NOVA SOUTHEASTERN UNIVERSITY

HALMOS COLLEGE OF ARTS AND SCIENCES

A rapid site selection assessment as an indicator of stony coral microfragment outplant success

By

John J. Alfirevich

Submitted to the Faculty of
Halmos College of Arts and Sciences
in partial fulfillment of the requirements for
the degree of Master of Science with a specialty in:

Marine Science

Nova Southeastern University

December 2023

Abstract:

Microfragmentation of massive stony coral species is a technique being utilized to propagate corals asexually to help restore coral reefs. Microfragmentation consists of cutting corals into 3 cm diameter or less fragments, which boosts growth rates. However, in some locations the size of microfragments make them vulnerable to parrotfish predation and benthic overgrowth, reducing survival. As such, a method to identify key site characteristics which promotes microfragment outplant success, particularly one that can be performed quickly across multiple areas is needed. A rapid site assessment conducted prior to microfragment outplanting was performed at 12 randomly selected sites within the Kristin Jacobs Coral Reef Ecosystem Conservation Area in Broward County, Florida to predict the subsequent success of outplants. The assessment quantified habitat complexity and the species richness, density, size distribution, and health of the stony coral community. Following the assessment, a grid of 42 cement mounds, each holding one individual microfragment, was established at each site. After 6-months post-outplanting, microfragment success was determined based on survival and growth rates and compared to site characteristics captured by the rapid assessment. Survival was overall > 90% with little variation found between species. Growth rates were variable among species, but all were lower than natural growth rates. The rapid assessments were able to capture site characteristics that influenced microfragment outplant success: habitat complexity, wild coral density, and the prevalence of wild coral health conditions. Despite such little variation found between sites, these characteristics can be used as indicators for outplant site selection.

Keywords: Restoration, Site Selection, Rapid Assessment, Coral Microfragmentation, Southeast Florida

Acknowledgments

This study was partially supported by a subaward from the University of Miami as part of a prime award funded by the National Fish and Wildlife Foundation with flow-through from the National Oceanic and Atmospheric Administration (NA19NOS4730148). All corals used for this study were collected under Florida Fish and Wildlife Conservation Commission (FWC) Special Activity License SAL-20-1727-SCRP.

I would like to thank my major advisor Dr. David Gilliam for his support and providing me with many opportunities throughout my time as a student. You have allowed me to start this project from the ground up and present these findings at conferences. To my committee members, Drs. Joana Figueiredo and Diego Lirman, thank you for agreeing to be part of this project, providing great feedback and advice on methods and writing. Special thanks to Dr. Nick Jones who helped me above and beyond with project design, statistics, and support through many long nights. Thanks to all of the past and present members of the Coral Reef Restoration, Assessment, and Monitoring Lab for all their help with two long days of outplanting corals, many monitoring dives, and providing feedback to several drafts and presentations. Finally, I would like to thank my family, my girlfriend Faith, and my friends for their constant support of my pursuit in this field. I would not have made it this far without any of you.

Table of Contents

Table of Figures ..... **iv**
List of Tables ..... **v**
List of Equations ..... **vi**
Introduction **1**
Methods **4**
Site selection *4*
Rapid site assessment *6*
Study species *8*
Microfragmenting *8*
Microfragment outplanting *9*
Microfragment outplant monitoring *10*
Microfragment base community *10*
Microfragment growth *10*
Statistical analyses *11*
Results **13**
Rapid assessment *13*
Microfragment survival *15*
Microfragment growth *20*
Microfragment conditions and benthic community *26*
Discussion **36**
References **41**

List of Figures

Figure 1: Study area with habitats and randomly selected sites 5
Figure 2: Diagram of site assessment set-up 6
Figure 3: Bar graph of microfragment survival rates (%) after 6-months by site 16
Figure 4: Site conditions that indicated microfragment survival 17
Figure 5: Site conditions that indicated *M. cavernosa* survival. 18
Figure 6: Site conditions that indicated *P. clivosa* survival. 19
Figure 7: Site conditions that indicated *S. siderea* survival 20
Figure 8: Net growth (cm^2) of the outplanted microfragments by site 21
Figure 9: Site conditions that indicated microfragment growth 22
Figure 10: Net growth (cm^2) of the outplanted microfragments by species 23
Figure 11: Site conditions that indicated *M. cavernosa* relative growth 24
Figure 12: Site conditions that indicated *P. clivosa* relative growth 25
Figure 13: Site conditions that indicated S. siderea relative growth 26
Figure 14: Total predation prevalence on microfragments by site at each monitoring interval 27
Figure 15: Change in base community after 6-months 28
Figure 16: Conditions and benthic taxa that effected microfragment survival. 29
Figure 17: Conditions and benthic taxa that significantly effected microfragment growth 30
Figure 18: Conditions and benthic taxa that effected *M. cavernosa* survival 31
Figure 19: Conditions and benthic taxa that effected *M. cavernosa* relative growth 32
Figure 20: Conditions and benthic taxa that effected *P. clivosa* survival 33
Figure 21: Conditions and benthic taxa that effected *P. clivosa* relative growth 34
Figure 22: Conditions and benthic taxa that effected *S. siderea* survival 35
Figure 23: Conditions and benthic taxa that effected *S. siderea* relative growth 36

List of Tables

Table 1: Microfragment information based on species, parent colony, and date fragmented 9
Table 2: Rapid assessment measurements for each site 14
Table 3: Number of wild coral colonies and percent of conditions observed at each site 15
Table 4: Calculated growth rates for each coral species (mean ± SD) 23

List of Equations

Equation 1: Relative growth, arithmetic mean radius (AMR), and linear extension equations ... 11
Equation 2: Full model equations of GLMMs examining site characteristics..... 12
Equation 3: Full model equations of GLMMs examining benthic taxa and health conditions. ... 13

Introduction:

Reef-building coral species have been in decline worldwide for several decades, primarily due to climate-change induced thermal bleaching events, which are predicted to increase in frequency and severity (Bellwood et al., 2004; Gardner et al., 2003; Hoegh-Guldberg et al., 2017; Hughes et al., 2017; Jones et al., 2020; Kuffner & Toth, 2016). Such drastic declines have resulted in a lack of recovery in many areas particularly with the slow growing massive coral species (Jones et al., 2022). To combat coral decline and increase coral abundance, enhance structural complexity, preserve coral and associated fish species, and maintain genetic diversity, reef restoration efforts have been implemented (Brostrom-Einarsson et al., 2020). One technique shown to be effective is coral gardening, a form of active restoration which involves continuously fragmenting parent colonies, allowing proliferation of genetic individuals across multiple degraded reef areas (Epstein et al., 2003; Rinkevich, 1995; Rinkevich, 2000). Coral gardening favors reef-building branching species (such as *Acropora* spp.) due to their fast growth rates and ease of fragmenting, but these species are often highly susceptible to thermal stress (Loya et al., 2001; Rinkevich, 1995). Because coral gardening targets a few growth forms and species, restoration by this method prevents restoring the diversity of coral reefs (Pratchett et al., 2015). Inclusion of other growth morphologies in reef restoration efforts, such as massive reef-building coral species, enhances diversity and structural framework, and could prove beneficial, as they have been documented to be more resistant to thermal stress (Loya et al., 2001; Pratchett et al., 2015).

Coral restoration using massive corals has less frequently been implemented because these corals have large dense skeletons that are not easily fragmented and comparatively slower growth rates (Loya et al., 2001; Page et al., 2018). To combat these issues, coral microfragmentation was developed (Forsman et al., 2015). Microfragmentation consists of cutting a coral into fragments that can be as small as a single polyp, allowing for the proliferation of genetic individuals across multiple reefs, and cutting corals this way increases the coral perimeter (the zone where new polyps are added through polyp budding) relative to its area (Lirman, 2000; Page et al., 2018). As a result, microfragmented colonies have been shown to grow significantly faster than larger fragments or whole colonies (Page et al., 2018; Schlecker et al., 2022; Soper et al., 2022;). This accelerated growth could additionally be attributed to a shift in resource allocation from reproduction to lesion repair and regrowth in mature coral colonies that are fragmented (Lirman, 2000).

Corals are typically microfragmented using a bandsaw to cut a parent coral colony into small fragments (~3 cm in diameter or less), followed by housing the microfragments in an ex situ (land based) nursery for tissue recovery and growth before outplanting (Page et al., 2018). Ex situ nurseries provide microfragments with optimal water conditions for growth and protection from predators and competition during this high-stress period of post-cutting (Forsman et al., 2015; Page et al., 2018). Despite the successes of microfragmentation ex situ, success of outplanting microfragments onto natural reefs is lower due to size-specific mortality, mostly from predation and macroalgae overgrowth (Forsman et al., 2006; Koval et al., 2020).

Habitat structural complexity and stony coral community composition at an outplant site may predict microfragment outplant success, as both may contribute to the long-term health of a site (McField & Kramer, 2007). Structurally complex sites are found to support a higher diversity of reef organisms, particularly fish (Graham & Nash, 2013; McField & Kramer, 2007). Sites with high coral density are found to have a positive effect on coral growth rates, possibly due to increased flow turbulence and increased abundance of coral-associated organisms assisting corals with disease resistance and nutrient uptake and protecting from predation and fouling (Bracken et al., 2007; Shantz et al., 2011). High coral densities on a reef also reduce the frequency of predation on individual corals, allowing them time to recover before another predation event occurs (Jayewardene et al., 2009; Shantz et al., 2011). Coral density, size structure, health, and community composition may therefore be good predictors of microfragment outplant success.

Initial outplant success is often hindered by fish predation and dislodgement, which frequently occurs in the first two weeks post-outplanting (Koval et al., 2020; Page et al., 2018; Quimpo et al., 2020; Rivas et al., 2021). Reef structural complexity is a major driver of fish distribution, with fish abundance and richness generally found to increase with increasing hardbottom habitat complexity (rugosity) (Fukunaga et al., 2020; Gratwicke & Speight, 2005; Kuffner et al., 2007). Corallivorous and herbivorous fish, such as butterflyfish (Chaetodontidae) and, particularly, parrotfish (Scaridae), affect the survival and growth of microfragments (Burkepile et al., 2019; Koval et al., 2020; Quimpo et al., 2020). It is suggested that areas with greater abundances of large-bodied herbivores and corallivores will result in higher instances of outplanted coral mortality and detachment (Quimpo et al., 2020).

Overgrowth by sponges and macroalgae also hinders the success of coral microfragment outplants. Many Caribbean reef communities have experienced increases in sponge and macroalgae abundances, resulting in overgrowth and increased competition with corals (Hughes, 1994; Jones et al., 2020; Kramer, 2003; Maliao et al., 2008; Norström et al., 2009; Steneck et al., 2019). Significant increases in macroalgal cover have been attributed to nutrient availability, decreases of herbivores (parrotfish and *Diadema* urchins), and a reduction of coral cover from disease and bleaching, which increases substrate availability (Hughes, 1994; Jones et al., 2020; Kramer, 2003; Maliao et al., 2008; Steneck et al., 2019). High macroalgae and sponge cover increases instances of direct contact with corals, which may cause partial or whole colony mortality, especially during the summer months (Lirman, 2001; Lustic et al., 2020; Paul et al., 2011; Wulff, 2006;). Overgrowth by sponges, macroalgae, and other benthic organisms negatively impact corals primarily via shading, abrasion, and allelochemicals; they are also known vectors for coral diseases (Paul et al., 2011; Wulff, 2006). Microfragments are small and thus are more vulnerable to overgrowth. Overgrowing organisms on corals can be physically removed, but that requires time and effort to accomplish. A potential solution is to harness natural herbivory of the local populations of herbivorous fish and invertebrates, which can be related to the habitat complexity and coral community of the site (Helder et al., 2022; Paul et al., 2011).

Selecting sites with high complexity and coral cover that can potentially improve microfragment outplant success would require significant efforts to survey a wide area. A solution to this could be the inclusion of rapid site assessments, which allow for a site to be observed in a quick, simple, and low-cost manner that can be highly repeatable allowing for coverage over a wide area (Bradley et al., 2009; Price & Harris, 2009). Since multiple sites can be assessed in a day, the pool of sites to be considered within the selection process is greater. Additionally, rapid assessments are performed under uniform measures and effort, which allows for widespread use in restoration efforts worldwide (Bradley et al., 2009). Restoration efforts with limited funds and personnel can use this study as a guide to aid in selecting potential outplant sites. Site characteristics found in this study that indicate greater or lesser outplant success can be targets for restoration practitioners to quickly observe without a direct need to measure and statistically analyze the site.

The purpose of this study was to determine whether specific site characteristics which can predict outplant success can be identified during a rapid pre-outplant site assessment. To assess this, I quantified specific site characteristics within one dive at each of 12 sites on Broward County, Florida reefs to make the site selection effort rapid and standardized (McKenna & Etnoyer, 2010). These characteristics were compared to the survival and growth of microfragments of three commonly used restoration species (*Montastraea cavernosa*, *Pseudodiploria clivosa*, and *Siderastrea siderea*) that were outplanted onto the sites to test the effectiveness of the rapid site assessment in predicting microfragment success. Because I will be observing these microfragments over time, a secondary purpose of understanding what specifically was responsible for driving microfragment mortality and tissue loss was included. To help answer this, microfragment health conditions and the benthic community on the outplant base were assessed. Identifying specific site characteristics that indicate microfragment success will aid restoration efforts by providing key characteristics to look for when determining the quality of a potential outplant site. Also, by identifying drivers of mortality, practitioners can gain understanding of what happens to microfragments after outplanting without need for frequent monitoring or maintenance.

Methods:

Site selection

Twelve sites were randomly selected offshore Broward County in the Kristin Jacobs Ecosystem Conservation Area (Coral ECA) in QGIS using available benthic habitat and bathymetry maps (QGIS.org, 2023; Walker et al., 2008) (Figure 1). Six sites were randomly chosen north of the Port Everglades entrance channel and six sites were randomly chosen south of the channel. The nearshore ridge complex and inner reef areas were each manually traced over bathymetry maps to create a shapefile within which to randomly generate points. All site points were at least 100 meters from each other and 1 km from the Port Everglades Channel to avoid Port Everglades expansion activities. The northern extent of potential site locations was Hillsboro Inlet, and the southern extent was the Broward/Miami-Dade County line. On each side of the channel, three sites were randomly selected on both the nearshore ridge complex and the inner reef. These reef habitats vary by distance from shore and depth. The nearshore ridge complex in Broward County is generally located within 300 m of shore with depths of 5-6 m; the inner reef, generally 1 km offshore with depths of 8-10 m (Banks et al., 2007; Banks et al., 2008; Jones, 2022).

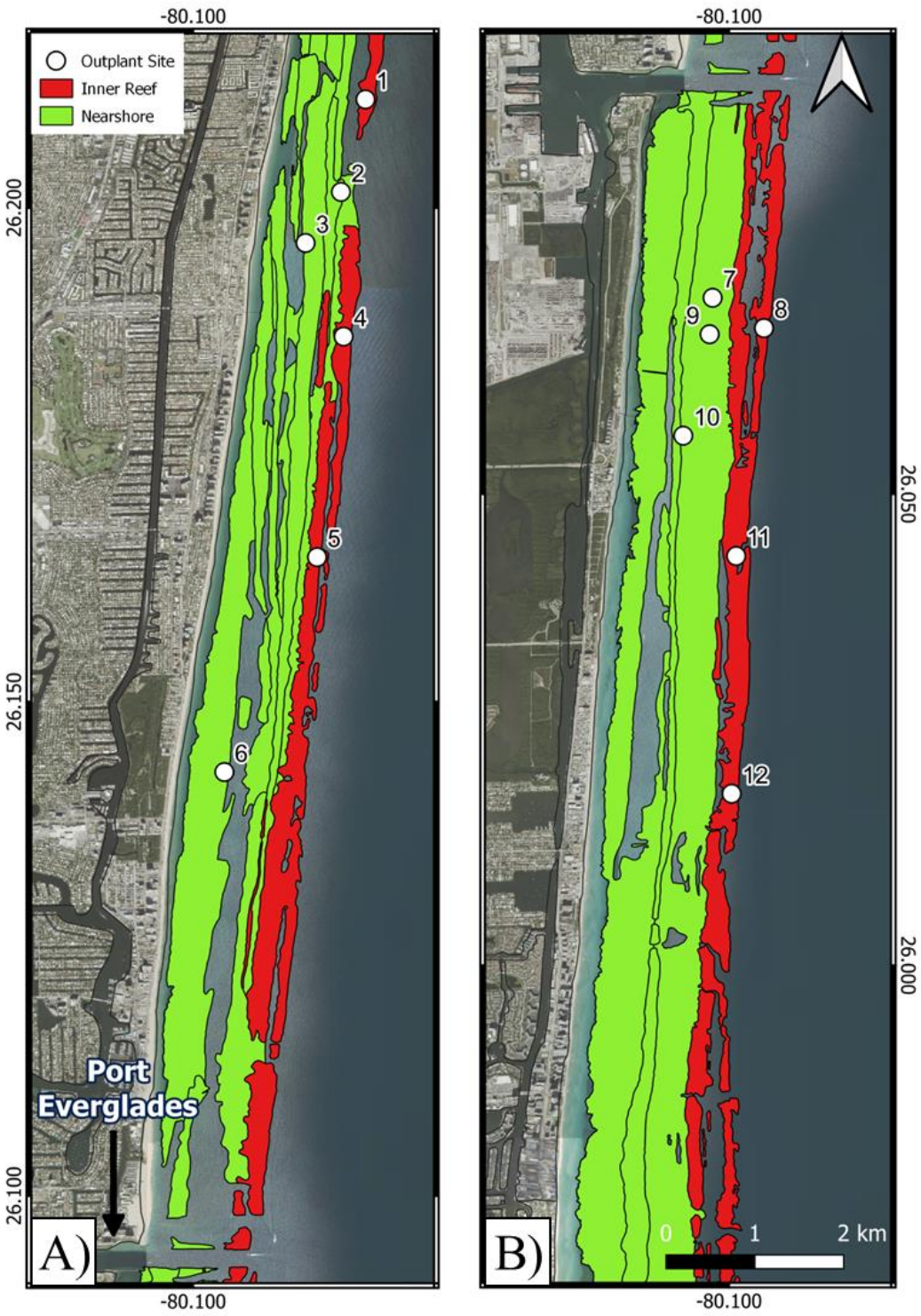

Figure 1: Study area with habitats and randomly selected sites. A) Sites north of Port Everglades (sites 1-6). B) Sites south of Port Everglades (sites 7-12).

Rapid site assessment

All sites were assessed by a team of four divers prior to outplanting microfragments (December 2022-January 2023), with a goal of completing the rapid assessments in less than 60 minutes. At each site, a 30 cm metal pin was hammered into the substrate and marked as the center point of the established site. To rapidly quantify habitat complexity, coral community composition, and stony coral colony density, size class, and health at the site, four 20 m transects that extended along each cardinal direction from the center point (north, south, east, and west) were used to divide the site into four sections: northeast, southeast, southwest, and northwest (Figure 2). A five-meter buffer from the center pin separated the north/south transects and the east/west transects. Divers recorded bottom depth at each site and conducted rugosity measurements and stony coral community surveys along the four transects.

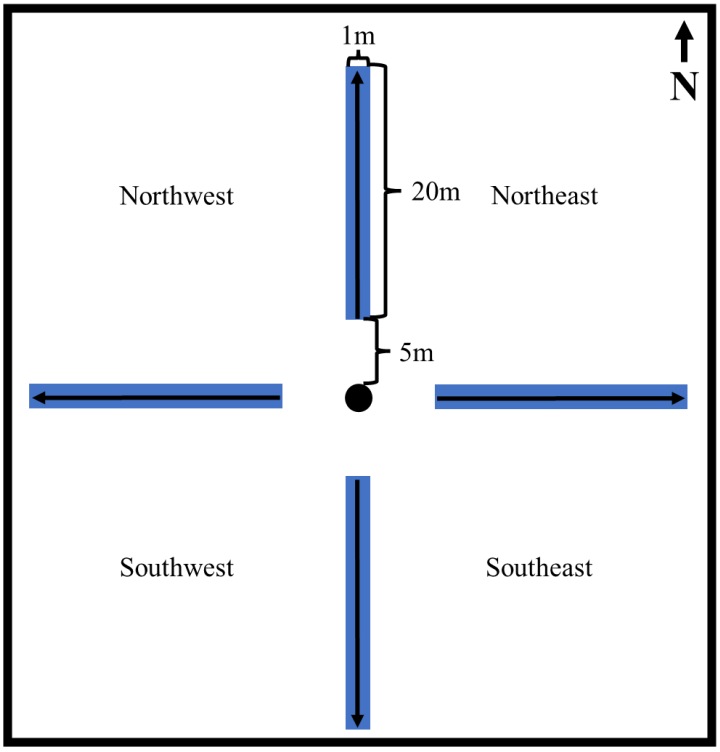

Figure 2: Diagram of site assessment set-up. Five meters from the center pin, a 20 m x 1 m transect (highlighted in blue) was established in each cardinal direction dividing the site into the four sections.

Habitat complexity was determined using a rugosity index, which compares the three-dimensional roughness of the substrate to its planar distance (Oakley-Cogan et al., 2020; Rogers et al., 1982). The rugosity index was created using the chain-and-tape method: a 20 m chain is placed over the contours of the reef directly under each of the four 20 m transect tapes, and the linear distance along each of the 20 m transect tapes where the chain ends is recorded (Oakley-Cogan et al., 2020; Rogers et al., 1982). The rugosity index is determined by dividing the linear distance of the tape (20 m) by the distance the chain reaches (Oakley-Cogan et al., 2020; Rogers et al., 1982). An index value of 1 indicates flat substrate, while values greater than 1 indicate greater complexity. To further assess any large-scale habitat complexity that may otherwise not be captured in the transects, the maximum relief was measured within each section of the site. A diver swam in a lawnmower pattern within each of the 25 m x 25 m sections of the site looking for the highest hard-relief feature, which included ledges, stony corals (alive or dead), and *Xestospongia muta* (giant barrel sponge) (Stein & Ruzicka, 2021). The highest relief feature was measured from the highest point of the structure to the substratum to the nearest centimeter.

Stony coral colony data was collected along four 20 x 1 m belt transects, bringing the total sample area per site to 80 m^2 (Gilliam et al., 2021). Along each transect, all stony coral colonies greater than 2 cm in diameter were identified to species, assigned to a colony diameter size class bin, and assessed for the presence/absence of health conditions (classified as disease, bleaching, predation, and 'other conditions'). The designation 'other conditions' represented additional causes of recent mortality, including sediment burial, abrasion, or overgrowth interactions with other benthic taxa. The colony size class bins were 2-10 cm, 11-20 cm, 21-30 cm, 31-40 cm, 41-50 cm, 51-70 cm, 71-90 cm, and > 91 cm. This data was used to estimate stony coral species density, coral size distribution skewness, and prevalence of health conditions. Stony coral density was determined by taking the number of colonies within each transect divided by the transect area (20 m x 1 m) and expressed as colonies/m^2. Coral size distribution was measured as the skewness of the coral community size frequency distribution curve as corals were only recorded as counts within each size class. Skewness values close to zero represents a more even distribution of coral size, whereas higher values represent a dominance of smaller sized coral (2-20 cm), and lower values represent a large coral dominated community (> 21 cm). Health conditions of wild colonies (bleaching, disease, predation, and 'other conditions') were represented as a percentage by dividing the number of colonies expressing conditions over the total number of colonies at a site.

Study species

Three coral species were collected and used for this study: *Montastraea cavernosa*, *Pseudodiploria clivosa*, and *Siderastrea siderea*. All three species are commonly found in the Coral ECA and have been used in microfragmentation restoration efforts (Figueiredo et al., 2021; Jones et al., 2020; Jones, 2022; Koval et al., 2020; Moyer et al., 2003; Page et al., 2018). Despite being commonly found in the Coral ECA, *M. cavernosa* and *P. clivosa* have undergone significant declines in cover and density from recent bleaching and disease events whereas *S. siderea* has not seen such declines (Darling et al., 2012; Hayes et al., 2022; Jones et al., 2020; Toth et al., 2014; Walton et al., 2018). Coral colonies were collected as corals of opportunity (COOs), which are defined as corals that have been detached from the substrate through natural processes or unknown events. The COOs used in this study were collected throughout 2021 and 2022. Colonies between 20-30 cm in diameter were targeted for collection. Colonies with greater than 25% partial mortality, presence of boring sponge (*Cliona* spp.), or disease were not collected. Collections occurred within Broward County waters in the Coral ECA. A total of 11 colonies were collected (4 *M. cavernosa*, 3 *P. clivosa*, and 4 *S. siderea*). The collected colonies were completely submerged in seawater in a cooler and separated by bubble wrap during transport to the NSU Onshore Nursery facility. At the facility, the collected colonies were immediately placed in a quarantine system, where the colonies resided for 30 days. During the quarantine period, corals were monitored for any signs of disease or recent mortality. After the quarantine period, colonies showing no signs of disease were transferred to the main system, where the corals resided until they were microfragmented and outplanted.

Microfragmenting

A band saw (Gryphon C-40) was used to cut the colonies into approximately 3 cm diameter microfragments to fit onto 3 cm diameter ceramic plugs. One hundred and sixty-eight microfragments were cut for each species totaling 504 microfragments. Newly cut microfragments were dipped for 10 minutes in Brightwell Frag Recover, which aids in healing the cut tissue and prevents infection. After the dip, two-part epoxy was used to attach the microfragments onto ceramic coral plugs, which were separated by species and parent colony and placed onto egg crates in the Onshore Nursery. Colonies were microfragmented April – November 2022, and all resided in the Onshore Nursery for at least two months to recover from the cuts and grow onto the plugs (Table 1). Microfragment monitoring in the Onshore Nursery was conducted weekly to record

health conditions (disease, bleaching) and mortality. Images of microfragments were taken every other week using a set-distanced framer with a measuring tape and Olympus TG-6 camera.

Table 1: Microfragment information based on species, parent colony, and date fragmented.

Species	Parent Colony	Date Fragmented
M. cavernosa	MCAV02	4/13/2022
M. cavernosa	MCAV03	4/13/2022
M. cavernosa	MCAV04	8/9/2022
M. cavernosa	MCAV05	10/31/2022
P. clivosa	PCLI04	9/8/2022
P. clivosa	PCLI05	10/31/2022
P. clivosa	PCLI06	11/7/2022
S. siderea	SSID10	5/11/2022
S. siderea	SSID11	9/8/2022
S. siderea	SSID12	10/31/2022
S. siderea	SSID13	11/7/2022

Microfragment outplanting

After the sites had been assessed, the microfragments were outplanted in January 2023. Microfragments were removed from the Onshore Nursery and placed into mesh bags separated by species and parent colony. The microfragments were completely submerged in seawater inside coolers for transport. Near the center pin at each outplant site, microfragments were individually cemented to the substrate into a 7 x 6 microfragment grid with approximately 50 cm between the microfragments. Approximately two handfuls of cement were used to attach microfragments to the substrate, and the cement was of a mixture of Portland type I/II cement with 10% silica fume by weight (Unsworth et al., 2021) and will be referenced as cement bases for this study. The cement was prepared on the boat by one team member using seawater and placed into large plastic bags to be easily carried by divers. The underlying substrate where the cement bases were placed was cleaned using brushes and hammers to ensure a good seal for the cement. At each site, each species

had 14 individual microfragments randomly oriented within each grid. Each parent colony was equally represented at all sites.

Microfragment outplant monitoring

Outplant monitoring periods occurred 1 day, 1 week, 2 weeks, 1 month, 2-months, 3-months, and 6-months post-outplanting for a total of seven monitoring events. During monitoring events, images were taken of each individual cement base using a set-distanced framer with a measuring tape and an Olympus TG-6 camera. Each microfragment was recorded as dead or alive and the presence of predation, disease, and bleaching was recorded. The bases were not cleaned upon revisiting the sites so that overgrowth by competing benthic taxa could be assessed. At the 6-month interval, a second set of images were taken after cleaning the bases in order to capture the true microfragment live tissue area. Images taken at each monitoring event were also used to quantify overgrowth.

Microfragment-base community

Microfragment-base images were analyzed using CPCe image analysis software (Kohler et al., 2006). The software generated ten random points on a set border of 15 x 15 cm, centered on the microfragment, covering the cement base and the immediate surrounding natural substrate. Underneath each point, the benthic functional group was identified as macroalgae, crustose coralline algae, sponge, stony coral, bare substrate, and epilithic algal matrix (EAM) (Wilson & Bellwood, 1997). These points were used to determine the percent cover of each functional group at the initial, 3-month, and 6-month interval, indicating any long-term competition/overgrowth that may have contributed to microfragment mortality.

Microfragment growth

Microfragment growth was determined using the monitoring images. Planar live tissue area of each microfragment was measured using ImageJ software (Abramoff et al., 2004). Images were calibrated using the scale bar attached to the framer, and the outline of the microfragment was traced using a tracing pad to calculate the total area of living tissue at each timepoint (cm^2). The initial and second set of 6-month monitoring images were the only timepoints used for analyzing growth. Net growth was calculated as the difference between live tissue area during the 6-month monitoring interval and the initial monitoring interval. Relative growth, arithmetic mean radius,

and linear extension were also measured for each individual microfragment in order to compare with other growth studies (Equation 1) (Pratchett et al., 2015).

Equation 1: Relative growth, arithmetic mean radius (AMR), and linear extension equations. (1.1) Relative growth was calculated by the net growth divided by the initial area (t0) times 100. (1.2) The AMR (in cm) was calculated from the square root of the live tissue area at a given timepoint (tn) divided by pi. (1.3) Linear extension (cm per month) was calculated by subtracting AMR0 (AMR of initial timepoint) from AMR6 (AMR of 6-month timepoint) and dividing by n (the time between monitoring intervals).

$$(1.1) \quad \text{Relative growth } (\%\ 6\ \underline{\text{mo}}^{-1}) = \frac{((t6)-t0)}{t0} * 100$$

$$(1.2) \quad \text{AMR (cm)} = \frac{\sqrt{tn}}{\pi}$$

$$(1.3) \quad \text{Linear Extension (cm } \underline{\text{mo}}^{-1}) = \frac{((AMR6)-AMR0)}{n}$$

Statistical analyses

All statistical analyses were implemented in R Studio (Version 4.1.1) (R Studio Team 2020). The R package 'moments' was used to quantify the skewness of the coral community size frequency distribution curve (Komsta, 2022). An analysis of covariance was performed on all of the measured characteristics (rugosity index, maximum relief, coral density, coral size skewness, coral species richness, coral bleaching prevalence, coral predation prevalence, and coral 'other conditions' prevalence) to determine if any of them covary. Analysis found that the rugosity index and coral density covaried by 90%; therefore, the rugosity index was removed as a factor for all further analyses. Additionally, only one instance of disease was recorded (at site 8); therefore, disease was included within 'other conditions' for wild coral conditions.

Generalized linear mixed models (GLMMs) from the package 'glmmTMB' were used to test if microfragment survival and relative growth overall and by species could be predicted by the modified site characteristics measured during the rapid site assessment as fixed effects (maximum relief, species richness, coral density, coral size, wild colony bleaching, disease, and 'other conditions') (Brooks et al., 2017). A model of best fit approach was used, beginning with the full

model of the modified site characteristics as factors and site and species as a random effect (Equation 2). Models for each microfragment species only had site as the random effect. Microfragments that did not survive were excluded from the GLMMs investigating microfragment growth. Testing of multicollinearity on the factors under the package 'performance' was done on the complete model and the model of best fit, which was selected based on the lowest Akaike Information Criterion (AIC). Validation of the model was performed using the package 'DHARMa' (Hartig 2017). To observe the effect the factors had on survival and growth, the probability estimate was calculated as the inverse of the logit probability from the estimated coefficients in the fitted model 1/1+1/exp(x) where x is the estimated coefficient. For example, a probability estimate of 0.25 coral density means that for every unit increasing the microfragment survival likelihood increases by 25%. A non-parametric Kruskal-Wallis test was used for net growth to examine significant differences amongst sites, species, and reef.

Equation 2: Full model equations of GLMMs examining variations in microfragment survival (A) and growth rates (B) off the site characteristics.

A) Survival ~ maximum relief + species richness + coral density + coral size + natural colony bleaching + natural colony predation + natural colony other conditions + (1|site) +(1|species)

   Family = Binomial

B) Relative growth ~ maximum relief + species richness + coral density + coral size + natural colony bleaching + natural colony predation + natural colony other conditions + (1|site) +(1|species)

   Family = Gaussian

To answer the supportive question of this study of what specifically is driving microfragment mortality, the cover of benthic taxa on the cement base, and the number of health condition instances for each individual microfragment was tested against microfragment survival and relative growth (overall and by species) using GLMMs. The conditions (bleaching, predation,

and ‘other conditions’) were represented by the number of instances each microfragment experienced across all monitoring intervals. For the benthic community, only the taxa that likely influenced the health of the microfragments (EAM, sponge, cyanobacteria, and macroalgae) were included in the model to avoid making it too complex. Only the 3-month and 6-month intervals were used for the models because no benthic taxa had settled on the base in the initial monitoring. Like the GLMMs observing the effects of the site conditions, a single model approach was used, with the full model of the modified site characteristics as factors and site and species as random effects (Equation 3). Again, the models testing individual species only had site as a random effect. Testing of multicollinearity on the factors was performed on the complete model and the model of best fit, and validation of the model was performed using the package ‘DHARMa’ (Hartig, 2017).

Equation 3: Full model equations of GLMMs examining variations in microfragment survival (A) and growth rates (B) off the cover of benthic community taxa and the instances of health conditions.

A) Survival ~ microfragment predation + microfragment bleaching+ microfragment other recent mortality + epilithic algal matrix + sponge + cyanobacteria +macroalgae + (1|site) +(1|species)

   Family = Binomial

B) Relative growth ~ microfragment predation + microfragment bleaching+ microfragment other recent mortality + epilithic algal matrix + sponge + cyanobacteria +macroalgae + (1|site) +(1|species)

   Family = Gaussian

Results:

Rapid assessment

The rapid assessment at each site, from the beginning to the end of the dive, was completed in an average of 37 minutes and 40 seconds, with the longest time spanning only 45 minutes. The rapid assessment determined rugosity index for all sites indicated relatively low complexity (all

sites close to 1), with site 5 having the overall highest rugosity index of 1.4, and also having the highest maximum relief (Table 2). Site 5 additionally had the highest species richness and coral density. Stony coral size, expressed as the skewness of the corals in each size class, found that site 3 had the most even distribution, meaning that it had a greater number of large corals compared to the sites with primarily smaller corals. In comparison, site 2 had the highest skewness, with a coral community primarily consisting of smaller corals. For all sites, corals within 2-10 cm in diameter were the most numerous (Table 2). For the health conditions of wild colonies, presence/absence of disease, bleaching, predation, and 'other conditions' were recorded (Table 3). Site 8 was the only site that had an instance of disease within the transects (1 *Dichocoenia stokesi* colony). Bleaching, defined as any color loss or paling observed on the coral colony, was present at 10 sites, but no sites had > 5% bleaching prevalence, with site 4 having the most bleached corals (4.4%). Predation was present at all sites, with site 11 having the highest prevalence (13%). The species with the most observed predation scars were *P. porites* and *P. astreoides* (accounting for 80.2% of all predation instances). *Siderastrea siderea* experienced moderate predation and all other species had low to zero predation instances.

Table 2: Rapid assessment measurements for each site. The rugosity index and coral density are shown as the mean ± SE. Gray shaded sites were those located on the inner reef and unshaded sites were located on the nearshore ridge complex.

Site	Rugosity Index	Max Relief (m)	Species Richness	Coral Density (colonies/ m^2)	Corals Between 2-10 cm (%)
1	1.05 ± 0.01	0.75	8	1.58 ± 0.11	73.8
2	1.11 ± 0.03	1	9	2.29 ± 0.75	95.1
3	1.09 ± 0.01	0.8	5	1.44 ± 0.29	62.6
4	1.24 ± 0.04	0.79	11	3.11 ± 0.47	82.7
5	1.40 ± 0.07	1	13	4.44 ± 0.35	82.5
6	1.12 ± 0.01	1.27	9	0.99 ± 0.21	74.7
7	1.22 ± 0.01	0.75	9	3.10 ± 0.27	85.5
8	1.18 ± 0.02	0.76	12	2.05 ± 0.32	81.1
9	1.17 ± 0.01	0.6	8	1.84 ± 0.4	80.3
10	1.15 ± 0.01	0.6	9	1.88 ± 0.35	65.3
11	1.17 ± 0.02	0.75	10	1.85 ± 0.21	78.4

12	1.19 ± 0.04	0.75	10	2.24 ± 0.29	73.2

Table 3: Number of wild coral colonies and percent of conditions observed at each site. Gray shaded sites were those located on the inner reef and unshaded sites were located on the nearshore ridge complex.

Site	# of colonies	Bleaching	Disease	Predation	Other Conditions
1	126	3.97%	0%	3.97%	0%
2	183	1.09%	0%	4.92%	0.55%
3	115	2.61%	0%	5.22%	1.74%
4	249	4.42%	0%	6.02%	0.80%
5	355	3.10%	0%	4.23%	1.69%
6	79	3.78%	0%	5.06%	1.27%
7	248	0.81%	0%	4.44%	2.42%
8	164	0.61%	0.61%	6.71%	2.44%
9	147	0%	0%	5.44%	2.04%
10	150	0.67%	0%	7.33%	2.67%
11	148	0.68%	0%	12.84%	1.35%
12	179	0%	0%	3.35%	1.12%

Microfragment survival

After 6-months, microfragment survival was 94.4%, with only 28 out of the 504 total outplanted microfragments having complete mortality. Six microfragments were dislodged and missing from the cement base and were included in the count of microfragments with complete mortality. The sites with the highest survival were 1, 2, and 6, at 100%, and the site with the lowest survival was 10, at 85.7% (Figure 3). *Montastraea cavernosa* experienced the highest survival of 96.4%, followed by *S. siderea* at 94.4%, and *P. clivosa* at 92.3%.

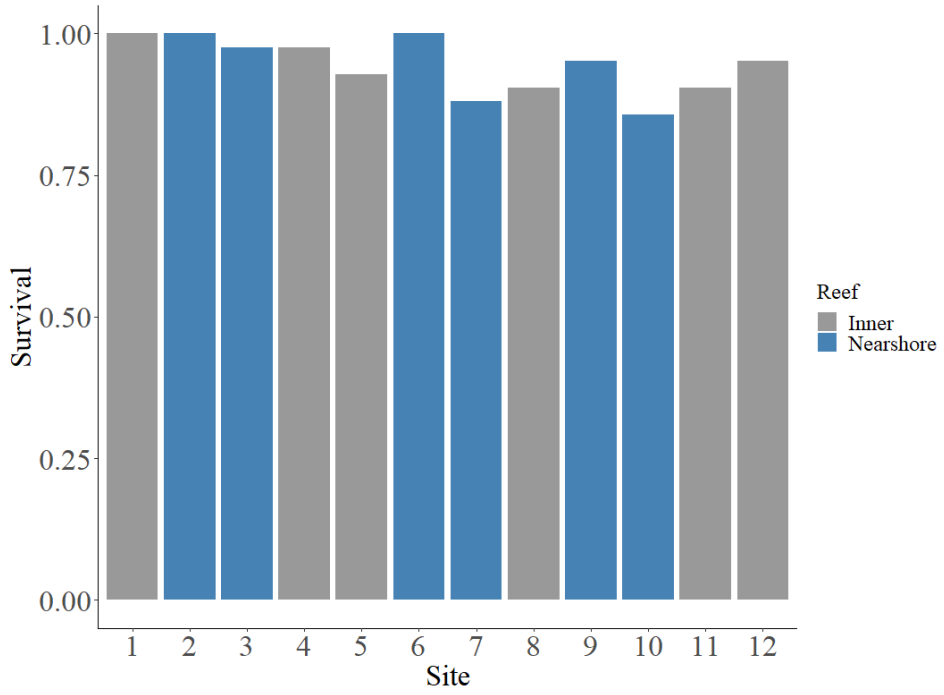

Figure 3: Bar graph of microfragment survival rates (%) after 6-months by site. Gray shaded sites were located on the inner reef and blue shaded sites were located on the nearshore ridge complex.

For all species, outplant site wild coral density and the prevalence of conditions the wild colonies had appeared to be indicators for site level processes that could in turn affect microfragment survival (GLMM, Marginal R^2 = 0.38) (Figure 4). Coral density (probability estimate = -0.3, p = 0.04), wild coral predation (probability estimate = -0.5, p = 0.01), and 'other conditions' (probability estimate = -0.3, p = 0.01) were all found to be significant indicators of microfragment survival, with 'other conditions' having the strongest relationship. Wild coral bleaching was found to be an indicator that benefited microfragment survival, but it was not significant (probability estimate = 0.6, p = 0.07).

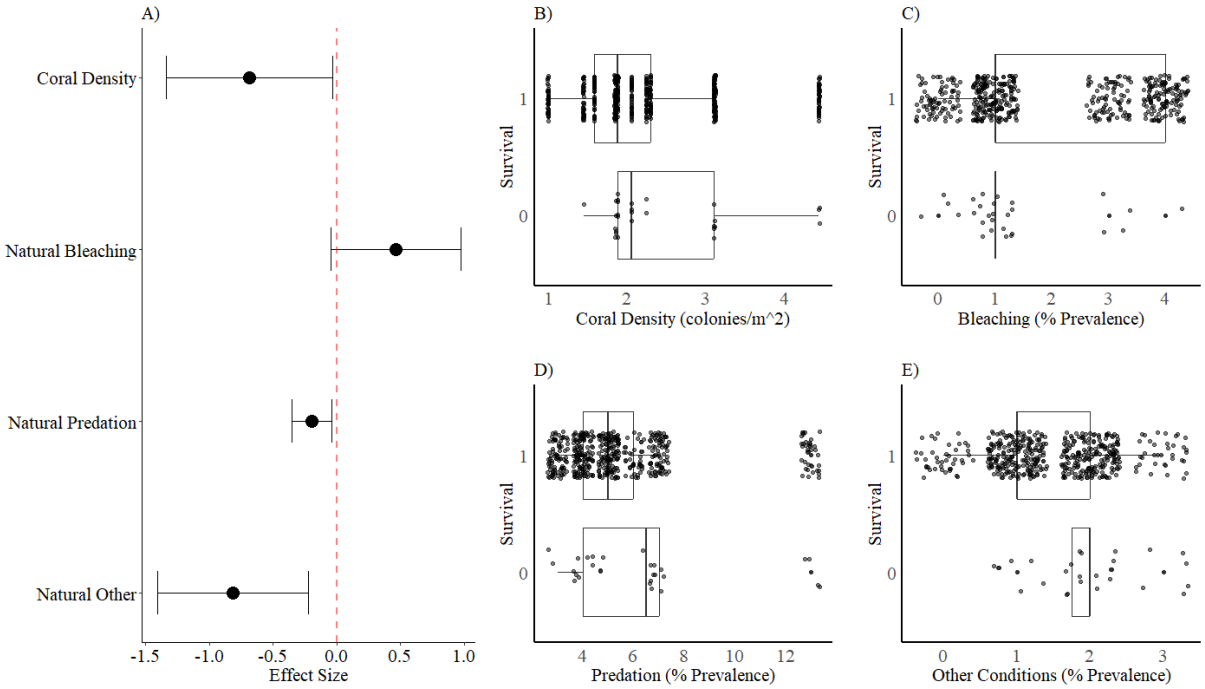

Figure 4: Site conditions that indicated microfragment survival. A) Forest plot explaining the effect size of conditions on microfragment survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-E) Box plot with jittered points of microfragment survival (1 = alive, 0 = dead) and its relationship to site conditions. Points represent the survival of each individual microfragment. Bleaching, predation, and other conditions refer to the prevalence on wild corals.

On a species level, *M. cavernosa* survival appeared to be indicated only by the maximum relief (GLMM, Marginal $R^2 = 0.27$) (Figure 5). The maximum relief (probability estimate = 1.0, $p = 0.123$), although found to indicate higher survival, was not found to be significant due to the overall high survival rates of this species. The best fit model found site characteristics that appear to influence *P. clivosa* survival (GLMM, Marginal $R^2 = 0.99$) (Figure 6). The site characteristics that appear to indicate *P. clivosa* survival are maximum relief (probability estimate = 1.0, $p = 0.13$), species richness (probability estimate = 0.002, $p = 0.14$), coral density/ rugosity index (probability estimate = 1.5×10^{-11}, $p = 0.13$), wild coral bleaching (probability estimate = 1.0, $p = 0.15$), and wild coral predation (probability estimate = 0.07, $p = 0.15$), but again these were all not found to be significant. For *S. siderea*, the model found that survival was indicated by the

prevalence of predation (probability estimate = 0.44, $p = 0.03$) and 'other conditions' (probability estimate = 0.30, $p = 0.08$) on wild corals (GLMM, Marginal $R^2 = 0.21$) (Figure 7). This was the only model that witnessed a significant relationship with predation prevalence on wild corals indicating lower *S. siderea* survival.

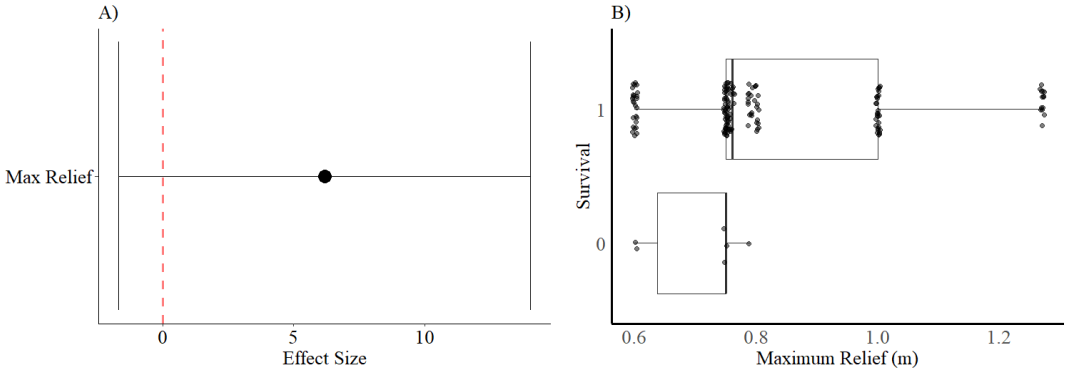

Figure 5: Site conditions that indicated *M. cavernosa* survival. A) Forest plot explaining the effect size of conditions on survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B) Box plot with jittered points of survival (1 = alive, 0 = dead) and its relationship to the site condition. Points represent the survival of each individual microfragment.

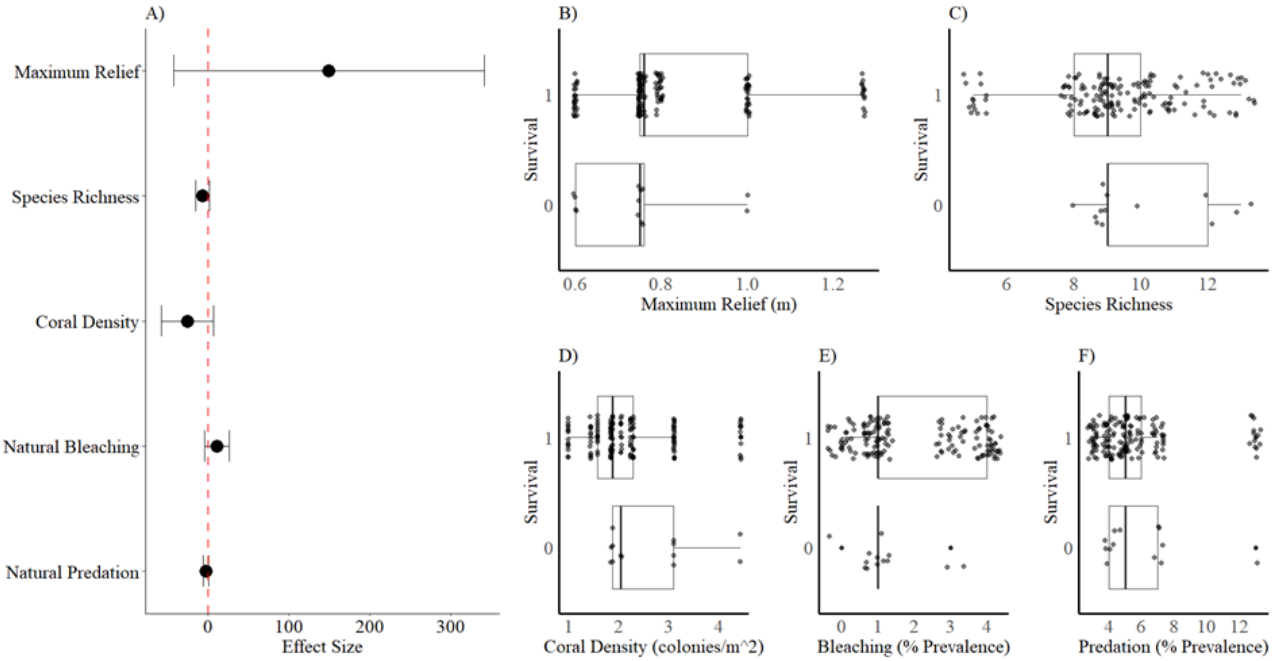

Figure 6: Site conditions that indicated *P. clivosa* survival. A) Forest plot explaining the effect size of conditions on survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-F) Box plot with jittered points of microfragment survival (1 = alive, 0 = dead) and its relationship to site conditions. Points represent the survival of each individual microfragment. Bleaching and predation refer to the prevalence on wild corals.

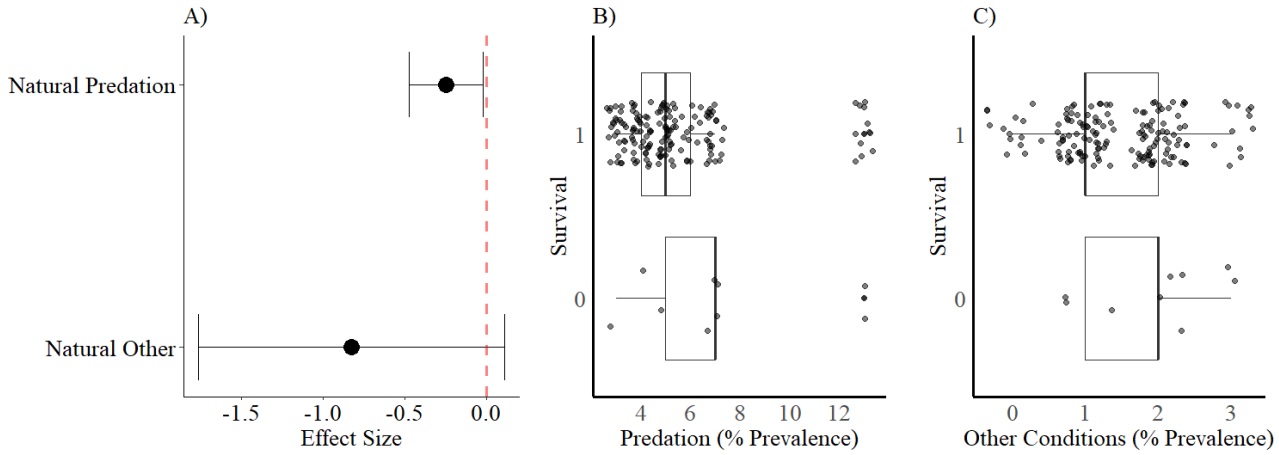

Figure 7: Site conditions that indicated *S. siderea* survival. A) Forest plot explaining the effect size of conditions on survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-C) Box plot with jittered points of microfragment survival (1 = alive, 0 = dead) and its relationship to site conditions. Points represent the survival of each individual microfragment. Predation and other conditions refer to the prevalence on wild corals.

Microfragment growth

Microfragment growth across all species was found to vary among sites (Figure 8). Average net growth in sites 1, 2, 3, 4, 6, and 9 were positive, whereas sites 5, 7, 8, 10, 11, and 12 were negative. The two sites with the highest net growth were sites 2 and 4, whereas sites 8 and 11 had the lowest. Site 4 was found to be significantly greater than sites 7, 8, 11, and 12, while site 2 was only significantly greater than sites 7, 8, and 11. Site 8 was additionally significantly lower than site 1 and 9. Of the surviving microfragments, 31% decreased in live tissue area, and 20% increased by more than 1 cm^2.

Microfragment growth was indicated by site maximum relief (probability estimate = -4.7×10^{-17}, $p = 0.03$), species richness (probability estimate = -0.06, $p = 0.02$), wild coral size (probability estimate = -2.0×10^{-20}, $p = 0.04$), wild coral bleaching prevalence (probability estimate = 0.99, $p = 0.01$), and predation prevalence (probability estimate = -0.14, $p = 0.03$) (GLMM, Conditional $R^2 = 0.17$, Marginal $R^2 = 0.07$) (Figure 9). Species richness was the strongest indicator that hindered growth. Wild coral bleaching was the only indicator for growth.

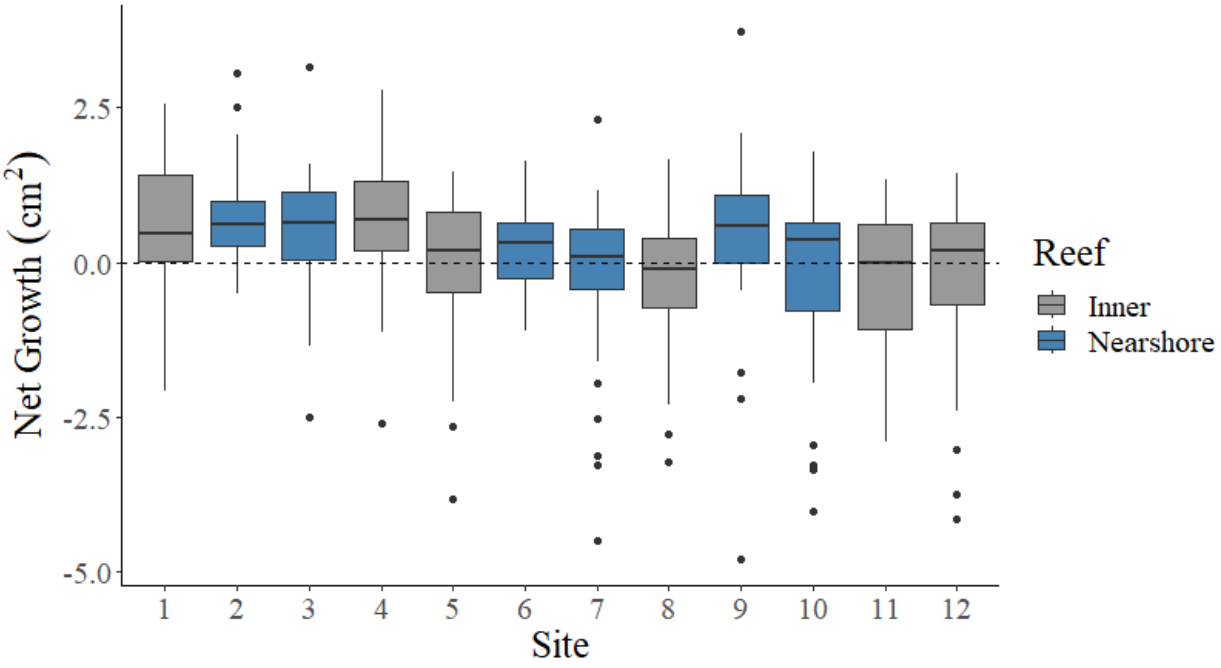

Figure 8: Net growth (cm^2) of the outplanted microfragments by site. A mean above the dashed line represents positive net growth, and below represents negative net growth.

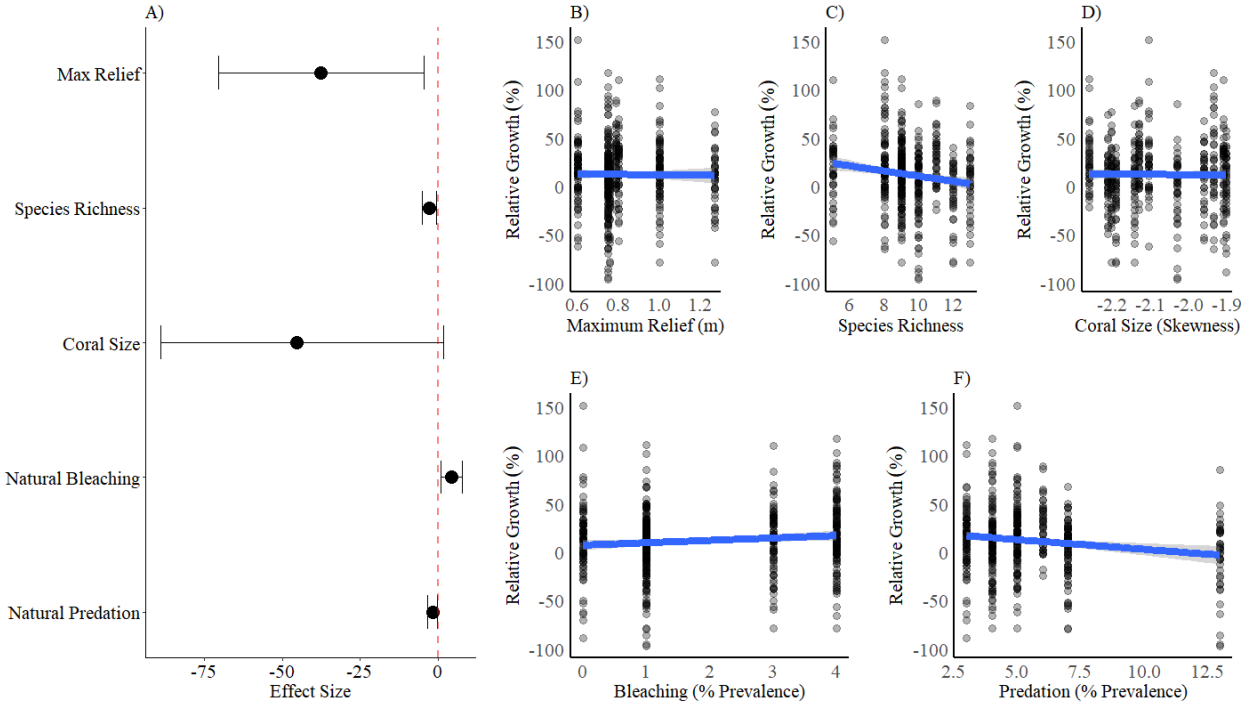

Figure 9: Site conditions that indicated microfragment growth. A) Forest plot explaining the effect the site conditions have on microfragment growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-F) Microfragment relative growth and its relationship to site conditions, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate. Bleaching and predation refer to the prevalence on wild corals.

Growth rates varied among species (Figure 10). *Montastraea cavernosa* and *P. clivosa* were not significantly different and displayed overall positive relative growth through the duration of the study. On average, *P. clivosa* microfragments experienced nearly a 16% increase in growth, followed by *M. cavernosa*, with nearly 5% growth. *Siderastrea siderea*, however, was significantly different and was the only species observed to have an overall negative relative growth, declining by nearly 3%. Of the surviving microfragments, 16% of *P. clivosa*, 29% of *M. cavernosa*, and 43% of *S. siderea* decreased in live tissue area. Linear extension rates were low overall and were better represented as the change in millimeters rather than centimeters (Table 4). *Pseudodiploria clivosa* was the only species that displayed positive linear extension. Reef type was not found to be significantly different in growth.

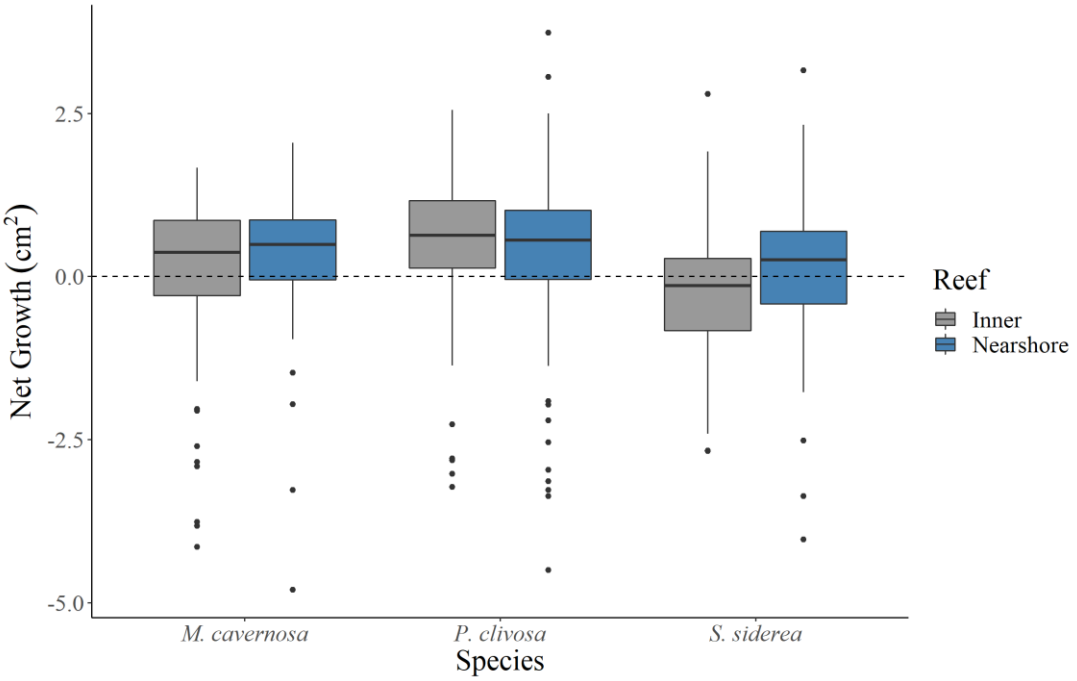

Figure 10: Net growth (cm^2) of the outplanted microfragments by species. A mean above the dashed line represents positive net growth, and below represents negative net growth.

Table 4: Calculated growth rates for each coral species (mean ± SD). AMR = arithmetic mean radius of microfragments when initially outplanted. Relative growth rate = mean percentage change in planar area for the 6-month duration of the study (% 6 mo^{-1}), then converted to capture yearly percent change (% yr^{-1}). Linear extension = difference in AMR from the first and last monitoring intervals, then converted to capture yearly extension in mm.

Species	**AMR (cm)**	**Relative growth rate (% 6 mo^{-1})**	**Relative growth rate (% yr^{-1})**	**Linear Extension (mm yr^{-1})**
Montastraea cavernosa	0.61 ± 0.01	4.96 ± 2.47	9.91 ± 4.93	-0.02 ± 0.21
Pseudodiploria clivosa	0.54 ± 0.01	15.68 ± 3.5	31.36 ± 7.00	0.22 ± 0.27
Siderastrea siderea	0.48 ± 0.01	-2.77 ± 3.45	-5.55 ± 6.90	-0.61 ± 0.22

Montastraea cavernosa growth appeared to be indicated by species richness, coral size and wild coral bleaching (GLMM, Marginal $R^2 = 0.08$) (Figure 11). Coral size (probability estimate = 6.12×10^{-12}, $p = 0.12$), although found to indicate reduced growth, was not found to

be significant, but species richness (probability estimate = 0.05, $p = 0.01$) was significant. Additionally, the prevalence of bleaching on the wild coral colonies (probability estimate = 0.95, $p = 0.02$) indicated more growth in this species. The best fit model found that *P. clivosa* growth was primarily indicated by the prevalence of predation and 'other conditions' on wild coral colonies (GLMM, Conditional $R^2 = 0.16$, Marginal $R^2 = 0.08$) (Figure 12). Wild coral predation (probability estimate = 0.12, $p = 0.11$) was not a significant influence, but 'other conditions' (probability estimate = 0.08, $p = 2.38 \times 10^{-4}$), coral density/ rugosity index (probability estimate = 1.5×10^{-11}, $p = 0.131$) was significant. For *S. siderea*, the model found that survival was indicated by the maximum relief (probability estimate = 3.42×10^{-9}, $p = 0.37$), species richness (probability estimate = 3.3×10^{-4}, $p = 9.01 \times 10^{-4}$), coral density (probability estimate = 1.0, $p = 0.08$), and coral size (probability estimate = 9.73×10^{-35}, $p = 0.02$) on wild corals (GLMM, Conditional $R^2 = 0.12$, Marginal $R^2 = 0.1$) (Figure 13). For this model, only the relationships between species richness and coral size were found to be significant.

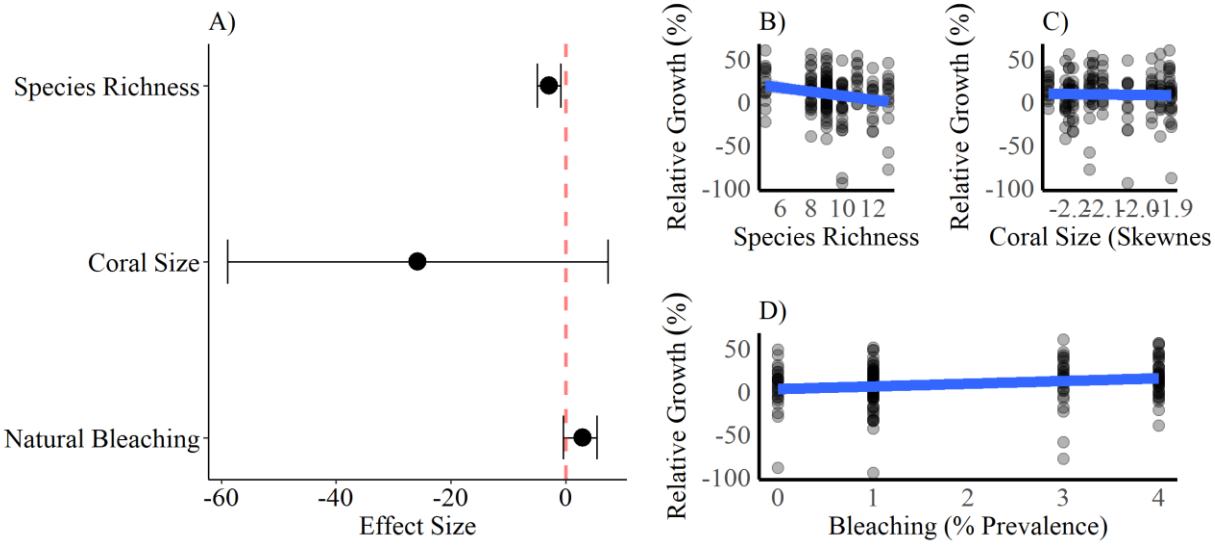

Figure 11: Site conditions that indicated *M. cavernosa* relative growth. A) Forest plot explaining the effect size of conditions on relative growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-D) Microfragment relative growth and its relationship to site conditions, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate. Bleaching refers to the prevalence on wild corals.

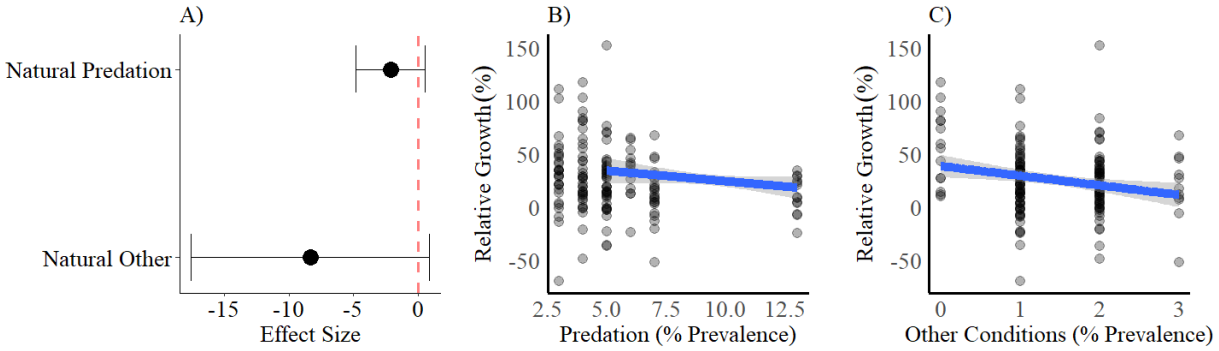

Figure 12: Site conditions that indicated *P. clivosa* relative growth. A) Forest plot explaining the effect size of conditions on relative growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-C) Microfragment relative growth and its relationship to site conditions, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate. Predation and 'other conditions' refers to the prevalence on wild corals.

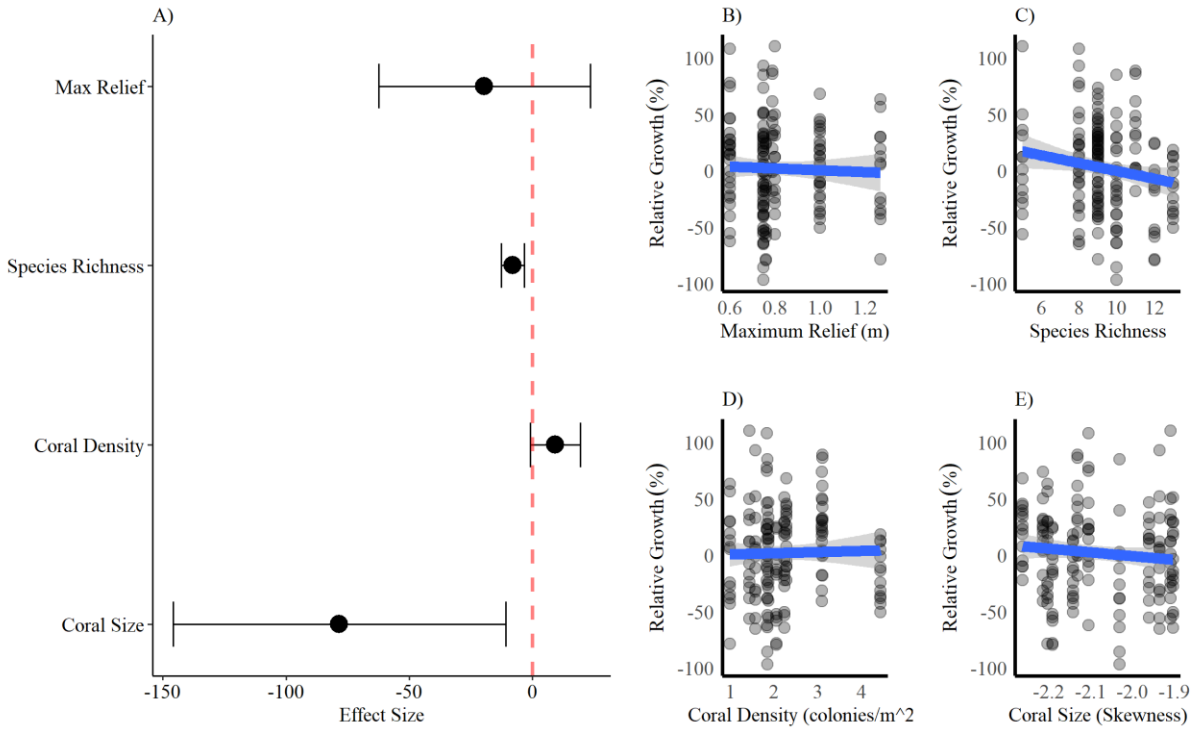

Figure 13: Site conditions that indicated *S. siderea* relative growth. A) Forest plot explaining the effect size of conditions on relative growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-E) Microfragment relative growth and its relationship to site conditions, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate.

Microfragment conditions and benthic community

Despite such high survival documented after 6-months, conditions that can affect microfragment long term health were observed, which can help explain relative growth. The most common condition affecting health observed on the outplanted microfragments at every site was predation which affected 49.8% of all microfragments (Figure 14). Most notable was the predation levels witnessed after one week post-outplanting, where ≥ 50% of the microfragments in sites 5, 8, 10, 11, and 12 experienced predation. The other sites did not experience such high predation throughout all monitoring periods and never exceeded 30%. Most sites began to show declines in predation intensity 1-month post-outplanting. After 6-months post-outplanting, predation across all sites dropped to nearly zero instances.

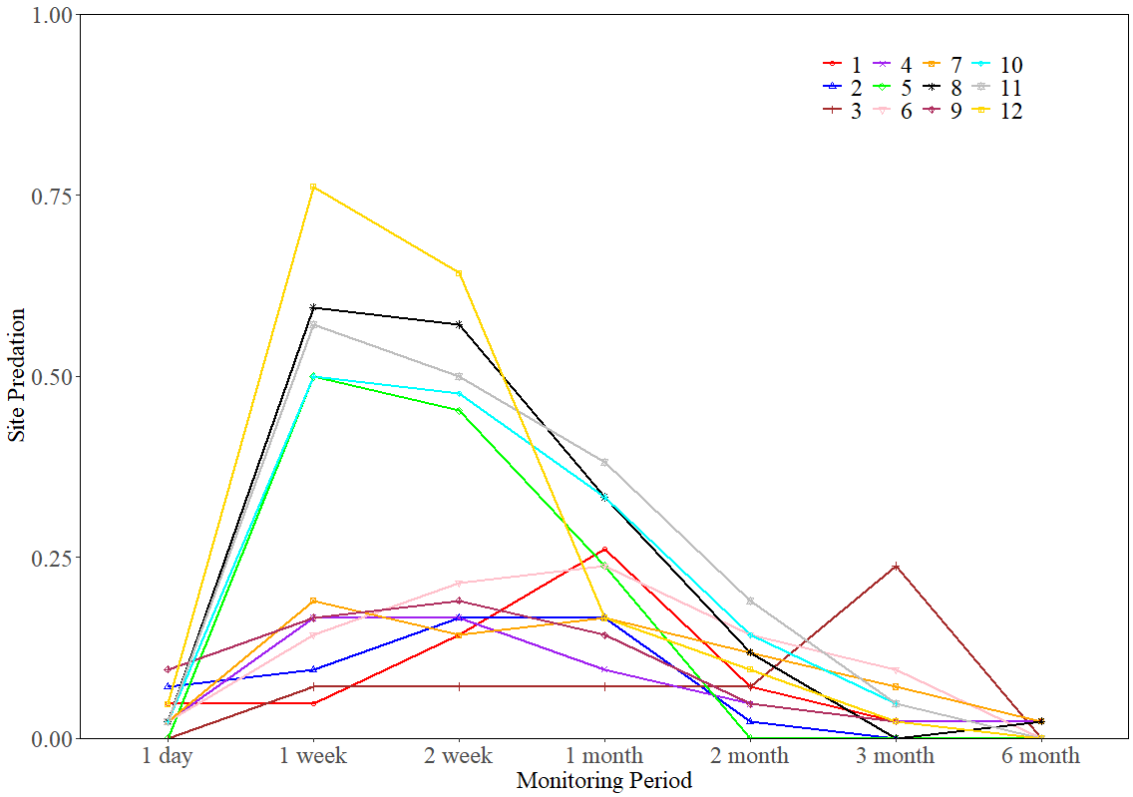

Figure 14: Total predation prevalence on microfragments by site at each monitoring interval. Each colored line represents an individual site.

On the species level, instances of predation were highest for *S. siderea*, followed by *P. clivosa*, and then *M. cavernosa* for all monitoring periods. Predation was highest for *S. siderea* (58%) after two weeks, whereas the highest predation for *P. clivosa* (26%) and *M. cavernosa* (17%) occurred after one week. Predation on *S. siderea* reached ≥ 50% across 8 sites on at least one monitoring interval (sites 1, 3, 5, 6, 8, 10, 11 and 12) and was the only species to show 100% predation prevalence at two sites (5 and 12) after one week post outplanting. Only one site (site 7) had below 25% predation across all monitoring intervals.

After predation, bleaching was the second most common condition observed. Ten *M. cavernosa* microfragments spanning across 7 of the 12 sites bleached after one-week post-outplanting; however, this occurrence was isolated to one parent colony and was considered stress from the outplanting. Bleached *Pseudodiploria clivosa* microfragments were never observed.

Bleached *S. siderea* microfragments (22% of all *S. siderea* microfragments) were observed across 11 sites during the 6-month monitoring event bleaching took place during the summer. 'Other recent mortality' conditions were uncommon, only affecting < 5% of all microfragments.

The benthic community on the cement base followed a pattern of initially consisting of bare substrate post-outplanting, and over time gradually being covered by EAM and macroalgae (Figure 15). When assessing the effect of benthic community and presence of microfragment conditions on microfragment survival and growth, predation (estimate = -0.2 ± 0.2, $p = 0.26$) and presence of macroalgae (estimate = -0.7 ± 0.4, $p = 0.08$) around the microfragment negatively affected survival, but was not significant (GLMM, Conditional $R^2 = 0.1$; Marginal $R^2 = 0.06$) (Figure 16). Predation (estimate = -9.3 ± 1.4, $p = 6.0 \times 10^{-11}$), 'other recent mortality' (estimate = -18.8 ± 5.1, $p = 2.3 \times 10^{-4}$), and presence of EAM (estimate = -14.9 ± 5.2, $p = 4.1 \times 10^{-3}$), sponge (estimate = -88.7 ± 40.4, $p = 0.03$), and macroalgae (estimate = -16.1 ± 6.9, $p = 0.02$) around the microfragment significantly affect growth (GLMM, Conditional $R^2 = 0.23$; Marginal $R^2 = 0.16$), with predation having the strongest effect (Figure 17).

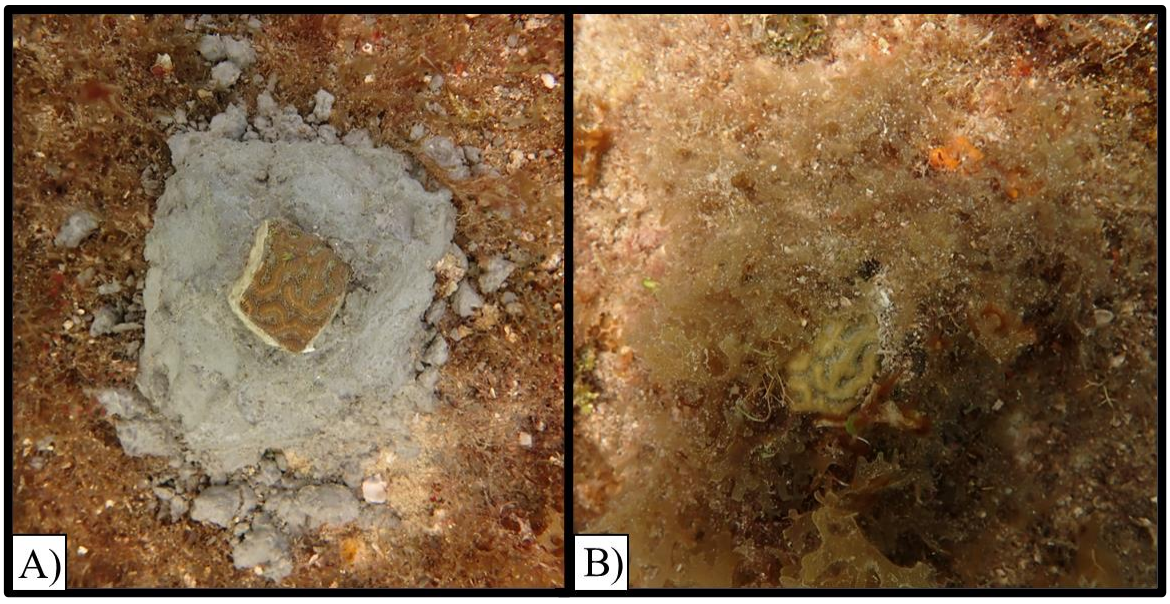

Figure 15: Change in base community after 6-months. A) Image of an individual *P. clivosa* microfragment day 1 post-outplanting. B) Image of the same individual microfragment 6-months post-outplanting with macroalgae covering the cement base.

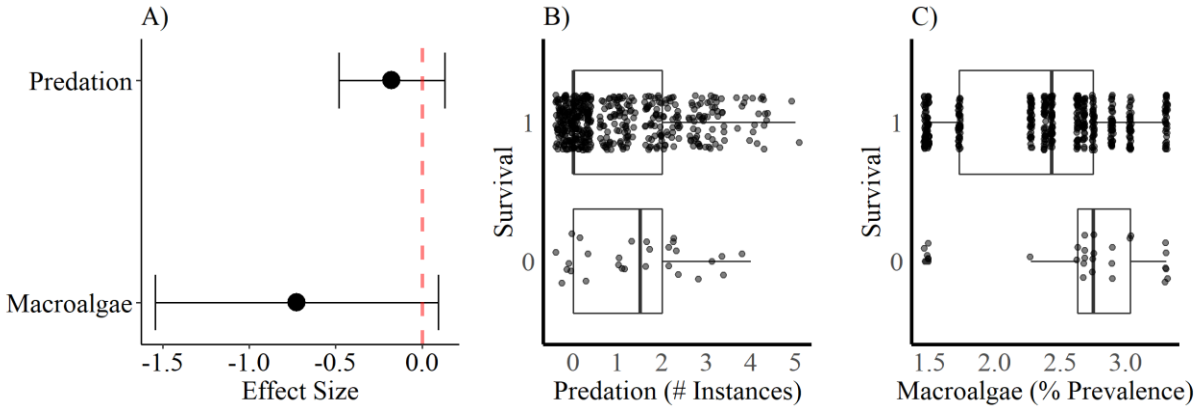

Figure 16: Conditions and benthic taxa that effected microfragment survival. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on microfragment survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-C) Box plot with jittered points explaining survival (1 = alive, 0 = dead) vs benthic community and condition, with points representing the survival of each individual microfragment.

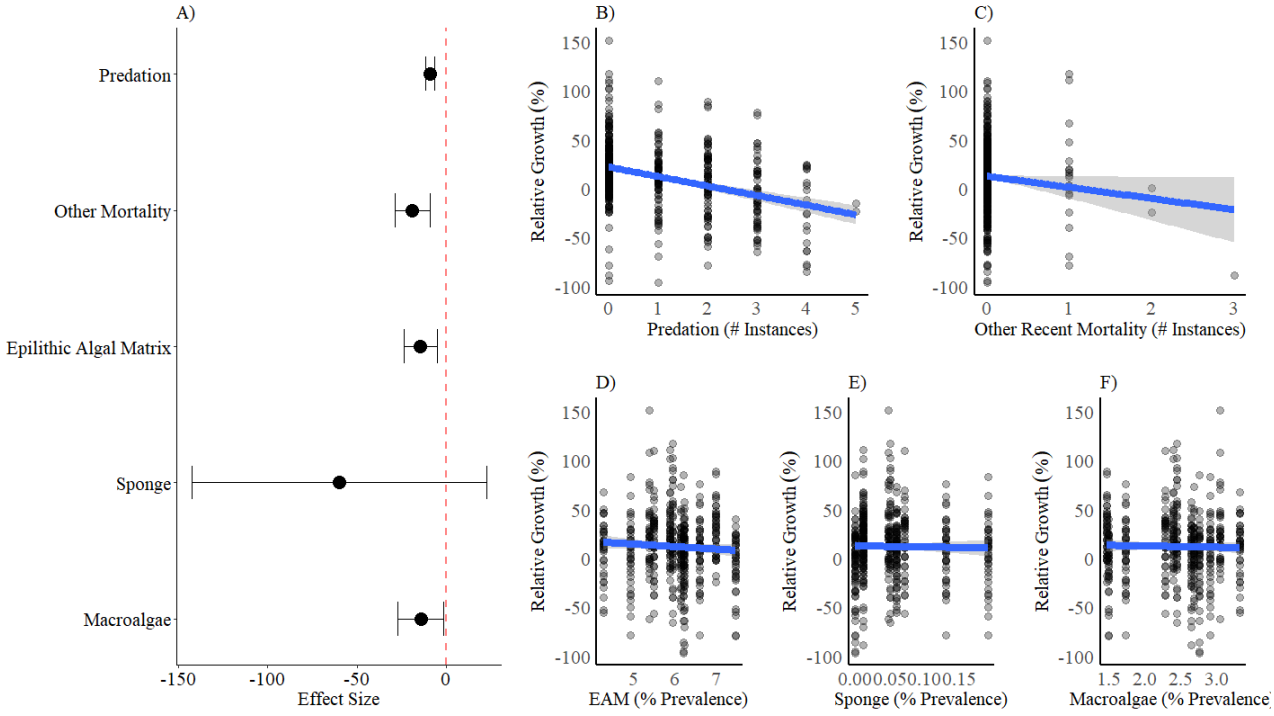

Figure 17: Conditions and benthic taxa that significantly effected microfragment growth. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on microfragment growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-F) Relative growth vs benthic community or condition, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate.

Models observing the effect of benthic community and presence of microfragment conditions found differences among microfragment species. *Montastraea cavernosa* survival was only affected by the cover of sponges (estimate = 10.8 ± 11.1, $p = 0.3$) on the cement base but was not significant from the high survival rates of the species and the overall low abundance of sponges on the cement base (GLMM, Marginal $R^2 = 0.097$) (Figure 18). Growth however was significantly affected by the number of predation (estimate = -7.9 ± 2.3, $p = 4.8 \times 10^{-4}$) and 'other conditions' (estimate = -28.0 ± 5.6, $p = 5.6 \times 10^{-7}$) as well as the cover of EAM (estimate = -15.4 ± 5.1, $p = 2.5 \times 10^{-3}$), sponge (estimate = -69.0 ± 38.1, $p = 7.0 \times 10^{-2}$), and macroalgae (estimate = -23.4 ± 6.7, $p = 5.1 \times 10^{-4}$) (GLMM, Marginal $R^2 = 0.25$) (Figure 19). *Pseudodiploria clivosa* survival was only affected by the cover of macroalgae (estimate = -1.1 ± 0.7, $p = 0.1$) but was not found to be

significant (GLMM, Conditional R^2 = 0.23, Marginal R^2 = 0.09) (Figure 20). Growth was significantly affected by the number of predation (estimate = -8.3 ± 2.8, p = 3.1 x 10^{-3}) and 'other conditions' (estimate = -14.4 ± 6.8, p = 3.3 x 10^{-2}), and was affected by the cover of cyanobacteria (estimate = 32.0 ± 20.1, p = 0.1) but was not significant (GLMM, Conditional R^2 = 0.2, Marginal R^2 = 0.1) (Figure 21). *Siderastrea siderea* survival was only affected by bleaching (estimate = 2.3 x 10^1 ± 7.6 x 10^4, p = 1.0) but this relationship is not significant and appears to benefit the survival of microfragments (GLMM, Marginal R^2 = 1.0) (Figure 22). Growth was significantly affected by predation (estimate = -11.0 ± 2.1, p = 1.9 x 10^{-7}) and bleaching (estimate = 19.3 ± 7.0, p = 5.7 x 10^{-3}), and was affected by the cover of sponges (estimate = -79.4 ± 53.8, p = 0.1) but was not significant (GLMM, Marginal R^2 = 0.19) (Figure 23).

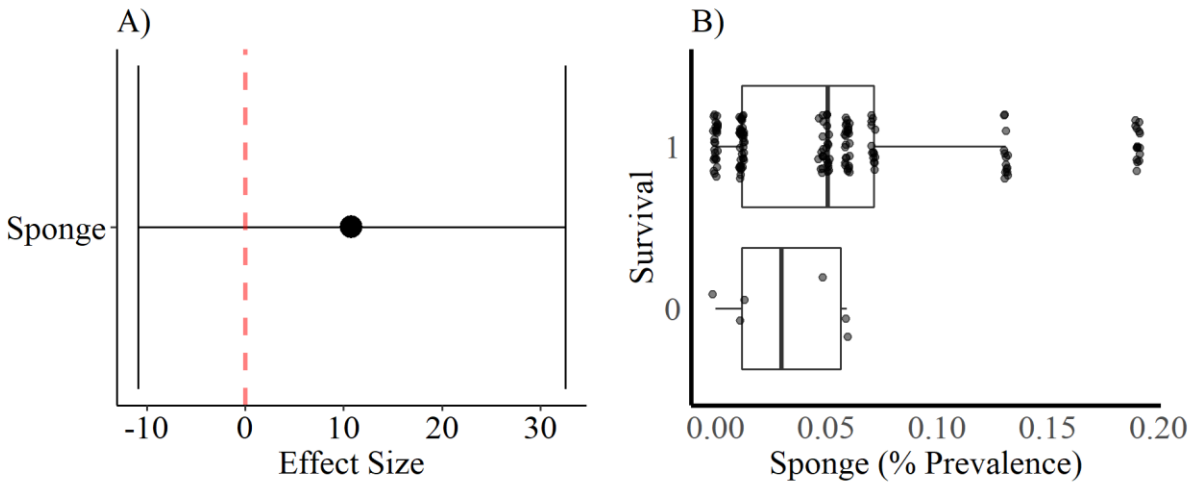

Figure 18: Conditions and benthic taxa that effected *M. cavernosa* survival. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B) Box plot with jittered points explaining survival (1 = alive, 0 = dead) vs benthic community and condition, with points representing the survival of each individual microfragment.

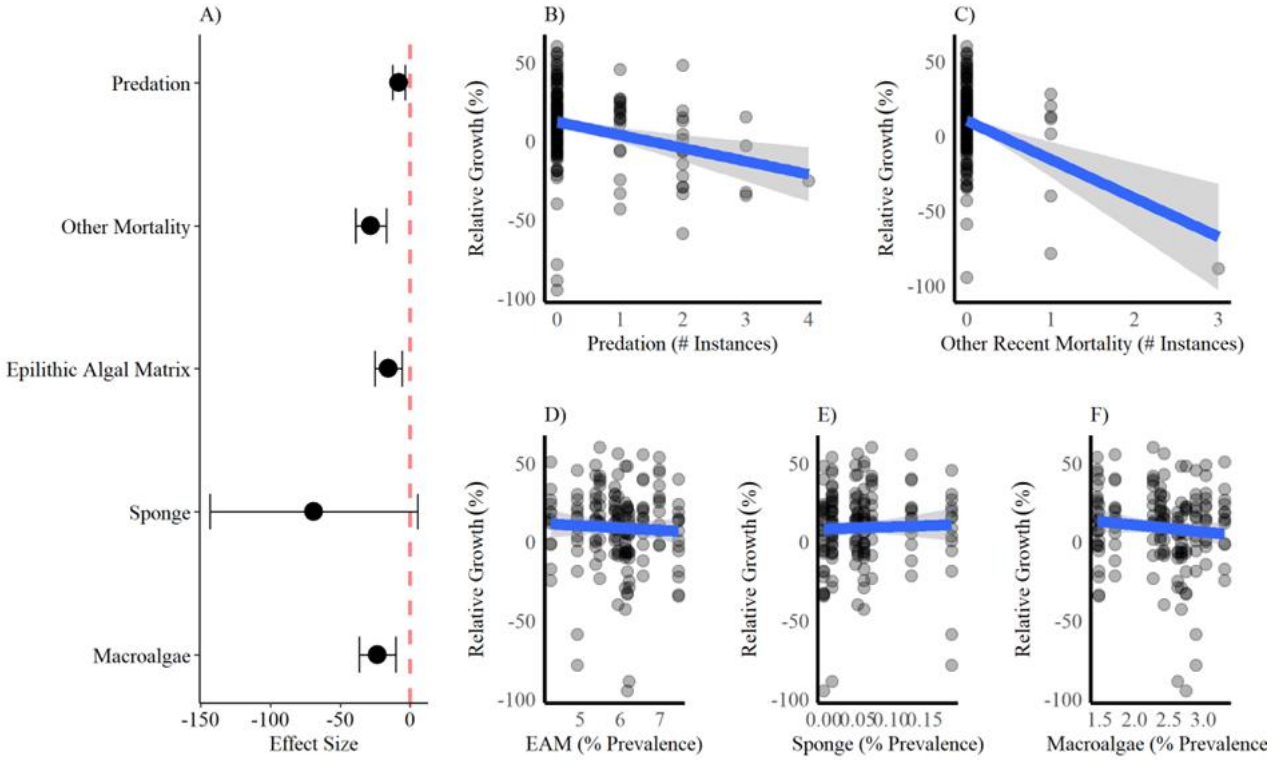

Figure 19: Conditions and benthic taxa that effected *M. cavernosa* relative growth. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on relative growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-F) Relative growth vs benthic community or condition, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate.

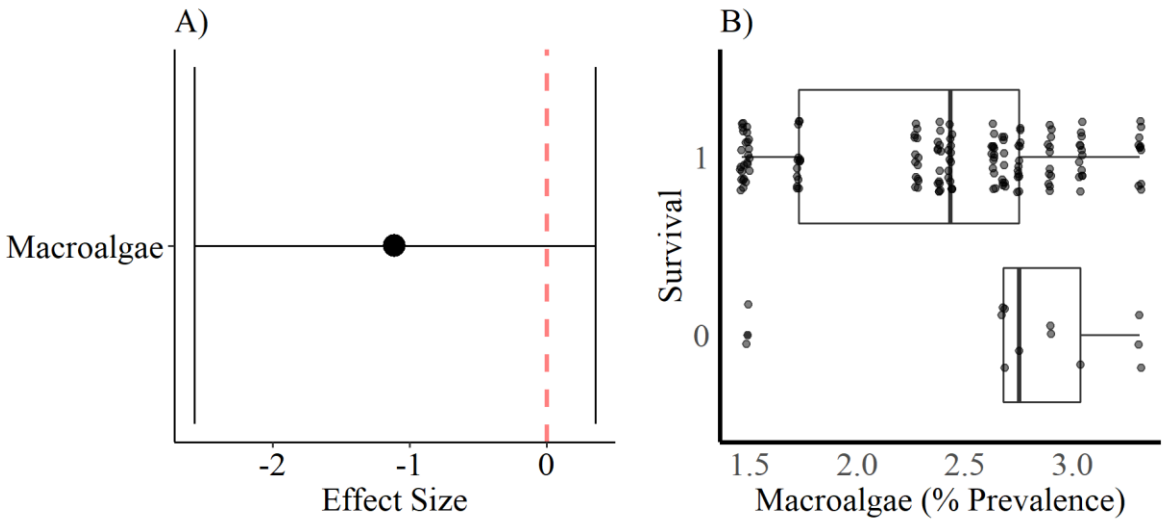

Figure 20: Conditions and benthic taxa that effected *P. clivosa* survival. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B) Box plot with jittered points explaining survival (1 = alive, 0 = dead) vs benthic community and condition, with points representing the survival of each individual microfragment.

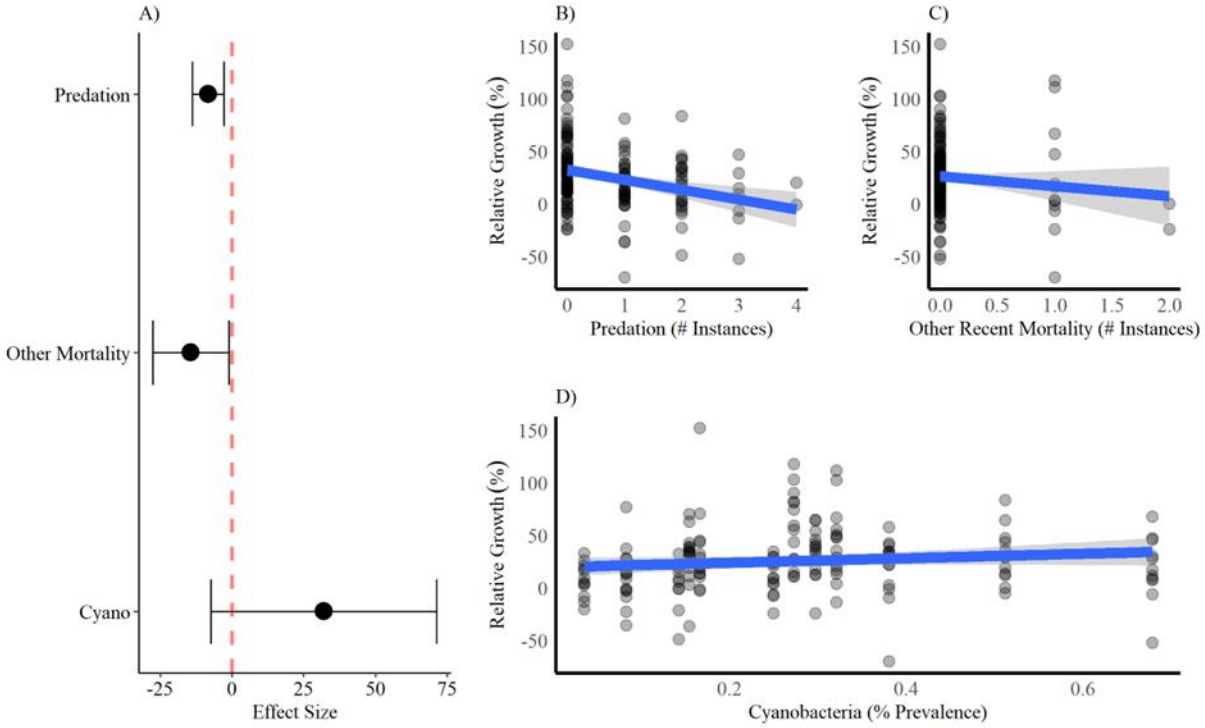

Figure 21: Conditions and benthic taxa that effected *P. clivosa* relative growth. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on relative growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-D) Relative growth vs benthic community or condition, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate.

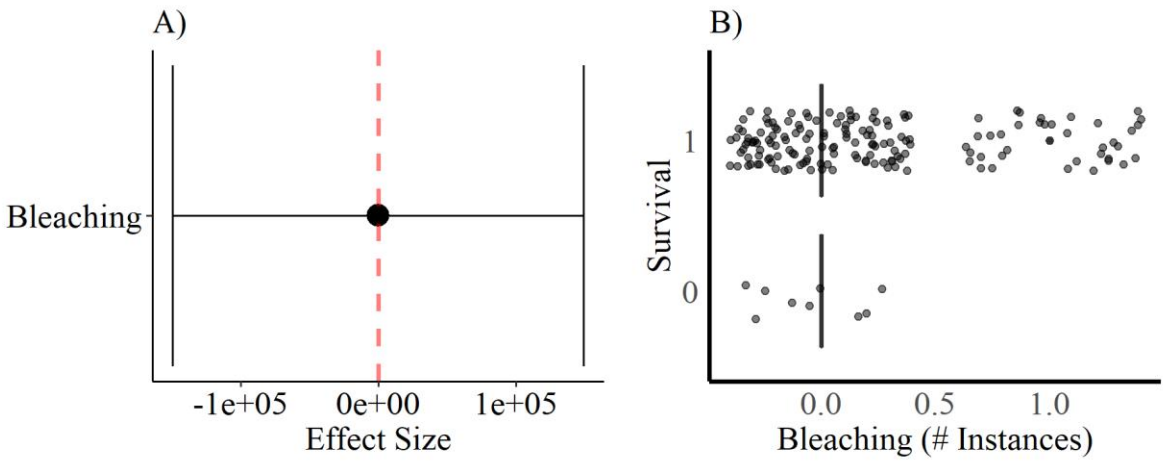

Figure 22: Conditions and benthic taxa that effected *S. siderea* survival. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on survival. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B) Box plot with jittered points explaining survival (1 = alive, 0 = dead) vs benthic community and condition, with points representing the survival of each individual microfragment.

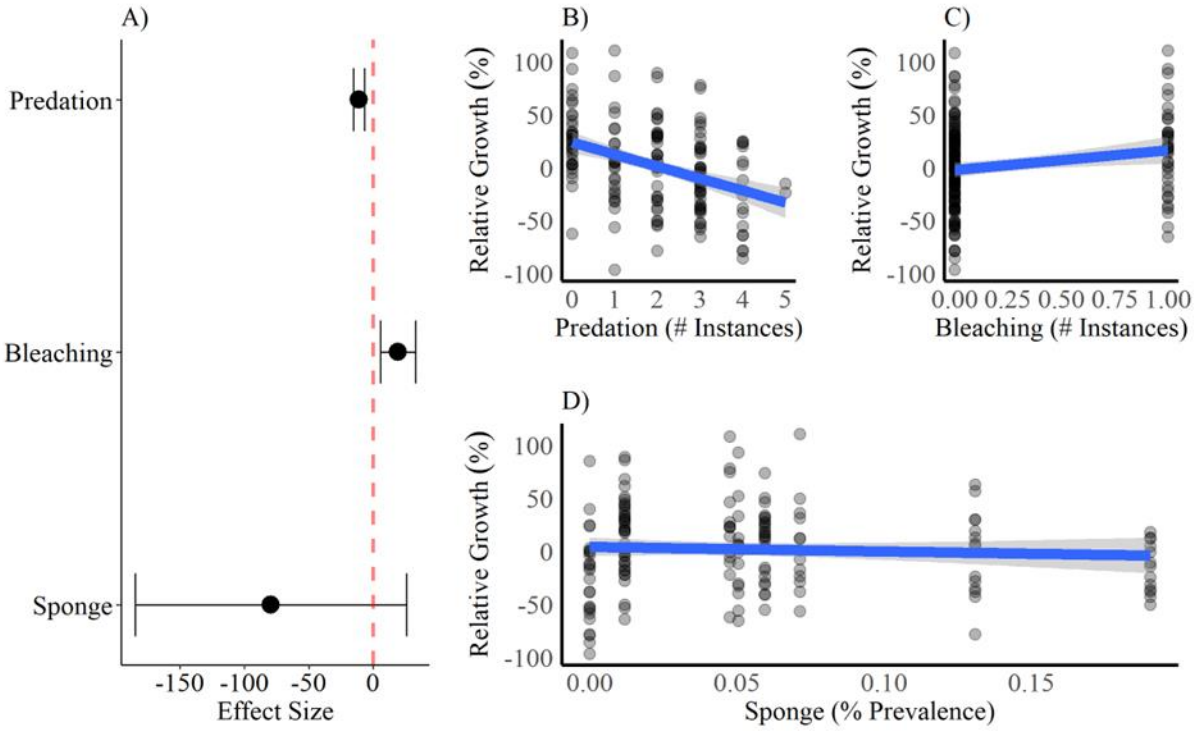

Figure 23: Conditions and benthic taxa that effected *S. siderea* relative growth. A) Forest plot explaining the effect of the benthic community and microfragment health conditions on relative growth. Any point to the left of the red dashed line means that the factor has a negative effect, and any point to the right has a positive effect. B-D) Relative growth vs benthic community or condition, with points representing the relative growth rate of each individual microfragment and the blue regression line representing the trend in mean relative growth rate.

Discussion:

In this study, a rapid assessment method was utilized to quickly characterize multiple reef sites, with a goal to capture the site conditions that may predict success for coral microfragment outplanting. The rapid assessments were successful in defining reef site differences s across the Coral ECA among the measured site characteristics. After 6-months post-outplanting, total microfragment survival was > 94% and was > 85% at all sites. Due to such high survival rates, variations among sites and species were very low; however, certain site characteristics (coral density, rugosity, and wild coral bleaching, predation, and other conditions) were able to predict nearly 30% of the survival variability. Because of this low variation, the only site characteristic

that was able to significantly predict success of any of the three microfragment species was the prevalence of wild coral predation on *S. siderea*, with higher wild coral predation prevalence leading to lower survival in this species. Comparatively, microfragment growth showed more variability among sites and species (20% of variability), with much lower variation between certain site characteristics (maximum relief, species richness, coral size, and wild coral bleaching and predation) which only predicted 7% of the variability. Some variability could be found when observing each species but was much lower (< 13%) in comparison to the overall which was most likely due to the lower sample sizes of a single species. Most notable was that *P. clivosa* growth appeared to be influenced by the conditions of wild corals specifically predation and 'other conditions', whereas *M. cavernosa* and *S. siderea* growth were more influenced by the species richness, and coral size. *Siderastrea siderea* additionally appeared to have improved growth in sites that had higher coral densities and more rugosity, most likely due to < 5cm wild *S. siderea* colonies contributing the most to coral density. These findings show that the makeup of the site characteristics serve as better indicators for outplant survival but play a much smaller role in indicating the growth of the microfragments. Instead, growth was found to be better explained by observing microfragment health conditions and the benthic community on the cement base. The variation in growth found between sites and species accounted for 23% of variability in growth, and the factors that directly affected microfragment mortality (predation, other recent mortality, and the cover of EAM, sponge, and macroalgae) accounted for 16% of the variability.

Trends from the GLMMs observed that coral density, rugosity (covaries with coral density), and presence of health conditions on the wild corals all act as predictors of microfragment outplant survival. Within the coral community, increasing coral density and prevalence of health conditions were found to indicate lower microfragment survival. Each of these measurements correspond to the long-term health of the reef area. Healthy reefs support a greater abundance and diversity of coral species along with other reef-associated organisms. Higher densities of corals in an area additionally makes the area more complex, as density was found to covary with rugosity. Howard et al. (2009) and Eggertsen et al. (2020) found that habitat complexity and percent live coral cover were positively correlated with parrotfish abundance (particularly for adult scraping parrotfish). This suggests that areas with more corals and higher complexity are preferred habitats of parrotfish which in turn drives predation rates on outplanted corals. The addition of new,

unacclimated, and stressed corals would provide abundant corallivores, like parrotfish, with easily obtainable nutrients.

Microfragment outplant success is not only predicted by the structure of the site coral community, but also by the condition of the wild corals. Prevalence of predation and 'other recent mortality' conditions on wild corals (such as from sediment, overgrowth interactions, and disease) were strong indicators of microfragment survival and influenced microfragment growth, particularly for *P. clivosa* and *S. siderea*. At sites with greater predation prevalences on the wild corals, it is likely that outplants at those sites will also experience predation, hindering their growth and survival. Parrotfish are known to occupy home ranges; high frequencies of predation on wild colonies could indicate that a large terminal phase male and a harem of initial phase females frequently forage in that area (Manning & McCoy, 2022). 'Other recent mortality' conditions strongly indicate microfragment survival and should be categorized more specifically in future assessments. These conditions can indicate ecosystem level occurrences at a site that can be exerted on outplanted microfragments. For instance, high prevalences of disease show that a disease event in the area is occurring or poor water quality, or high rates of sedimentation show that the area gets affected severely by sediment movement and storm events.

Interestingly, these results show that prevalence of bleaching on wild coral colonies had a positive association with microfragment success. Although it was not found to be significant in microfragment survival, it was for microfragment growth. One possible explanation for this finding was that high levels of bleaching occurred in the Coral ECA in September 2022 with low amounts of bleaching continuing through January 2023, when the rapid assessments took place (Florida Department of Environmental Protection, 2022; personal observation). Although the high temperatures that induced bleaching had dissipated, sites where greater bleaching prevalence was observed could have still been relatively warmer than other areas. This could provide outplanted microfragments with more favorable conditions for increased growth as colder temperatures reduce the photosynthetic capabilities of corals (Saxby et al., 2003). An alternate explanation is that parrotfish may rely on coral microalgal endosymbionts for nutrients. Although the direct causes for parrotfish corallivory are undetermined (Rotjan & Dimond, 2010), higher abundances of bleached corals may signal unstable/unproductive regions, causing parrotfish to focus grazing intensity elsewhere, thus releasing colonies from predation pressures.

Of the observed factors that appeared to affect microfragment survival in all of the models (overall and for each species) none were significant, due to the overall high survival rates observed across all sites. Because of this, microfragment relative growth was better able to capture the conditions that primarily affect microfragment success. Microfragment growth significantly declined with increasing predation and 'other recent mortality' instances on the microfragments, with predation having the strongest relationship for all three species. Observed predation rates on microfragments were similar to other coral microfragment outplant studies, which show relatively high predation within the first month post-outplanting, then decline to near zero (Koval et al., 2020; Page et al., 2018; Quimpo et al., 2020; Rivas et al., 2021). Bleaching, while being a well-documented cause of coral decline (Hoegh-Guldberg et al., 2017), did not appear to affect growth because it only occurred in a few microfragments. Several microfragments of *M. cavernosa* from one parent colony experienced partial bleaching likely due to stress from outplanting. This was most likely because these microfragments were spread across several sites and all originated from the same parent colony. Partial bleaching occurred one-week post-outplanting and all microfragments recovered by the three-month monitoring period. Some *S. siderea* microfragments began to bleach at the 6-month monitoring period which was toward the beginning of the warmer summer months.

The components of the immediate benthic community around microfragments also affected their growth, with increasing percent cover of EAM, sponge, and macroalgae all negatively impacting growth. In this study, it was common for microfragments to come in direct contact with macroalgae and instances of macroalgae shading microfragments were witnessed. Although no microfragments were in direct contact with sponges, their presence on the base significantly affected growth, possibly through allelochemicals at a distance (Wulff, 2006). EAM does not overgrow corals like macroalgae does, but it can affect coral growth, as the algae and sediment that make up EAM collects on top of the substrate, either burying the tissue extensions of the coral or preventing it from growing outwards (Nugues & Roberts 2003; Rogers, 1990; Wilson & Bellwood, 1997).

Differences in survival and growth among species were captured in this study. *Montastraea cavernosa* displayed moderate growth and the highest survival rates. The calculated relative growth rate per year for *M. cavernosa* was consistent with previous studies looking at

unfragmented colonies; however, the linear extension was lower (Crabbe, 2009; Jones et al., 2023; Manzello et al., 2015). *Pseudodiploria clivosa* exhibited the lowest survival yet had the highest relative growth. *Pseudodiploria* spp. appear to be preferred by parrotfish as *P. strigosa* microfragments have higher mortality from predation rates compared to other species, suggesting that parrotfish predation may contribute to the low survival rates observed in *P. clivosa* in this study (Harrell & Lirman, 2023; Koval et al., 2020). Relative growth in *P. clivosa* is not well recorded, but compared to unfragmented *P. strigosa* colonies, the linear extension of microfragments was lower (Rippe et al., 2018). *Siderastrea siderea* was the only species that did not exhibit positive relative growth or linear extension, most likely due to the intense levels of predation observed on this species. Compared to unfragmented colonies, the *S. siderea* microfragments have much lower growth rates (Crabbe, 2009; Elahi & Edmunds, 2007; Jones et al., 2023). Wild *S. siderea* colonies are observed to have high densities and recruitment across Florida reefs (Jones, 2022), and in this study had the highest density of wild colonies across all sites; suggesting that environmental conditions for this species should favor growth, but that was not observed in this study. Elahi and Edmunds (2007) found that small asexual *S. siderea* colonies grew much slower compared to small colonies of sexual origin. Additionally, *S. siderea* colonies in Florida reach sexual maturity at sizes as small as 1 cm in diameter (St. Gelais et al., 2016). It could then be suggested that the *S. siderea* microfragments used in this study are behaving similarly to the findings of Elahi and Edmunds (2007) and St. Gelais (2016). Based on the relative growth trends of the three species used in this study, *P. clivosa* is the species best suited for microfragmentation.

The assessments used in this study were completed to obtain specific measured values to define a site to reveal patterns in outplant success based on the characteristics. Despite such high survival, the measured site characteristics were found to explain some variability in microfragment success. Such little variation between survival and growth may be due to the short study duration that only spanned 6 months primarily through the colder seasons. Further monitoring can help determine relationships between the interaction of microfragments and the benthic community, as well as the effect of the warmer summer waters. Additionally, there are some unexplainable indicators such as the prevalence of bleaching on wild corals, a process known to drive coral mortality, that was found to indicate improved microfragment survival. Coral reefs are dynamic with many organisms and processes driving ecosystem functioning (Brandl et al., 2019), so

analyzing only the stony coral community and habitat complexity is leaving out many other ecological important taxa that may account for more variability. Therefore, including other functional groups within a rapid site assessment may improve results.

My study was able to find features within reef sites that are responsible for indicating microfragment outplant success, mainly survival. Restoration practitioners may not have the time to implement these rapid assessments exactly, as analyzing data for multiple potential outplant sites can be time consuming. Instead, practitioners can utilize the findings of this study along with a more visual based ranking system on habitat complexity (Wilson et al., 2007) and the stony coral community to gain understanding of the area before outplanting. Coral reefs are on the decline worldwide, with a need for coral restoration. Particularly in Florida, low abundances of wild corals hinder restoration efforts by the number of corals they can work with to propagate and outplant. That is why site selection is important for coral restoration because understanding how site characteristics influence outplant success, practitioners can maximize outplant success.

References

Abramoff, M. D., Magalhaes, P. J., & Ram, S. J. (2004). Image processing with ImageJ. *Biophotonics International*, *11*(7), 36-42.

Banks, K. W., Riegl, B. M., Shinn, E. A., Piller, W. E., & Dodge, R. E. (2007). Geomorphology of Southeast Florida continental reef tract (Miami-Dade, Broward, and Palm Beach Counties, USA). *Coral Reefs*, *26*, 617-633. DOI:10.1007/s00338-007-0231-0

Banks, K. W., Riegl, B. M., Richards, V. P., Walker, B. K., Helmle, K. P., Jordan, L. K. B., Phipps, J., Shivji, M. S., Spieler, R. E., & Dodge, R. E. (2008). The reef tract of continental Southeast Florida (Miami-Dade, Broward and Palm Beach Counties, USA). *Coral Reefs of the USA*, *5*, 175-220. DOI:10.1007/978-1-4020-6847-8_5

Bellwood, D. R., Hughes, T. P., Folke, C., & Nystrom, M. (2004). Confronting the coral reef crisis. *Nature*, *429*. DOI:10.1038/nature02691

Boström-Einarsson, L., Babcock, R. C., Bayraktarov, E., Ceccarelli, D., Cook, N., A. Ferse, S. C., Hancock, B., Harrison, P., Hein, M., Shaver, E., Smith, A., Suggett, D., Stewart-Sinclair, P. J., Vardi, T., & McLeod, I. M. (2020). Coral restoration – A systematic review of current methods, successes, failures and future directions. *PLOS ONE*, *15*(1), e0226631. https://doi.org/10.1371/journal.pone.0226631

Bracken, M. E. S., Gonzalez-Dorantes, C. A., & Stachowicz, J. J. (2007). Whole-community mutualism: Associated invertebrates facilitate a dominant habitat-forming seaweed. *Ecology*, *88*(9), 2211-2219. https://doi.org/10.1890/06-0881.1

Bradley, P., Fisher, W. S., Bell, H., Davis, W., Chan, V., LoBue, C., & Wiltse, W. (2009). Development and implementation of coral reef biocriteria in U.S. jurisdictions. *Environmental Monitoring and Assessment*, *150*, 43-51. DOI:10.1007/s10661-008-0670-2

Brandl, S. J., Rasher, D. B., Cote I. M., Casey, J. M., Darling, E. S., Lefcheck, J. S., & Duffy, J. E. (2019). Coral reef ecosystem functioning: Eight core processes and the role of biodiversity. *Frontiers in Ecology and the Environment*, *17*(8), 445-454. https://doi.org/10.1002/fee.2088

Brooks, M. E., Kristensen, K., van Benthem, K. J., Magnusson, A., Berg, C. W., Nielsen, A., Skaug, H. J., Machler, M., & Bolker, B. M. (2017). GlmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *The R Journal*, *9*, 378-400.

Burkepile, D. E., Adam, T. C., Roycroft, M., Ladd, M. C., Munsterman K. S., & Ruttenberg, B. I., (2019). Species-specific patterns in corallivory and spongivory among Caribbean parrotfishes. *Coral Reefs*, *38*, 417-423. https://doi.org/10.1007/s00338-019-01808-6

Crabbe, M. J. C. (2009). Scleractinian coral population size structures and growth rates indicate coral resilience on the fringing reefs of North Jamaica. *Marine Environmental Research*, *67*, 189-198. DOI:10.1016/j.marenvres.2009.01.003

Darling, E. S., Alvarez-Filip, L., Oliver, T. A., McClanahan, T. R., & Cote, I. M. (2012). Evaluating life history strategies of reef corals from species traits. *Ecology Letters, 15,* 1378-1386. https://doi.org/10.1111/j.1461-0248.2012.01861.x

Dustan, P., Doherty, O., & Pardede, S. (2013). Digital reef rugosity estimates coral reef habitat complexity. *PLOS ONE, 8*(2), e57386. https://doi.org/10.1371/journal.pone.0057386

Eggertsen, M., Chacin, D. H., Van Lier, J., Eggertsen, L., Fulton, C. J., Wilson, S., Halling, C., & Berkstrom, C. (2020). Seascape configuration and fine-scale habitat complexity shape parrotfish distribution and function across a coral reef lagoon. *Diversity, 12,* 391. DOI: 10.3390/d12100391

Elahi, R., & Edmunds, P. J. (2007). Consequences of fission in the coral *Siderastrea siderea*: Growth rates of small colonies and clonal input to population structure. *Coral Reefs, 26,* 271-276. https://doi.org/10.1007/s00338-006-0190-x

Epstein, N., Bak, R. P. M., & Rinkevich, B. (2003). Applying forest restoration principles to coral reef rehabilitation. *Aquatic Conservation: Marine and Freshwater Ecosystems, 13,* 387-395. https://doi.org/10.1002/aqc.558

Figueiredo, J. D. Gilliam, D. A. Renegar, & B.K. Walker. (2021). Coral propagation: Landbased and offshore nursery (phase II). Final Report. Florida DEP.

Florida Department of Environmental Protection, (2022). SEAFAN BleachWatch program current conditions report #20221126.

Forsman, Z. H., Rinkevich, B., & Hunter, C. L. (2006). Investigating fragment size for culturing reef-building corals (*Porites lobata* and *P. compressa*) in *ex situ* nurseries. *Aquaculture, 261*, 89-97. https://doi.org/10.1016/j.aquaculture.2006.06.040

Forsman, Z. H., Page, C. A., Toonen, R. J., & Vaughan, D. (2015). Growing coral larger and faster: Micro-colony-fusion as a strategy for accelerating coral cover. *PeerJ, 3,* e1313. https://doi.org/10.7717/peerj.1313

Fukunaga, A., Kosaki, R. K., Pascoe, K. H., & Burns, J. H. R. (2020). Fish assemblage structure in the Northwestern Hawaiian Islands is associated with the architectural complexity of coral-reef habitats. *Diversity, 12,* 430. DOI: 10.3390/d12110430

Gardner, T. A., Cote, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. *Science, 301*(5635). DOI: 10.1126/science.1086050

Gilliam, D. S., Hayes, N. K., Ruzicka, R., & Colella, M. (2021). Southeast Florida coral reef evaluation and monitoring project 2020 year 18 final report. *Florida DEP & FWC*.

Graham, N. A. J., & Nash, K. L. (2013). The importance of structural complexity in coral reef ecosystems. *Coral Reefs, 32,* 315-326. https://doi.org/10.1007/s00338-012-0984-y

Gratwicke, B., & Speight, M. R. (2005). Effects of habitat complexity on Caribbean marine fish assemblages. *Marine Ecology Progress Series, 292,* 301-310. DOI: 10.3354/meps292301

Harrell, C., & Lirman, D. (2023). *Dictyota* defense: Developing effective chemical protection against intense fish predation for outplanted massive corals. *PeerJ, 11*, e14995. https://doi.org/10.7717/peerj.14995

Hartig, F. (2017) DHARMa: Residual diagnostics for hierarchical (multi-level/mixed) regression models R package. https://cran.r-project.org/web/packages/DHARMa/vignettes/DHARMa.html

Hayes, N. K., Walton, C. J., & Gilliam, D. S. (2022). Tissue loss disease outbreak significantly alters the Southeast Florida stony coral assemblage. *Frontiers in Marine Science, 9*, 975894. https://doi.org/10.3389/fmars.2022.975894

Helder, N. K., Burns, J. H. R., & Green, S. J. (2022). Intra-habitat structural complexity drives the distribution of fish trait groups on coral reefs. *Ecological Indicators*, *142*. https://doi.org/10.1016/j.ecolind.2022.109266

Hoegh-Guldberg, O., Poloczanska, E. S., Skirving, W., & Dove, S. (2017). Coral reef ecosystems under climate change and ocean acidification. *Frontiers in Marine Science*, *4*(158). https://doi.org/10.3389/fmars.2017.00158

Howard, K. G., Schumacher, B. D., & Parrish, J. D. (2009). Community structure and habitat associations of parrotfishes on Oahu, Hawaii. *Environmental Biology of Fishes*, *85*, 175-186. DOI: 10.1007/s10641-009-9478-3

Hughes, T. P. (1994). Catastrophes, phase shifts, and large scale degradation of a Caribbean coral reef. *Science*, *265*, 1547-1551. DOI: 10.1126/science.265.5178.1547

Hughes, T. P., Kerry, J. T., Alvarez-Noriega, M., Alvarez-Romero, J. G., Anderson, K. D., Baird, A. H., Babcock, R. C., Beger, M., Bellwood, D. R., Berkelmans, R., Bridge, T. C., Butler, I. R., Byrne, M., Cantin, N. E., Comeau, S., Connolly, S. R., Cumming, G. S., Dalton, S. J., Diaz-Pulido, G., … & Wilson, S. K. (2017). Global warming and recurrent mass bleaching of corals. *Nature, 543*, 373-377. DOI: 10.1038/nature21707

Jayewardene, D., Donahue, M. J., & Birkeland, C. (2009). Effects of frequent fish predation on corals in Hawaii. *Coral Reefs*, *28*, 499-506. DOI:10.1007/s00338-009-0475-y

Jones, N. P., Figueiredo, J., & Gilliam, D. S. (2020). Thermal stress-related spatiotemporal variations in high-latitude coral reef benthic communities. *Coral Reefs*, *39*, 1661-1673. https://doi.org/10.1007/s00338-020-01994-8

Jones, N. P. (2022). Benthic community dynamics and stony coral demographics in Florida in relation to acute disturbances and chronic pressures. [Doctoral dissertation, Nova Southeastern University]. *NSUWorks*, (104) https://nsuworks.nova.edu/hcas_etd_all/104.

Jones, N. P., Pratchett, M. S., Milligan, R. J., & Gilliam, D. S. (2023). High incidence of partial colony mortality constrains realized growth for three coral species in southeast Florida. Marine *Ecology Progress Series*, *721*, 59-70. DOI:10.3354/meps14420

Kohler, K. E., & Gill, S.M. (2006). Coral point count with excel extensions (CPCe): A visual basic program for the determination of coral and substrate coverage using random point

count methodology. *Computers and Geosciences*, *32*(9), 1259-1269, DOI: 10.1016/j.cageo.2005.11.009.

Komsta, L. (2022). Moments, cumulants, skewness, kurtosis, and related tests. https://cran.r-project.org/web/packages/moments/moments.pdf

Koval, G., Rivas, N., D'Alessandro, M., Hesley, D., Santos, R., & Lirman, D. (2020). Fish predation hinders the success of coral restoration efforts using fragmented massive corals. *PeerJ*, *8*, e9978. https://doi.org/10.7717/peerj.9978

Kramer, P. A. (2003). Synthesis of coral reef health indicators for the western Atlantic: Results of the AGRRA program (1997-2000). *Atoll Research Bulletin*, *496*, 1-57. https://doi.org/10.5479/si.00775630.496-3.1

Kuffner, I. B., Brock, J. C., Grober-Dunsmore, R., Bonito, V. E., Hickey, T. D., & Wright, C. W. (2007). Relationships between reef fish communities and remotely sensed rugosity measurements in Biscayne National Park, Florida, USA. *Environmental Biology of Fishes*, *78*, 71-82. https://doi.org/10.1007/s10641-006-9078-4

Kuffner, I. B., & Toth, L. T. (2016). A geological perspective on the degradation and conservation of Western Atlantic coral reefs. *Conservation Biology*, *30*(4), 706-715. DOI: 10.1111/cobi.12725

Lirman, D. (2000). Fragmentation in the branching coral *Acropora palmata* (Lamarck): Growth, survivorship, and reproduction of colonies and fragments. *Journal of Experimental Marine Biology and Ecology*, *251*, 41-57. DOI:10.1016/S0022-0981(00)00205-7

Lirman, D. (2001). Competition between macroalgae and corals: Effects of herbivore exclusion and increased algal biomass on coral survivorship and growth. *Coral Reefs*, *19*, 392-399. https://doi.org/10.1007/s003380000125

Loya, Y., Sakai, K., Yamazato, K., Nakano, Y., Sambali, H., & van Woesik, R. (2001). Coral bleaching: The winners and losers. *Ecology Letters*, *4*, 122-131. https://doi.org/10.1046/j.1461-0248.2001.00203.x

Lustic, C., Maxwell, K., Bartels, E., Reckenbeil, B., Utset, E., Schopmeyer, S., Zink, I., & Lirman, D. (2020). The impacts of competitive interactions on coral colonies after transplantation: A multispecies experiment from the Florida Keys. *US. Bulletin of Marine Science*, *96*, 805-818. https://doi.org/10.5343/bms.2019.0086

Maliao, R. J., Turingan, R. G., & Lin, J. (2008). Phase-shift in coral reef communities in the Florida Keys National Marine Sanctuary (FKNMS), USA. *Marine Biology*, *154*, 841-853. DOI:10.1007/s00227-008-0977-0

Manning, J. C., & McCoy, S. J. (2022). Territoriality drives patterns of fixed space use in Caribbean parrotfishes. *Ecology and Evolution*, *13*, e9833. https://doi.org/10.1002/ece3.9833

Manzello, D. P., Enochs, I. C., Kolodziej, G., & Carlton, R. (2015). Coral growth patterns of *Montastraea cavernosa* and *Porites astreoides* in the Florida Keys: The importance of

thermal stress and inimical waters. *Journal of Experimental Marine Biology and Ecology, 471*, 198-207. https://doi.org/10.1016/j.jembe.2015.06.010

McField, M., & Kramer, P. R. (2007). Healthy reefs for healthy people: A guide to indicators of reef health and social well-being in the Mesoamerican Reef Region. With contributions by M. Gorrez and M. McPherson. 208 pp. https://www.researchgate.net/publication/228627962

McKenna, S. A., & Etnoyer, P. (2010). Rapid assessment of stony coral richness and condition on Saba Bank, Netherlands Antilles. *PLOS ONE*, *5*(5), e10749. https://doi.org/10.1371/journal.pone.0010749

Moyer, R. P., Riegl, B., Banks, K., & Dodge, R. E. (2003). Spatial patterns and ecology of benthic communities on a high-latitude South Florida (Broward County, USA) reef system. *Coral Reefs, 22,* 447-464. DOI: 10.1007/s00338-003-0334-1

Norström, A. V., Nyström, M., Lokrantz, J., & Folke, C. (2009). Alternative states on coral reefs: Beyond coral-macroalgal phase shifts. *Marine Ecology Progress Series, 376,* 295-306. https://doi.org/10.3354/meps07815

Nugues, M. M., & Roberts, C. M. (2003). Coral mortality and interaction with algae in relation to sedimentation. *Coral Reefs, 22,* 507-516. https://doi.org/10.1007/s00338-003-0338-x

Oakley-Cogan, A., Tebbett, S. B., & Bellwood, D. R. (2020). Habitat zonation on coral reefs: Structural complexity, nutritional resources and herbivorous fish distributions. *PLOS ONE, 15*(6), e0233498. https://doi.org/10.1371/journal.pone.0233498

Page, C. A., Muller, E. M., & Vaughan, D. E. (2018). Microfragmenting for the successful restoration of slow growing massive corals. *Ecological Engineering, 123,* 86-94. https://doi.org/10.1016/j.ecoleng.2018.08.017

Paul, V. J., Kuffner, I. B., Walters, L. J., Ritson-Williams, Beach, K. S., & Becerro, M. A. (2011). Chemically mediated interactions between macroalgae *Dictyota* spp. And multiple life-history stages of the coral *Porites astreoides*. *Marine Ecology Progress Series, 426,* 161-170. http://www.jstor.org/stable/24874661

Pratchett M. S., Anderson K. D., Hoogenboom M. O., Widman E., Baird A. H., Pandolfi J. M., Edmunds P. J., & Lough J. M. (2015). Spatial, temporal and taxonomic variation in coral growth-implications for the structure and function of coral reef ecosystems. *Oceanography and Marine Biology: An Annual Review, 53,* 215-295. https://www.researchgate.net/publication/280570769

Precht, W. F., & Miller, S. L. (2007). Ecological shifts along the Florida Reef Tract: The past as a key to the future. In: Aronson R. B., editor. *Geological Approaches to Coral Reef Ecology*. New York, NY; 2007. (pp. 237-312). DOI:10.1007/978-0-387-33537-7_9

Price, A. R. G., & Harris, A. (2009). Decadal changes (1996-2006) in coastal ecosystems of the Chagos archipelago determined from rapid assessment. *Aquatic Conservation: Marine and Freshwater Ecosystems, 19*(6). DOI:10.1002/aqc.1029

QGIS.org. (2023). QGIS geographic information system. *Open Source Geospatial Foundation Project*. http://qgis.org.

Quimpo, T. J. R., Cabaitan, P. C., & Hoey, A. S. (2020). Detachment of *Porites cylindrica* nubbins by herbivorous fishes. *Restoration Ecology*, *28*(2), 418-426. https://doi.org/10.1111/rec.13091

Rinkevich, B. (1995). Restoration strategies for coral reefs damaged by recreational activities: The use of sexual and asexual recruits. *Restoration Ecology*, *3*, 241-251. https://doi.org/10.1111/j.1526-100X.1995.tb00091.x

Rinkevich, B. (2000). Steps towards the evaluation of coral reef restoration by using small branch fragments. *Marine Biology, 136*, 807-812. https://doi.org/10.1007/s002270000293

Rippe, J. P., Baumann, J. H., De Leener, D. N., Aichelman, H. E., Friedlander, E. B., Davies, S. W., & Castillo, K. D. (2018). Corals sustain growth but not skeletal density across the Florida Keys Reef Tract despite ongoing warming. *Global Change Biology*, *24*(11), 5205-5217. https://doi.org/10.17615/88sf-h976

Rivas, N., Hesley, D., Kaufman, M., Unsworth, J., D'Alessandro, M., & Lirman, D. (2021). Developing best practices for the restoration of massive corals and the mitigation of predation impacts: Influences of physical protection, colony size, and genotype on outplant mortality. *Coral Reefs*, *40*, 1227-1241. DOI: 10.1007/s00338-021-02127-5

Rogers, C. S., Suchanek, T. H., & Pecora, F. A. (1982). Effects of hurricanes David and Frederic (1979) on shallow *Acropora palmata* reef communities: St. Croix, U.S. Virgin Islands. *Bulletin of Marine Science*, *32*(2), 532-548.

Rogers, C. S. (1990). Responses of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series*, *62*, 185-202. http://dx.doi.org/10.3354/meps062185

Rotjan, R. D., & Dimond, J. L. (2010). Discriminating causes from consequences of persistent parrotfish corallivory. *Journal of Experimental Marine Biology and Ecology*, *390*, 188-195.

RStudio Team. (2020). RStudio: Integrated development for R. RStudio. http://www.rstudio.com/.

Saxby, T., Dennison, W. C., & Hoegh-Guldberg, O. (2003). Photosynthetic responses of the coral *Montipora digitata* to cold temperature stress. *Marine Ecology Progress Series*, *248*, 85-97. DOI: 10.3354/meps248085

Schlecker, L., Page, C., Matz, M., & Wright, R. M. (2022). Mechanisms and potential immune tradeoffs of accelerated coral growth induced by microfragmentation. *PeerJ*, *10*, e13158 . https://doi.org/10.7717/peerj.13158

Shantz, A. A., Stier, A. C., & Idjadi, J. A. (2011). Coral density and predation affect growth of a reef-building coral. *Coral Reefs*, *30*, 363-367. DOI: 10.1007/s00338-010-0694-2

Soper, D. M., Villafranca, N., Dieffenthaller, J. P., Dieffenthaller, M., Tompkins, H., Weglarz, M., Vaughan, D., Hamlyn, S. B., & Stenesen, D. (2022). Growth and Cyclin-E

expression in the stony coral species *Orbicella faveolata* post-microfragmentation. *The Biological Bulletin*, *242*(1), 40-47. https://doi.org/10.1086/717926

St. Gelais, A. T., Chaves-Fonnegra, A., Brownlee, A. S., Kosmynin, V. N., Moulding, A. L., 7 Gilliam, D. S. (2016). Fecundity and sexual maturity of the coral *Siderastrea siderea* at high latitude along the Florida Reef Tract, USA. *Invertebrate Biology*, *135*(1), 46-57. http://www.jstor.org/stable/24698436

Stein, J., & Ruzicka, R. (2021). Florida reef resilience program disturbance response monitoring 2021 quick look report. *Fish and Wildlife Research Institute*.

Steneck, R. S., Arnold, S. N., Boenish, R., de León, R., Mumby, P. J., Rasher, D. B., & Wilson, M. W. (2019). Managing recovery resilience in coral reefs against climate-induced bleaching and hurricanes: A 15 year case study from Bonaire, Dutch Caribbean. *Frontiers in Marine Science*, *6*, 265. https://doi.org/10.3389/fmars.2019.00265

Toth, L. T., Van Woesik, R., Murdoch, T. J. T., Smith, S.R., Ogden, J.C., Precht, W.F., & Aronson, R.B. (2014). Do no-take reserves benefit Florida's corals? 14 years of change and stasis in the Florida Keys National Marine Sanctuary. *Coral Reefs*, *33*, 565-577. 10.1007/s00338-014-1158-x

Walker, B. K., Riegl, B., & Dodge, R. E. (2008). Mapping coral reef habitats in Southeast Florida using a combined technique approach. *Journal of Coastal Research*, *24*(5). https://doi.org/10.2112/06-0809.1

Walton, C. J., Hayes, N. K., & Gilliam, D. S. (2018). Impacts of a regional, multi-year, multi-species coral disease outbreak in Southeast Florida. *Frontiers in Marine Science*, *5*, 323. https://doi.org/10.3389/fmars.2018.00323

Wilson, S., & Bellwood, D. R. (1997). Cryptic dietary components of territorial damselfishes (Pomacentridae, Labroidei). *Marine Ecology Progress Series*, *153*, 299-310. DOI: 10.3354/meps153299

Wilson, S. K., Graham, N. A. J., & Polunin, N. V. C. (2007). Appraisal of visual assessments of habitat complexity and benthic composition on coral reefs. *Marine Biology*, *151*, 1069-1076. https://doi.org/10.1007/s00227-006-0538-3

Wulff, J. L. (2006). Ecological interactions of marine sponges. *Canadian Journal of Zoology*, *84*, 146-166. https://doi.org/10.1139/z06-019