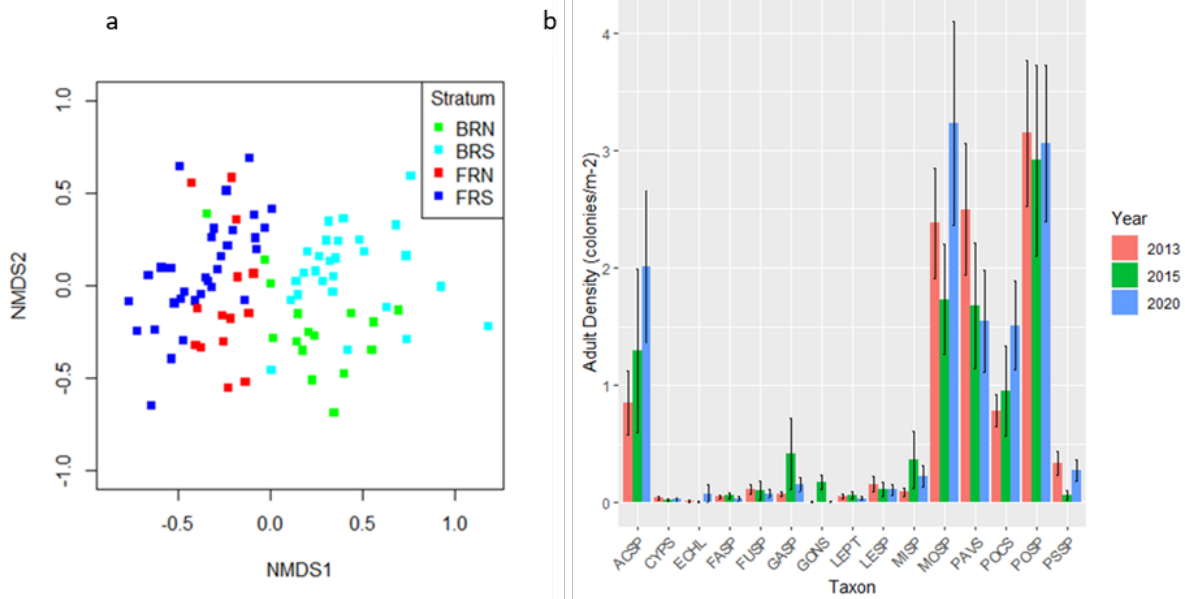
Figure 4. Adult coral colony densities (colonies/m²) (a) nMDS plot illustrating the colony density multivariate dissimilarly between strata. (b) Comparison of mean colony density (± SE) between survey years highlighting the elevated level of temporal variation within taxa. See Tables 1 and 2 for stratum and taxa abbreviations.
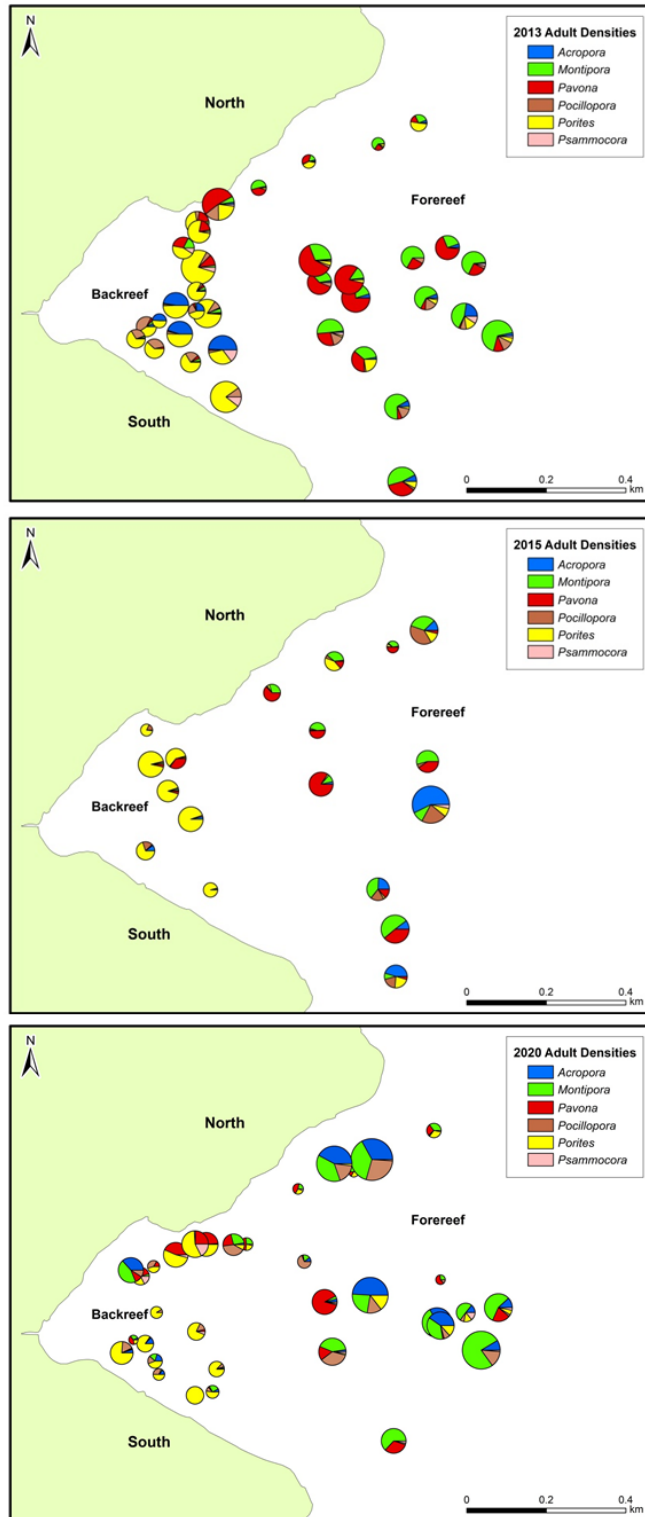
Figure 5. Site-level adult coral colony densities (colonies/m^2) for the most abundant coral taxa by survey year.

Mean total juvenile density increased across the study period from 1.80 ± 0.23 colonies/m^2 in 2013 to 4.77 ± 0.80 colonies/m^2 in 2020, with both 2015 and 2020 densities significantly exceeding baseline values measured in 2013 (one-way Welch's ANOVA, $F_{2,86} = 9.4$, $p < 0.001$; Figure 6).

Figure 6. Mean juvenile coral colony density (colonies/m^2 ± SE) between survey years. Letters above the bars indicate significant differences among years, where shared letters indicate no statistical difference between years and differing letters indicate significant difference (Dunn's test, p < 0.05).

A multivariate PERMANOVA (Table 6) indicated that juvenile densities varied significantly between years and strata. As with the univariate analysis of total juvenile density, multivariate juvenile densities differed between survey years 2013 and 2020 (pairwise post-hoc comparisons, $p = 0.027$) with *Montipora*, *Pavona*, *Pocillopora*, *Porites*, and *Acropora* accounting for more than 70% of the temporal difference (SIMPER). Pairwise post-hoc comparisons yielded significant differences among all strata except for the FRN and FRS, and the BRN and FRN, and the same five taxa comprised 55%–77% of the spatial juvenile colony density variability (SIMPER). Similar to the adult colony densities, the lack of interactions between survey years and strata corroborates that juvenile coral colony assemblage on the Faga'alu reef also follows distinct spatial patterns which have remained fairly unchanged over the study period. The spatial multivariate dissimilarity and the temporal distribution of juvenile colony density are illustrated in Figures 7 and 8.

Table 6. PERMANOVA results table for juvenile coral colony densities (colonies/m^2). df: degrees of freedom; SS: sum of squares; MS: mean sum of squares; Pseudo-F: F value by permutation; P(perm): p-value based on permutations.

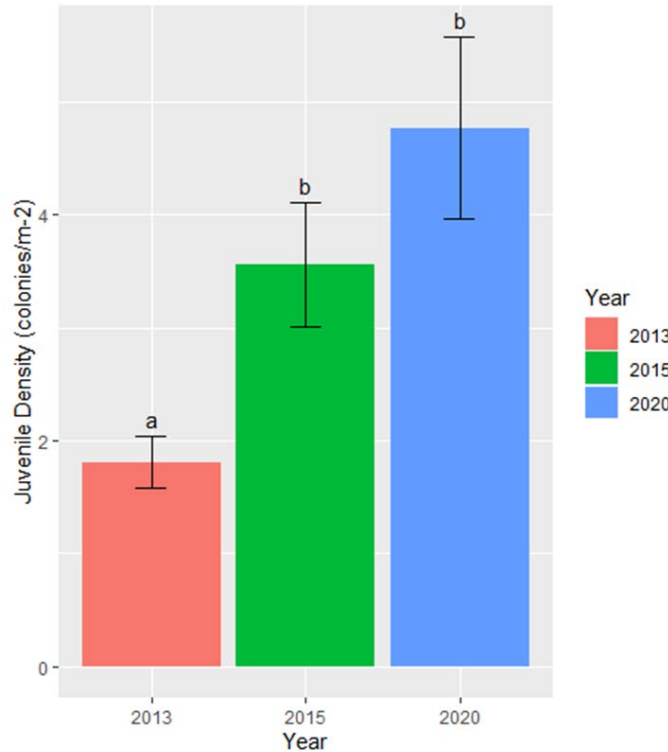| Juvenile colony density					
Source	df	SS	MS	Pseudo-F	P(perm)
YEAR	2	0.710	0.355	3.362	**<0.001**
STRATUM	3	3.824	1.274	12.067	**<0.001**
YEAR*STRATUM	6	0.853	0.142	1.347	0.145
Residuals	77	8.133	0.105		
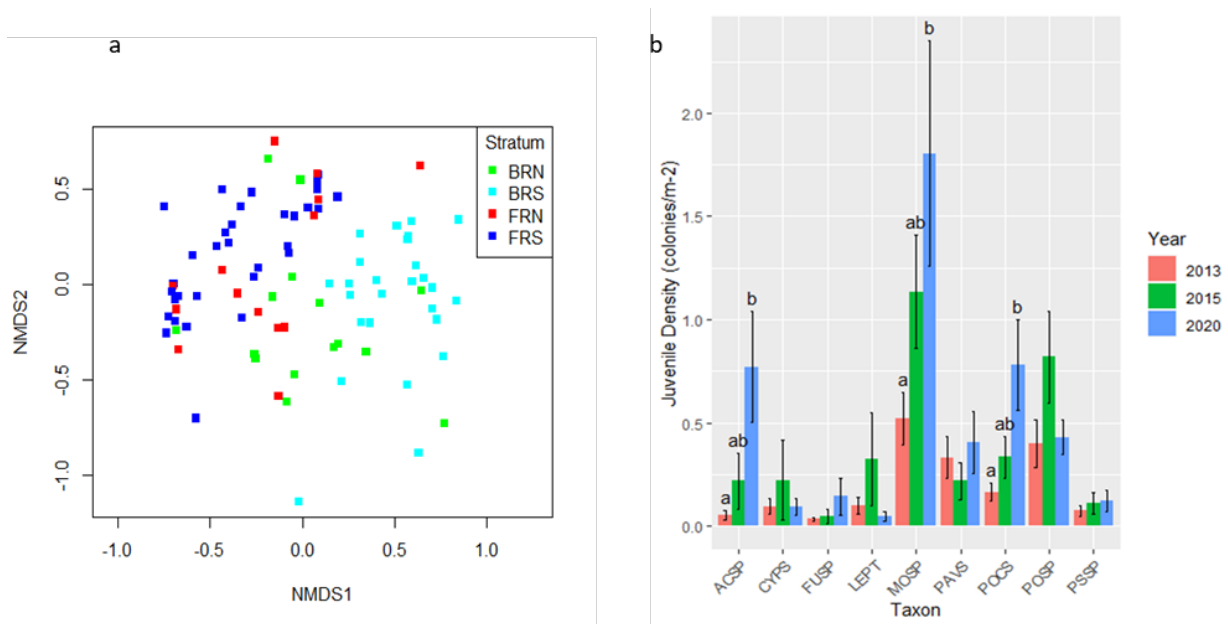
Figure 7. Juvenile coral colony densities (colonies /m^2). (a) nMDS plot illustrating colony density multivariate dissimilarly between strata. (b) Comparison of mean colony density (± SE) between survey years highlighting the elevated level of temporal variation within taxa. Letter groups above the bars indicate significant differences among years within taxa, where shared letters indicate no statistical difference between years and differing letters indicate a significant difference (Dunn's test, p < 0.05). If no letters are present for a functional group, the Kruskal-Wallis test was not significant. See Table 1 and Table 2 for stratum and taxa abbreviations.
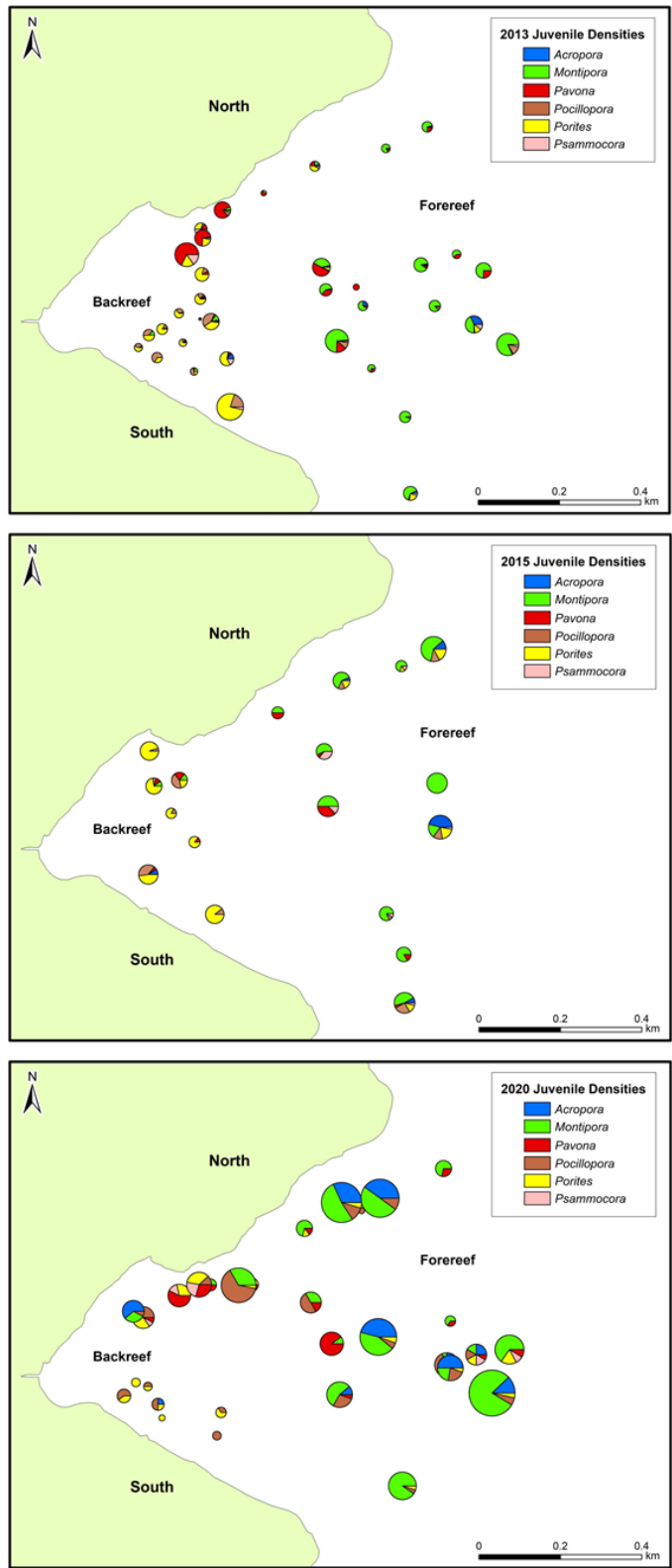
Figure 8. Site-level juvenile coral colony densities (colonies/m^2) for the most abundant coral taxa by survey year.

Univariate permutation ANOVA (999 permutations) and pairwise comparisons showed that significant mean juvenile colony density increases of *Acropora* (p < 0.01), *Montipora* (p = 0.01), and *Pocillopora* (p = 0.01) drove the pattern of total juvenile density change over time (Figure 7b).

Coral colony size and frequency distribution

With the exception of *Psammocora,* mean colony size of the most abundant taxa, varied significantly over time (p < 0.05; Kruskal-Wallis and post hoc Dunn's tests; Figure 9).
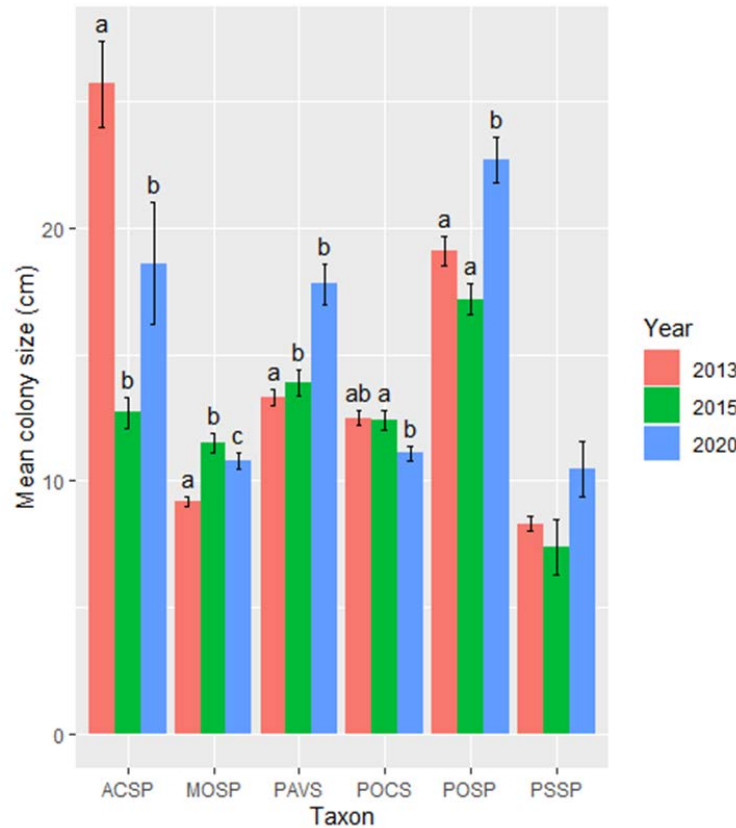
Figure 9. Mean colony size (± SE) of the most abundant coral taxa across survey years and strata (ACSP: *Acropora*, MOSP: *Montipora*, PAVS: *Pavona*, POCS: *Pocillopora*, POSP: *Porites*, and PSSP: *Psammocora*). Letter groups above the bars indicate significant differences among years within a functional group, where shared letters indicate no statistical difference between years and differing letters indicate significant difference (Dunn's test, p < 0.05). If no letters are present for a functional group, the Kruskal-Wallis test was not significant.

Acropora mean colony size decreased significantly in 2015 and 2020 compared to baseline levels from 2013 (Figure 9), likely reflecting the greater input of juveniles into the populations in these years (Figure 7b). *Acropora* size frequency distributions (SFD) also reflect this pattern, showing a significant shift of the entire SFD towards smaller colonies over time (Table 7, Figure 10), yet no overall change in the shape of that SFD (i.e., standardized KDE analysis, Table 7).

Table 7. Results of the size frequency distribution Kernel Density analyses on the most abundant coral taxa.

Genus	SFD: KDE			SFD: KDE standardized		
	2013–2015	2015–2020	2013–2020	2013–2015	2015–2020	2013–2020
Acropora	$p < 0.001$	$p = 0.002$	$p < 0.001$	$p = 0.426$	$p = 0.090$	$p = 0.564$
Montipora	$p < 0.001$	$p = 0.088$	$p = 0.014$	$p = 0.016$		$p = 0.034$
Pavona	$p < 0.001$	$p = 0.090$	$p < 0.001$	$p = 0.284$		$p = 0.030$
Pocillopora	$p < 0.001$	$p = 0.054$	$p < 0.001$	$p = 0.270$		$p = 0.566$
Porites	$p = 0.008$	$p = 0.886$	$p < 0.001$	$p = 0.006$		$p = 0.032$
Psammocora	$p = 0.036$	$p = 0.140$	$p = 0.418$	$p = 0.970$		

For the *Montipora*, colony sizes were on average larger in 2015 compared to 2013 and 2020 (Fig 9). Mean colony size dropped between 2013 and 2020, likely driven by an increase in juvenile density over this time period (Figure 7b). The SFD in 2015 and 2020 differed significantly from 2013 in both the position and shape of the SFD indicating both larger colonies in 2015 and 2020, but also greater proportions of juvenile colonies in the population in these later years (Figure 10, Table 7).

Colonies of *Pavona* increased in size throughout the study period, with significantly larger colonies in 2015 and 2020 compared to 2013 (Fig 9). The SFD analysis also shifted to larger colonies in 2015 and 2020 compared to 2013, mainly due to a greater proportion of mid-sized colonies (Figure 10, Table 1).

Porites colony size increased in 2020 compared to 2013 and 2015 (Figure 9). The increase trend in *Porites* juveniles from 2013 to 2015 (Figure 7b) was captured as a significant change in the position and shape of the SFD over this time (Figure 10, Table 7). Juvenile inputs to the population declined by 2020 (Figure 7b), yet total density remained constant and mean size increased as the colonies present in 2015 survived and grew into larger size classes by the 2020 survey.

Of the six most abundant coral genera, *Pocillopora* was the only taxon to show a net colony size reduction between 2015 and 2020 (Figure 9). This was a result of the size frequency distribution shifting toward mid-sized colonies as the proportion of larger colonies (>30 cm) decreased (Figure 10, Table 7), while juvenile densities significantly increased throughout the study period (Figure 7b).

Finally, mean colony sizes of *Psammocora* did not change over time and there were no differences in standardized SFD in any year indicating the shape of the SFD for these genera was stable over the survey years (Figure 10, Table 7). A leftward shift in position of the SFD from 2013 to 2015 was observed, largely reflecting less mid-sized colonies in the population in 2015.
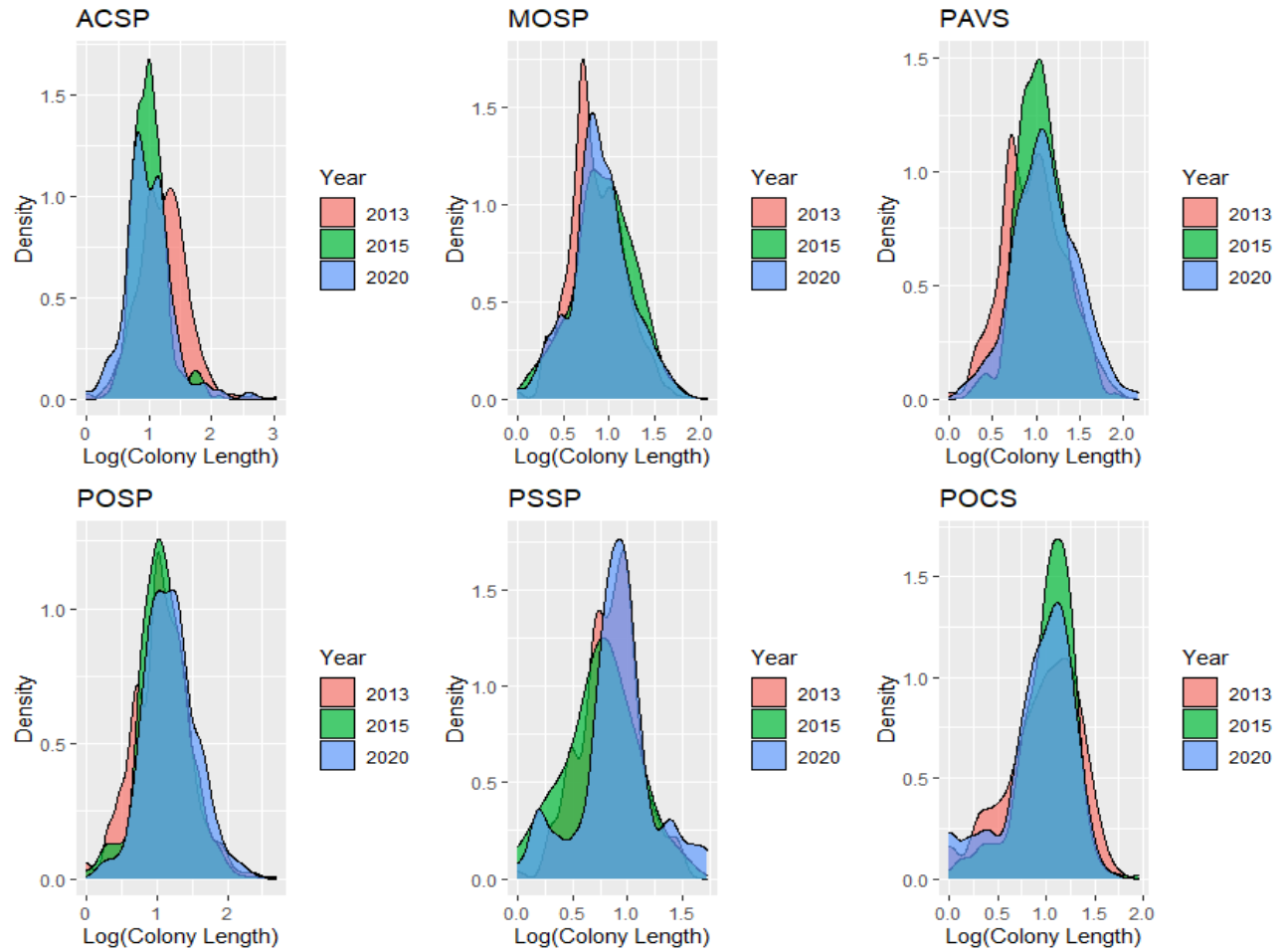
Figure 10. Size frequency distribution schematic for the most abundant coral taxa across survey years (ACSP: *Acropora*, MOSP: *Montipora*, PAVS: *Pavona*, POCS: *Pocillopora*, POSP: *Porites*, and PSSP: *Psammocora*).

Discussion

Our analyses highlight several findings and trends. (1) There was a 12% post-intervention, reef-wide decrease in macroalgae cover. (2) Despite the 2015 reduction in coral cover, likely driven by the 2015 bleaching event, coral cover recovered to pre-disturbance levels by 2020, underpinning the resilient nature of the Faga'alu coral assemblages. (3) No temporal changes in adult colony densities were documented, which was driven by the large within-stratum coral community variability. (4) Increased juvenile density of *Acropora, Montipora,* and *Pocillopora* in 2020 compared to the pre-intervention levels suggests favorable larval recruitment; active larval recruitment and post-settlement survival are fundamental to the persistence of coral populations. (5) Over time, *Montipora* and *Pavona* trended toward larger colony sizes, which is important because larger colonies promote increased resilience and sexual reproductive output.

It has been extensively recognized that coral reef macroalgal proliferation displaces coral colonies through a variety of physical and chemical mechanisms that impact all stages of the coral life cycle (Box and Mumby 2007; Chadwick and Morrow 2011; Clements et al. 2018). Conversely, macroalgal reductions decrease coral-algal competitive interactions for space, reduce algal-mediated microbe-induced coral disease and mortality (Smith et al. 2006; Wild et al. 2014), and promote the recruitment and post-settlement survival of corals (Harrington et al. 2004; Ritson-Williams et al. 2014). These mechanisms are fundamental to the persistence of coral reef habitats. Macroalgal distribution and abundance reflect nutrient supply and herbivory levels (Smith et al. 2001; De'ath and Fabricius 2010; Fong and Paul 2011). As such, a transition from algal to coral dominance is possible (e.g., Kane'ohe Bay, Hawai'i; Smith et al. 1981) to the extent that the environmental stress of allochthonous nutrient enrichment can be reduced (Dudgeon et al. 2010; Bahr et al. 2015).

The reef wide macroalgae cover loss documented in our study is noteworthy because: (1) reef macroalgal cover is known to vary across nutrient gradients and therefore is recommended as a bioindicator of water quality (Fabricius 2005; Diaz-Pulido and McCook 2008; Cooper et al. 2009); (2) declines in macroalgae percent cover occurred post-intervention and were first noticed on the BRN, which is directly adjacent to the Faga'alu stream outflow and the area severely affected by chronic increased sediment loading (Shuler et al. 2019). Increased macroalgae is also linked to sediment loading via increased nutrient inputs from terrestrial runoff (McManus and Polsenberg 2004; Fabricius 2011); and (3) the reef wide macroalgae reduction suggests a consistent decline over time. Thus, in the absence of any known changes to the Faga'alu reef herbivore community (Alice Lawrence, pers. comm.), we propose that the observed macroalgae reductions may be associated with changes in water quality due to reduced nutrients and suspended sediment. Water quality and herbivore community assessments are indicted to confirm this and future management outcome determinations.

Our results also report a net 12% coral cover reduction between 2012 and 2015, followed by an increase comparable to the pre-intervention levels in 2020. These substantial variations in coral cover occurred within a relatively short time period. We suggest the 2015 widespread bleaching event and subsequent mortality could be implicated (CRAG unpubl. data; Douglas Fenner unpubl. data; Vargas-Ángel et al. 2019a). While not catastrophic, sparse patchy bleaching and mortality of *Porites cylindrica* and *Acropora formosa* were observed along the shallow backreef habitat as early as February 2015 during the peak of the event (Douglas Fenner unpub. data), as

well as this study's coral 2015 surveys. This is supported by the field data indicating that on the BRS *Acropora* and *Porites* cover dropped from 10.4% to 2.5% and from 22.2% to 10.3% between 2012 and 2015, respectively. A concomitant cover reduction of *Porites* (11.0% to 2.6%) was also noted on the BRN. A subsequent return to pre-intervention coral cover levels by 2020 underpins the resilience and fast growing nature of the corals affected by the bleaching event. This is paramount because encrusting macroalgae and turf algae have been reported to readily colonize open space following post-bleaching coral mortality (Vargas-Ángel et al. 2019b). Additionally, the qualities of fast growth, ability to out-compete other coral species for living space, and tolerance to environmental impacts allow *P. cylindrica* to be a major reef-building coral as well as a prime candidate species for transplantation efforts (Seebauer 2001).

Research has indicated that patterns of adult and juvenile coral assemblages tend to be highly correlated not only in abundance but also composition (Adjeroud et al. 2019). This link suggests either that recruitment drives the density and spatial distribution of adults, or a process where the density and fecundity of adult colonies drives recruitment (Miller et al. 2000; Penin and Adjeroud 2013; Adjeroud et al. 2017). Our results revealed that the temporal coral population patterns on the Faga'alu reef were strongly driven by short-term changes in colony size and recruitment (based on juvenile density). More specifically, size reductions for the *Acropora*, *Montipora*, and *Pocillopora* were driven by juvenile colony density increases and net colony growth for *Pavona* and *Porites.* The increase in juvenile coral density, particularly for three dominant genera characterized by competitive and weedy life histories (Darling et al. 2012), may have been facilitated by the observed reduction in macroalgal cover throughout the bay (Edmunds and Carpenter 2001; Box 2007), and is a positive indicator of improved coral conditions.

There are challenges and limitations presented in this report worth summarizing. First, it is important to note that our study revealed a high level of spatial heterogeneity, highlighting the distinctiveness of the biological assemblages that characterize each stratum—*Porites* and *Psammocora* dominating on the BRN, thickets of *Porites* and *Acropora* on the BRS, *Porites* and *Pocillopora* on the FRN, and an amalgam of *Montipora*, *Pocillopora* and *Porites* on the FRS. Despite the relatively short time and small spatial scale of our study, our results suggest that spatial coral community composition variation is driven by the bay's geomorphology and the associated light, depth, water circulation patterns in addition to the attendant chronic (i.e., land-based sources of pollution) and acute (e.g., bleaching, storms) impact regimes. Also, the ecological response variables in this study; i.e., benthic cover, colony densities, colony sizes and frequency distribution, were generated based on surveys conducted during three time periods (2012–2013, 2015, and 2020) implementing slightly modified and improved field methodologies and sampling effort, which preclude broad conclusive statements pertaining to spatiotemporal trends. Finally, we generated coral colony metrics (i.e., mean colony densities, mean size, and SFDs) from pooled, genus-level groupings on data with a disparate number of species between sites and contrasting life history strategies. By combining all coral species for data analysis, species-specific responses that could more unambiguously inform status and trends may be masked. Although there is evidence of positive temporal benthic and coral community changes, those need to be cautiously discussed in the context of coral community recovery.

Provided that environmental conditions return to a pre-established state, time-scales for natural recovery of coral reefs are in the order of a few years to several decades, depending on the type

and degree of impact, species affected, recruitment capacity, and time required to achieve the measurable physical environmental goals (Erftemeijer et al. 2012). In addition, recovery is generally poorer following chronic sources of stress (e.g., sedimentation, overfishing) than acute disturbances (e.g., hurricanes, bleaching) (Connell 1997; Mumby and Steneck 2011). Decades of natural and anthropogenic impacts have contributed to shape the current makeup of the Faga'alu reef; accordingly, long-term and sustained stress mitigation will be necessary to restore ecological function. Moving forward we provide the following recommendations:

1. It is fundamental that specific quantifiable management outcomes be identified; for example, a measurable percent change in cover of relevant functional groups, or a measurable positive changes in the adult and juvenile coral community.

2. Herbivore densities/biomass assessments are recommended to further contextualize macroalgal community changes.

3. Reef-wide, transect-based assessments are appropriate at providing a spatially comprehensive valuation of the survey domain. However, at this point in time we suggest that the inherent spatial heterogeneity be used to inform the future monitoring sampling design and strategy to facilitate assessing the desired management outcomes. This can be achieved by establishing representative permanent transects/plots in the relevant strata; e.g., a selection from 2020 survey transects sites in the BRN and FRN.

4. Focus the assessment of adult colony density and size to live coral tissue surfaces rather than individual colony boundaries (which is particularly difficult on those taxa with indeterminate, encrusting and thicket forming growth morphologies). This will expedite colony density data collection and interpretation in upcoming monitoring efforts.

5. Coupled with the aforementioned, implement photo-quadrats/mosaics collection and image analysis techniques to reduce observer bias and simplify the estimation of live tissue surfaces and partial mortality, as well as trace the trajectories of individual colonies, and streamline the assessment of change over time.

6. Identify and select an array of 'sentinel' species from which to ascertain measureable responses.

7. Collect complementary time-series of environmental data (e.g., sediment accumulation, turbidity, and nutrient concentrations) to contextualize spatially explicit biotic and abiotic change.

8. Implement novel data analysis approaches that depart from traditional univariate comparisons incorporating alternate spatial scales that take into account ecological and disturbance gradients.

This work culminates the initial phase of efforts undertaken to evaluate the ecological outcomes of management interventions conducted at the Samoa Maritime quarry in 2014. Based on the timeline requested by local resource management practitioners for the submission of this report, we deliberately focused our analyses exclusively at detecting measurable change within the pre-established spatiotemporal framework; i.e., reef strata and pre- and post-intervention. A

forthcoming analysis of the Fagaʻalu biological monitoring data set will attempt to couple the measurable physical and biological gradients to better discriminate and ascribe change over space and time. This work plans to incorporate all available environmental and driver data for Fagaʻalu Bay, including sedimentation, turbidity, and nutrient concentrations among others.

Acknowledgements

References

Adjeroud M, Kayal M, Penin L. 2017. Importance of Recruitment Processes in the Dynamics and Resilience of Coral Reef Assemblages. In: Rossi S., Bramanti L., Gori A., Covadonga O. (eds) Marine Animal Forests: The Ecology of Benthic Biodiversity Hotspots. Springer International Publishing, Cham, pp 549–569

Adjeroud M, Poisson E, Peignon C, Penin L, Kayal M. 2019. Spatial patterns and short-term changes of coral assemblages along a cross-shelf gradient in the Southwestern Lagoon of New Caledonia. Diversity 11:21

Bahr KD, Jokiel PL, Toonen RJ. 2015. The unnatural history of Kāne'ohe Bay: coral reef resilience in the face of centuries of anthropogenic impacts. PeerJ 3:e950 https://doi.org/10.7717/peerj.950

Bartley R, Bainbridge ZT, Lewis SE, Kroon FJ, Wilkinson SN, Brodie JE, Silburn DM.2014. Relating sediment impacts on coral reefs to watershed sources, processes and management: A review. Sci Total Environ 468–469:

Baums IB.2008. A restoration genetics guide for coral conservation. Mol Ecol. 2008;17(12):2796-2811. doi:10.1111/j.1365-294X.2008.03787.x

Bégin C, Schelten CK, Nugues MM, Hawkins J, Roberts C, Côté IM (2016) Effects of protection and sediment stress on coral feefs in Saint Lucia. PLoS One 11 (2): e0146855. https://doi.org/10.1371/journal.pone.0146855

Beijbom O, Edmunds PJ, Roelfsema C, Smith J, Kline DI, Neal BP, Dunlap MJ, Moriarty V, Fan TY, Tan CJ, Chan S, Treibitz T, Gamst A, Mitchell BG, Kriegman D. 2015. Towards automated annotation of benthic survey images: Variability of human experts and operational modes of automation. PLoS One 10(7):e0130312. DOI: 10.1371/journal.pone.0130312

Box SJ, Mumby PJ. 2007. Effect of macroalgal competition on growth and survival of juvenile Caribbean corals. Mar Ecol Prog Ser 342:139–149

Burke L. 2011. Reefs at Risk Revisited, Reefs at Risk Series. World Resources Institute, Washington, DC, USA,

Chadwick N, Morrow K. 2011. Competition Among Sessile Organisms on Coral Reefs. In: Dubinsky Z., Stambler N. (eds) Coral Reefs: An Ecosystem in Transition. Springer Netherlands, Dordrecht. pp 347–371

Clements CS, Rasher DB, Hoey AS, Bonito VE, Hay ME. 2018. Spatial and temporal limits of coral-macroalgal competition: The negative impacts of macroalgal density, proximity, and history of contact. Mar Ecol Prog Ser 586:11–20

Connell JH. 1997. Disturbance and recovery of coral assemblages. Coral Reefs 16:101–113

Darling ES, Alvarez-Filip L, Oliver TA, Mcclanahan TR, Côté IM. 2012. Evaluating life-history strategies of reef corals from species traits. Ecol Lett 15:1378–1386

De'ath G, Fabricius K. 2010. Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. Ecol Appl 20:840–850

Diaz-Pulido, G. and McCook, L. 2008 Macroalgae (Seaweeds) in Chin A (ed) The State of the Great Barrier Reef On-line, Great Barrier Reef Marine Park Authority, Townsville. (Viewed on June 18, 2020)

Dudgeon S, Aronson R, Bruno J, Precht W. 2010. Phase shifts and stable states on coral reefs. Mar Ecol Ser - Mar Ecol Prog Ser 413:201–216

Edmunds PJ, Carpenter RC. 2001. Recovery of *Diadema antillarum* reduces macroalgal cover and increases abundance of juvenile corals on a Caribbean reef. Proc Natl Acad Sci 98:5067–5071

Erftemeijer PLA, Riegl B, Hoeksema BW, Todd PA. 2012. Environmental impacts of dredging and other sediment disturbances on corals: A review. Mar Pollut Bull 64:1737–1765

Fabricius KE. 2005. Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. Mar Pollut Bull 50:125–146

Fabricius KE. 2011. Factors Determining the Resilience of Coral Reefs to Eutrophication: A Review and Conceptual Model. In: Dubinsky Z., Stambler N. (eds) Coral Reefs: An Ecosystem in Transition. Springer Netherlands, Dordrecht, pp 493–505

Fenner D, Speicher M , Gulick S, Aeby G, Aletto SC, Anderson P, Carroll BP, DiDonato EM, DiDonato GT, Farmer V, Gove J , Gulick S, Houk P, Lundblad E, Nadon M, Riolo F, Sabater, MG, Schroeder R, Smith E, Speicher M, Tuitele C, Tagarino A, Vaitautolu S, Vaoli E, Vargas-Angel B, Vroom P. 2008. The state of coral reef ecosystems of American Samoa. In: The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States, p. 307–351.

Fong P, Paul VJ. 2011. Coral Reef Algae BT - Coral Reefs: An Ecosystem in Transition. In: Dubinsky Z., Stambler N. (eds) Springer Netherlands, Dordrecht, pp 241–272

Harrington L, Fabricius K, De'ath G, Negri A. 2004. Recognition and selection of settlement substrata determine post-settlement survival in corals. Ecology 85:3428–3437

Holst-Rice S, Messina AM, Biggs TW, Vargas-Angel B, Whitall D. 2016. Baseline ssessment of Faga'alu Watershed: A Ridge to Reef Assessment in Support of Sediment Reduction Activities and Future Evaluation of their Success. 44p

Kroon FJ, Schaffelke B, Bartley R. 2014. Informing policy to protect coastal coral reefs: Insight from a global review of reducing agricultural pollution to coastal ecosystems. Mar Pollut Bull 85:33–41

Langlois TJ, Fitzpatrick BR, Fairclough D V, Wakefield CB, Hesp SA, McLean DL, Harvey ES, Meeuwig JJ. 2012. Similarities between line fishing and baited stereo-video estimations of length-frequency: Novel application of Kernel density estimates. PLoS One 7:1–9

Latinis DK, Moore J, Kennedy J. 1996. Archaeological Survey and Investigations Conducted at the Faga'alu Quarry, Ma'oputasi County, Tutuila, American Samoa, February 1996: Prepared for George Poysky, Sr.,

Lozada-Misa P, Schumacher BD, Vargas-Ángel B. 2017.) Analysis of Benthic Survey Images via CoralNet: A Summary of Standard Operating Procedures and Guidelines. Pacific Islands Fisheries Science Center Administrative Report, H-17-02, 175 p. https://doi.org/V5/10.7289/V5/AR-PIFSC-H-17-02

McManus JW, Polsenberg JF. 2004. Coral–algal phase shifts on coral reefs: Ecological and environmental aspects. Prog Oceanogr 60:263–279

Messina AM, Biggs TW. 2016. Contributions of human activities to suspended sediment yield during storm events from a small, steep, tropical watershed. J Hydrol 538:726–742

Miller MW, Weil E, Szmant AM (2000) Coral recruitment and juvenile mortality as structuring factors for reef benthic communities in Biscayne National Park, USA. Coral Reefs 19:115–123

Mumby PJ, Steneck RS. 2011. The Resilience of Coral Reefs and Its Implications for Reef Management. In: Dubinsky Z., Stambler N. (eds) Coral Reefs: An Ecosystem in Transition. Springer Netherlands, Dordrecht, pp 509–519

Nugues MM, Roberts CM. 2003. Partial mortality in massive reef corals as an indicator of sediment stress on coral reefs. Mar Pollut Bull 46:314–323

Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin P, O'Hara R, Simpson G, Solymos P, Stevens H, Szöcs E, Wagner H. 2018. Vegan: Community Ecology Package. Ordination methods, diversity analysis and other functions for community and vegetation ecologists. Version 2.5-1. URL https://CRAN.R-project.org/package=vegan.

Penin L, Adjeroud M. 2013. Relative importance of recruitment and post-settlement processes in the maintenance of coral assemblages in an insular, fragmented reef system . Mar Ecol Prog Ser 473:149–162

Philipp E, Fabricius K. 2003. Photophysiological stress in scleractinian corals in response to short-term sedimentation. J Exp Mar Bio Ecol 287:57–78

R Core Team. 2013. R: A Language and Environment for Statistical Computing.

Restrepo JD, Park E, Aquino S, Latrubesse EM. 2016. Coral reefs chronically exposed to river sediment plumes in the southwestern Caribbean: Rosario Islands, Colombia. Sci Total Environ 553:316–329

Risk MJ, Edinger E. 2011. Impacts of Sediment on Coral Reefs. In: Hopley D. (eds) Encyclopedia of Modern Coral Reefs: Structure, Form and Process. Springer Netherlands, Dordrecht, pp 575–586

Ritson-Williams R, Arnold SN, Paul VJ, Steneck RS. 2014. Larval settlement preferences of *Acropora palmata* and *Montastraea faveolata* in response to diverse red algae. Coral Reefs 33:59–66

Rogers CS. 1990 Responses of coral reefs and reef organisms to sedimentation. Mar Ecol Prog Ser 62:185–202

Seebauer J. 2001. Zoology of *Porites cylindrica*: potential for use in reef-rehabilitation transplantation efforts. SUNY Geneseo J Sci Math 2:26–34

Shuler C, Amato D, Veronica Gibson V, Baker L, Olguin A, Dulai H, Smith C, Alegado R. 2019. Assessment of terrigenous nutrient loading to coastal ecosystems along a human land-use gradient, Tutuila, American Samoa. Hydrology 6:18

Smith J, Smith C, Hunter C. 2001. An experimental analysis of the effects of herbivory and nutrient enrichment on benthic community dynamics on a Hawaiian reef. Coral Reefs 19:332–342

Smith JE, Shaw M, Edwards RA, Obura D, Pantos O, Sala E, Sandin SA, Smriga S, Hatay M, Rohwer FL. 2006. Indirect effects of algae on coral: algae-mediated, microbe-induced coral mortality. Ecol Lett 9:835–845

Smith SV, Kimmerer WJ, Laws EA BR. 1981. Kaneohe Bay sewage diversion experiment: perspectives on ecosystem responses to nutritional perturbation. Pacific Sci 35:279–395

Smith TB, Nemeth RS, Blondeau J, Calnan JM, Kadison E, Herzlieb S. 2008. Assessing coral reef health across onshore to offshore stress gradients in the US Virgin Islands. Mar Pollut Bull 56:1983–1991

Sully S, van Woesik R. 2020. Turbid reefs moderate coral bleaching under climate-related temperature stress. Glob Chang Biol 26:1367–1373

Swanson D, Bailey H, Schumacher B, Ferguson M, Vargas-Ángel B. 2018. Ecosystem Sciences Division standard operating procedures: data collection for rapid ecological assessment benthic surveys. . U.S. Dept. of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-71, 63p, https://doi.org/10.25923/39jh-8993.

Vargas-Ángel B, Ayotte P, Barkley H, Couch C, Halperin A, Kindinger T, Winston M. 2019a. Coral reef ecosystem monitoring report for the National Marine Sanctuary of American Samoa: 2018. Pacific Islands Fisheries Science Center, PIFSC Data Report, DR-19-040, 56 p. doi:10.25923/1sk7-vp89

Vargas-Ángel B, Huntington B, Brainard RE, Venegas R, Oliver T, Barkley H, Cohen A. 2019b. El Niño-associated catastrophic coral mortality at Jarvis Island, central Equatorial Pacific.

Coral Reefs 38:731–741

Wild C, Jantzen C, Kremb SG. 2014. Turf algae-mediated coral damage in coastal reefs of Belize, Central America. PeerJ 2:e571–e571

Winston M, Couch C, Ferguson M, Huntington B, Swanson D, Vargas-Ángel B (2019) Ecosystem Sciences Division Standard Operating Procedures: Data Collection for Rapid Ecological Assessment Benthic Surveys, 2018 Update. NOAA Tech. Memo. NOAA-TMNMFS-PIFSC-92, 66 p. doi:10.25923/w1k2-0y84.