A Meta-Analysis Integrating Field Data to Quantify Corallivorous Fish Impacts on Coral Outplant Success through a Spatiotemporal Predation Pressure Index

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Final Report

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Management Summary

Corallivorous fish predation is a major obstacle to coral restoration success in Florida, particularly for massive morphology species. This project addressed critical knowledge gaps by synthesizing nearly a decade of coral predation data from the University of Miami and Nova Southeastern University, spanning reef sites in Miami-Dade County, Biscayne National Park, and the Kristin Jacobs Coral Aquatic Preserve. Across five NSU outplanting studies (2021–2026), 20% of 2,475 monitored corals showed evidence of predation, with *Colpophyllia natans* experiencing the highest prevalence across studies and sites. Across 15 UM outplanting studies (2018–2025), encompassing 3,667 corals across 14 species and 26 reefs, overall predation prevalence was 64.5%, with *Diploria labyrinthiformis* and *Colpophyllia natans* exceeding 85%. To characterize corallivorous fish pressures on coral outplants, a novel Coral-Baited Remote Underwater Video System (C-BRUVS) was designed, constructed, and deployed to characterize predator identity, size class, and feeding behavior without diver interference. The C-BRUVS captured 14 hours of real-time video per day over several days. Across three deployments with recorded predation, stoplight parrotfish dominated corallivory, with foureye butterflyfish and redband parrotfish also contributing substantially. Parrotfish bites were significantly correlated with coral tissue loss, confirming C-BRUVS as a valid tool for quantifying predation pressure. A prevalence-based Predation Pressure Index (PPI) was developed from the meta-analysis dataset. Because paired fish survey data were not collected concurrently with predation monitoring across most contributing studies, the PPI and a complementary parrotfish hot spot analysis of NCRMP reef fish visual census data (2018–2024) are presented as two spatial indicators of predation risk, with the hotspots of corallivorous fish integrated into the Decision Support Tool (SeaSketch). Together, they provide restoration practitioners with a more informed basis for outplant site selection for target coral species and for corallivorous parrotfish.

Executive Summary

Coral restoration in Florida has expanded beyond branching corals to include massive species that are critical for reef structural diversity but are highly susceptible to fish predation, particularly during the early post-outplanting period. Despite this, the identities and spatial distributions of responsible corallivorous fish remained poorly characterized, limiting practitioners' ability to make informed site selection decisions. This project addressed those gaps through three integrated objectives: (1) a meta-analysis of published and unpublished coral predation datasets, (2) design and deployment of an improved Coral-Baited Remote Underwater Video System (C-BRUVS), and (3) development of a spatially explicit Predation Pressure Index (PPI) integrated into Florida's Decision Support System.

The meta-analysis compiled monitoring data from five NSU outplanting studies across 20 reef sites and 10 coral species (n = 2,475 corals, 2021–2026) and 15 UM outplanting studies across 26 reef sites and 14 coral species (n = 3,667 corals, 2018–2025). Overall cumulative predation prevalence was 30.1% for NSU and 64.5% for UM, with substantial variation by species, site, and year across both datasets. For NSU, *Colpophyllia natans* and *Siderastrea siderea* showed the highest predation prevalence (60.0% and 56.8%, respectively). For UM, *Porites astreoides* (95.5%), *Solenastrea bournoni* (94.7%), *Diploria labyrinthiformis* (87.3%), and *Colpophyllia natans* (86.0%) showed the highest predation prevalence across sites and studies.

Five C-BRUVS units were constructed using Raspberry Pi-based camera systems capable of recording up to 40 hours of autonomous daytime footage per deployment (over several days). Three deployments with recorded fish predation, the Coral City Camera site (Port of Miami), Paradise Reef, and Oasis Reef, yielded 1,170 predation events and 2,546 total bites from 22 fish species. Stoplight parrotfish was the major predator across all deployments, followed by foureye butterflyfish and redband parrotfish. At the Oasis Reef deployment, parrotfish bite counts were significantly negatively correlated with change in live tissue area, providing empirical validation of C-BRUVS as a quantitative predation monitoring tool.

Spatial hot spot analyses of NCRMP reef fish visual census data (2018–2024) identified the southern Biscayne ecoregion as a persistent center of corallivorous parrotfish densities, with some species showing gradual northward shifts. These outputs were incorporated as GIS layers into SeaSketch, Florida's coral restoration Decision Support System, enabling practitioners to spatially assess predation risk prior to outplanting. The prevalence-based PPI quantified site-specific predation history at locations where outplanting has already occurred. Taken together, these results advance the field's capacity to predict and mitigate corallivorous fish impacts on restored corals of massive morphologies. Closing this data gap through consistent concurrent monitoring is the most important priority identified by this project for strengthening the predictive power of future PPI iterations across Florida's Coral Reef.

Table of Contents

1. INTRODUCTION

1.1. Background and Context

Coral restoration efforts in Florida have expanded to incorporate non-branching coral species, such as boulder and brain corals, which are important for maintaining the diversity of coral reefs and providing critical habitat. These slower-growing "massive" morphology coral species require greater investment in grow-out, which has been significantly improved by the development of micro-fragmentation and re-skinning of substrate by outplanting genetic clones of the same species in close proximity to encourage fusion and increased growth (Forsman et al. 2015; Page et al. 2018). However, restoration practitioners have also documented greater predation of these outplants by corallivorous fish, limiting their survival and reducing restoration efficiency (Koval et al. 2020; Rivas et al. 2021; Ladd et al. 2024). Although fish predation impacts occur predominantly within the first weeks post-outplanting (Koval et al 2020; Smith et al. 2021), the specific identities, sizes, and life stages of responsible predators remain poorly characterized, limiting practitioners' ability to strategically select sites with reduced predation pressure. Further, the traditional approaches of pairing reef fish visual surveys with assessments of predation damage on wild or restored corals (Burkepile 2012; Koval et al. 2020; Shantz and Ladd 2024) are limited by the temporal misalignment between fish surveys and coral tissue removal measurements. This ultimately weakens our ability to correlate predator abundance with predation outcomes.

There is a wide range of fish taxa known to consume coral tissue and skeleton across ocean basins. The majority of fish predation has been attributed to corallivorous butterflyfish (Chaetodontidae) and parrotfish (Scaridae) (Cole et al. 2008; Rotjan and Lewis 2008). Parrotfish diets are composed of macroalgae, diatoms, and, when available, coral skeleton and tissue (Mumby 2009). When parrotfish consume coral tissue, their bites create grazing scars that often lead to partial or total coral mortality (Welsh et al. 2015; Lirman et al. 2022). Parrotfish may exhibit preferences for the nutritional quality, morphology, and availability of their coral prey, and target coral species differentially (Rotjan and Lewis 2006; Burkepile et al. 2019). Among parrotfishes, only *Sparisoma viride* (Stoplight parrotfish), *Sparisoma aurofrenatum* (Redband parrotfish), and *Scarus vetula* (Queen parrotfish) regularly consume coral in the Caribbean, and have been shown to selectively target specific coral species, including *Orbicella faveolata* and *Porites porites* (Bruckner & Bruckner 1998; Rotjan & Lewis 2006, 2008; Burkepile 2012). Parrotfish size is a critical variable affecting predation impact, as significant excavation of substrate does not occur until fishes exceed approximately 20 cm total length (Shantz et al. 2020). Additionally, coral skeletal morphology plays an important role in predation susceptibility, with perforate skeletons such as those of *Porites* spp. being more readily excavated by corallivores compared to imperforate skeletons (Cameron and Edmunds 2014). Butterflyfish display diverse feeding strategies including consuming tissue from individual coral polyps, removing both coral tissue and skeleton, and feeding on coral mucus, causing minimal damage to the coral tissue (Cole and Pratchett 2011; Lenihan et al. 2011). Regardless of feeding strategy, even occasional or minimal corallivory by butterflyfish can have harmful effects on targeted coral species (Penin et al. 2010).

Despite research documenting predation impacts on restored corals, spatiotemporal relationships of predation on coral outplants remain elusive and inconclusive, with single studies often yielding varying results. Current predation mitigation strategies, including physical cages and other constructed exclusion devices, co-outplanting of *Acropora cervicornis* with massive corals, and chemical deterrents, have demonstrated mixed long-term effectiveness. No large-scale, long-term study of fish predation across a wide variety of coral species in Florida had been undertaken, leaving the general understanding in this field substantially limited.

1.2. Purpose of Project

To address these knowledge gaps, our project synthesized nearly a decade of coral predation datasets from the University of Miami and Nova Southeastern University, encompassing reef sites in Miami-Dade County, Biscayne National Park, and the Kristin Jacobs Coral Aquatic Preserve. We conducted a meta-analysis of published and unpublished studies to identify trends in corallivorous fish predation, developed a Predation Pressure Index (PPI) using multiple statistical approaches, constructed and deployed improved Coral-Baited Remote Underwater Video Systems (C-BRUVS) to test site predictions, and integrated results into Florida's Decision Support System SeaSketch tool to assist restoration practitioners in outplant site selection.

For these studies, corallivory is often documented via direct observation of grazing by SCUBA divers, or through before-and-after photographs (Alwany et al. 2003; Cole et al. 2008). However, the presence of SCUBA divers may influence fish behavior and thus bias visual surveys (Emslie et al. 2018). The use of Baited Remote Underwater Video Systems (BRUVS) has become an important tool for observing fish behavior without diver influence (Whitmarsh et al. 2017). Here, we designed C-BRUVS to use coral fragments as "bait" to attract corallivorous fish, allowing us to observe both predator behavior and preferences without diver interference. This approach provides critical insights into corallivore identity, size classes, and species-specific feeding preferences on different coral species.

The objectives of this project were:

1) **Objective I:** Generate a spatiotemporal dataset for predation impacts in south Florida using existing datasets related to coral predation by corallivorous fish and create a “predation pressure index” (PPI) based on the results of the meta-analysis.

2) **Objective II:** Construct a new C-BRUVS design and conduct pilot deployments to evaluate the efficacy of using 24–48 hour of video footage to quantify predation pressure.

3) **Objective III:** Create a GIS layer with PPIs to be added to existing the Decision Support System (DSS) to aid in site selection for restoration.

2. METHODS

2.1. Literature Review

We conducted a literature review to identify published and unpublished studies that paired fish abundance with coral predation data from reef sites in south Florida. Searches were carried out across five main repositories: Web of Science, the UM Digital Thesis Repository, the NSU Digital Thesis Repository, the Florida Department of Environmental Protection (DEP) report archive, and the National Oceanic and Atmospheric Administration (NOAA) report repository (links below). Further, we filtered for massive corals specifically, including boulder and mounding forms, by corallivorous fishes on Florida's Coral Reef. Based on Neudecker (1979), we targeted reef fish taxa known to feed on scleractinian corals: parrotfishes, wrasses, butterflyfishes, and triggerfishes. To be included, studies had to document fish predation on corals within Florida's Coral Reef.

Our search included five online repositories:

1) *Web of Science* (https://www.webofscience.com/wos/woscc/basic-search)
2) UM Digital Thesis Repository (https://www.library.miami.edu/universityarchives/um-theses-and-dissertations.html, https://scholarship.miami.edu/esploro/)
3) NSU Digital Thesis Repository (https://nsuworks.nova.edu/nsu_etds/)
4) Florida Department of Environmental Protection (DEP)'s report archive website (https://floridadep.gov/rcp/coral/content/coral-disease-response-completed-projects)
5) National Oceanic and Atmospheric Administration (NOAA) report repository (https://www.coris.noaa.gov/data/welcome.html)

2.2. Meta-Analysis

2.2.1. Data Collection

The meta-analysis was conducted using published and unpublished datasets of fish predation impacts for restoration-propagated and outplanted coral species in the Coral Aquatic Preserve, Miami-Dade County, and Biscayne National Park. Data was prepared from both completed and ongoing *in-situ* outplanting and monitoring studies by the following research groups: Dr. Diego Lirman (University of Miami Rosenstiel School for Marine, Atmospheric, and Earth Science) and Dr. David Gilliam (Nova Southeastern University Halmos College of Arts and Sciences). Coral data included **predation prevalence** (proportion of corals affected by predation), and some studies included **predation intensity** (percentage of live coral tissue removed by predation). Some studies also had associated fish survey data, which included species- or family-level abundances and size classes.

2.2.2. Data Analysis

Predation monitoring data were compiled from five independent coral outplanting studies conducted by Nova Southeastern University (NSU), and 15 independent coral outplanting studies conducted by the University of Miami (UM).

For NSU, the five studies included: the DEP Coral Propagation Program 2021–2022, NFWF Massive Outplants 2021–2022, DEP CPR DLAB Outplants 2023, DEP CPR Phase VII (2025–2026), and previous MS student John Alfirevich's rapid site-selection study (2023–2025). These five studies outplanted 2,475 corals in total, representing a combination of 10 coral species across 20 sites, including *Siderastrea siderea* (n = 757, 4 studies), *Montastraea cavernosa* (n = 430, 4 studies), *Diploria labyrinthiformis* (n = 402, 4 studies), *Pseudodiploria clivosa* (n = 312, 4 studies), *Colpophyllia natans* (n = 171, 2 studies), *Solenastrea bournoni* (n = 226, 1 study), *Pseudodiploria strigosa* (n = 145, 3 studies), *Agaricia agaricites* (n = 58, 2 studies), *Porites astreoides* (n = 29, 1 study), and *Orbicella faveolata* (n = 20, 3 studies). The overall dataset spans from February 2021–2026.

Across 15 projects (including the following theses, DEP reports, and publications: Koval 2018, Koval et al. 2020, Rivas 2020, Rivas et al 2021, Lirman et al. 2022, Harrell 2022, Harrell and Lirman 2023, Rule 2023, Lirman et al. 2023, Weisman 2024, Sharp et al. 2024, Tobias 2024, Tenberg 2025), the UM dataset included 3,667 coral outplants representing 14 species across 26 reef sites. The most frequently outplanted species were *Orbicella faveolata* (n = 1,255, 10 studies), *Pseudodiploria clivosa* (n = 701, 10 studies), *Montastraea cavernosa* (n = 568, 5 studies), *Pseudodiploria strigosa* (n = 400, 7 studies), and *Diploria labyrinthiformis* (n = 354, 6 studies). Additional species included *Colpophyllia natans* (n = 129, 3 studies), *Porites astreoides* (n = 44, 1 study), *Orbicella annularis* (n = 33, 2 studies), *Porites porites* (n = 29, 2 studies), *Siderastrea radians* (n = 25, 1 study), *Dichocoenia stokesii* (n = 20, 1 study), and *Solenastrea bournoni* (n = 19, 1 study).

For all studies, raw monitoring data were processed in RStudio using the tidyverse package. Each study's associated CSV files were standardized and combined into a single Excel workbook using Python. Column names were standardized prior to concatenation to ensure consistency across studies. A unique coral identifier (Coral_ID) was constructed to keep track of individual outplants by concatenating site, array, and position-level fields (e.g., substrate position, row, column, or individual tag number), when that information was provided. Each coral's predation history was tracked cumulatively across monitoring events: once a coral had a note about predation ("PRD" = predation, "FB" = fish bites), its "Cumulative Predation" status was set to "Y" (Yes) for all subsequent dates. This allowed for tracking of when predation was initially recorded (based on survey date) and for tracking of overall predation prevalence by the end of the study across sites and coral species. Predation prevalence was calculated at the last monitoring date for each coral as the proportion of individual corals with Cumulative_Predation == "Y". Any corals that were undergoing a treatment (including the use of a predation exclusion device) were filtered out of the dataset to strictly quantify predation of outplanted corals without protection. Summary statistics and figures were generated in R using ggplot2, patchwork, sf, and ggrepel.

2.3. C-BRUVS Construction

2.3.1. Description of Camera Designs

Five new C-BRUVS were constructed in order to track corallivorous fish predation

over longer periods of time and to increase the number of possible deployments at each reef site. A C-BRUVS is defined as a camera system attached to a PVC frame with a Coral Lok tray (Reef Cells, LLC) of coral plugs attached in a randomized pattern (Figure 1). The new C-BRUVS design constructed herein improved upon the first iteration, which used an action camera with underwater housing (GoPro Hero 10), and a frame which attached the camera to the coral tray. Figure 1A–B demonstrate the structural differences between the first and second iterations, which are overall minor but required a redesign of the PVC frame in order to hold the new camera system. The new PVC frames (dimensions 0.5-m x 0.25-m) were constructed using 1/2" diameter Schedule 40 PVC (Spin), with one 0.25-m length of 1/2" diameter Schedule 40 PVC affixed vertically at one end. Dimensions of the PVC stand, camera housing, and camera holder are provided in Figure 2. PVC pieces were sealed toogether using PVC primer and cement, and a marine adhesive was used to affix the 3D-printed custom attachment piece for mounting the camera system to the PVC frame. Before entering the water, the PVC frames were weighed down using two 2-lb lead dive weights, which were zip-tied to the frames on either tray mount. The camera housing was also weighed using a 2-lb lead dive weight threaded through a hose clamp. Monofilament line was used to attach the camera housing to the PVC frame in the event that the camera mount was to break underwater.

2.3.2. Camera Specifications

The new camera system consisted of a computer (Raspberry Pi Zero 2 W) connected to a camera module (Raspberry Pi High Quality Camera), a realtime clock add-on board (Witty Pi 4 Mini) which schedules recording intervals, and a 20,000 mAh external battery pack (Voltaic V75). This system can automatically turn itself on, record video, and turn off at pre-determined times based on a schedule script which was set prior to each deployment. This allows the camera to record during daylight hours only, when diurnal parrotfish are active. The new camera system was able to record up to 40 hours of video during each deployment, roughly 3 days of daytime footage (14 hours per day).

This camera system required a custom-built underwater housing, which consisted of 1/2" thick acrylic faceplate, Easy-Pods 4" PVC Flange (Bellamare), 4" diameter Schedule 40 PVC pipe, a 4" diameter Schedule 40 PVC cap, 316 stainless steel hardware (socket head screws, washers, and nylon-insert nuts), and a custom-design 3D-printed mounting bracket. Figure 1 depicts the different parts of the new C-BRUVS camera and PVC frame which were designed to hold the more buoyant camera system.

2.4. Coral Propagation and Husbandry

2.4.1. Propagation for C63EA2

Colonies of the genus *Porites* were collected and propagated following previously established protocols (Forsman et al. 2015; Page et al. 2018). Colonies were fragmented to a size of approximately 3–5 cm diameter using a Griffon diamond bandsaw, and were given at least two weeks to recover after attachment to coral plugs using a pea-sized dollop of marine superglue for grow-out prior to deployment. Corals were kept in ambient conditions (flow-through system consisting of fiberglass raceways with a 350 Liter capacity and supplied with UV sterilized (>180 mJ/cm^2), 1µm filtered seawater from Biscayne Bay; variable-speed pumps for water flow (Maxspect Gyre XF30); LED lights (Ecotech Radion

XR30) for photosynthetically active radation ranging from 90–140 µmol/m²/s; water temperature maintained at 26.5ºC using a temperature controller (Inkbird) and 300W titanium heaters (Bulk Reef Supply)). Corals were fed (50–75 mg of Reef Chili dissolved in approximately 120 mL of seawater) twice per week to ensure recovery following collection and fragmentation stress.

The genus *Porites* was selected for propagation and use in coral predation trials due to the fast rates of fish predation that have been previously documented in Miami-Dade County at different reef sites, as well as its lack of designations as threatened, disease susceptible, or targeted for restoration efforts. Branching *Porites* has been described as the preferred prey of Caribbean parrotfishes (Burkepile 2012) and the species *P. divaricata* was reduced in its distribution following intense parrotfish predation (Miller and Hay 1998). In this way, the *Porites* spp. can be seen as a proxy predation species and were used to trial the C-BRUVS.

2.4.2. Propagation for F2305

In this report, we will also be reporting on Task 2 activities of DEP project F2305 (Expanding the University of Miami Reef Restoration Capacity). Activities for this project focused on *Diploria labyrinthiformis* spawning in April and May and remounting coral recruits from sexually propagated 2024 and 2025 cohorts. The Coral Reef Futures Laboratory's spawning aquaria were designed and established using information published in 2019 by The Florida Aquarium (TFA). Based on this guidance, the number of LED lights over each spawning tank were doubled so that photosynthetically active radiation (PAR) matched TFA conditions and adjusted the light spectrum accordingly. Laboratory larval care and settlement systems were also upgraded, scaling up previously described settlement infrastructure to accommodate the larger number of larvae anticipated from 2026 spawning efforts. Additionally, a second spawning laboratory at the Rosenstiel School has begun installment, expanding the University of Miami's coral spawning capacity. The new lab will include four recirculating aquarium systems operated with artificial seawater: two 700-gallon systems and two 250-gallon systems. These systems will function similarly to the existing spawning lab, using an integrated automated approach to simulate seasonal solar, lunar, and temperature variation throughout the year and promote synchronized coral spawning. The systems are expected to be completed, cycled, and ready for spawning events later this year.

The nursery team also prioritized remounting juvenile coral recruits from settlement tiles onto individual substrates. Once remounted, recruits were assigned unique genotype-specific IDs and added to the facility-wide inventory. Remounting indicates that recruits are stable enough to be handled and maintained individually. During this reporting period, 678 recruits of priority species were remounted: 45 2024 *Pseudodiploria strigosa*, 18 2024 *Montastraea cavernosa*, 375 2025 *Diploria labyrinthiformis*, 120 2025 *Colpophyllia natans*, and 120 2025 *Pseudodiploria strigosa*. Juveniles from the 2025 spawning efforts were more numerous and growing more quickly than those from 2024, suggesting continued improvements in reproductive protocols and post-settlement care that enhance coral recruit quality and survivorship.

2.5. C-BRUVS Deployments

2.5.1. Pre-Deployment Setup

Prior to deployment, each C-BRUVS was assembled as much as possible to ensure efficiency during SCUBA dives. The camera and housing were programmed and sealed the day before or morning of reef deployment, and were attached to the vertical PVC stand using two hose clamps and monofilament line. Dive weights were attached prior to deployment with zip-ties. The Coral Lok tray was attached to the PVC frame during deployment using zip-ties.

When *Porites* coral fragments were used, Reef Cells CoralLok trays were assembled either in the *ex-situ* laboratory or *in-situ* coral nursery underwater. During C-BRUVS field deployments, the Coral Lok trays were attached to the PVC frames of the C-BRUVS using zipties. The Coral Lok trays dimensions are 12” x 18” x 1/2”, and fit 69 CoralLok plugs, although trays were never filled to 100% capacity to ensure spacing between corals. Figure 5 depicts an example tray map data sheet and an annotated image of a coral tray which was used to track coral position to genotype and unique ID for tracing back during video analysis.

2.5.2. Field Deployment and Data Collection

The C-BRUVS were attached to reef substrate in an area on the reef without live benthic cover, using galvanized steel 2-cm long nails on each side of the PVC frame and 2-cm long monofilament fishing line at each anchoring point. The C-BRUVS were deployed at a given reef site for at least 72 hours, but were up to one week when weather conditions prevented an earlier collection. During the date of collection, corals were individually photographed and scored for predation (ocurrence, yes/no; intensity, % tissue removal). After scoring, corals were returned back to the *in-situ* coral nursery. Figure 4 provides image examples of predation scars observed during multiple deployments, and showcases the standardized camera scaler attachment used to take photos from a set distance and with a set diameter for post-processing analysis of live tissue area. Photographs of each coral fragment at deployment and retrieval were analyzed using ImageJ (Schneider et al. 2012) following the lab's standard protocol to measure live tissue area (LTA, cm^2) from top-down images of individual coral fragments. Net change in LTA (ΔLTA = Initial LTA – Final LTA) and percent change were calculated for each fragment. These data were joined to the predation records by coral position ID to examine the relationship between bite rates (captured during video) and overall tissue loss.

At the end of the deployment, C-BRUVS were collected and brought back to the laboratory for video download. In the laboratory, observers reviewed all video footage and recorded each fish predation event, logging: fish species (identified to species where possible), age/life phase (juvenile, initial, terminal; for wrasses and parrotfish), fork length (cm) when possible (some deployments contained a cropped video field of view that prevented this estimate), the number of bites in each event, and the timestamp of the bite (minutes:seconds into the video). When it was unclear whether a fish bit the coral, plug, or tray, the event was recorded to ensure it was captured. Each fish was assigned a unique ID within each video session. If a fish left the frame and re-entered after more than 30 seconds, it was recorded as a new individual. For each video, the total video duration per session

(minutes) was also recorded, enabling calculation of per-individual bite rates. Video session metadata (start and end times) were linked to the fish predation records to assign each bite event a time-of-day category: Morning (06:00–11:59), Afternoon (12:00–16:59), Evening (17:00–20:59), or Night (21:00–05:59). The automated video script led to daily capture from 07:00–21:00 (14 hours).

2.6. Predation Pressure Index

Synthesized data from the UM and NSU meta-analysis were analyzed to identify trends in predation prevalence across experimental studies and outplanting projects. The PPI was defined as the proportion of coral outplants at a given site and year that experienced at least one predation event (i.e., predation prevalence), expressed as a value between 0 and 1. This prevalence-based formulation was adopted as the primary index because tissue loss measurements were not collected consistently across all contributing studies.

To identify predictors of predation prevalence at the coral individual level, a suite of binomial logistic regression candidate models was constructed with predation occurrence (0 = absent, 1 = present) as the response variable. Predictor variables included coral species identity, region (Miami-Dade County vs. Broward County), and year (as a continuous variable). Five candidate models were compared using Akaike's Information Criterion (AIC): species only, species + region, region + year, species + region + year, and a full interaction model of species × region + year. Model fit was evaluated using ΔAIC, and odds ratios with 95% confidence intervals were extracted from the best-supported model. A random forest regression was additionally applied to the site-year-level PPI dataset to assess the relative importance of region, year, and coral species identity in explaining variation in prevalence-based PPI values, using percent increase in mean squared error (% IncMSE) upon variable permutation as the importance metric.

2.7. GIS Integration and Decision Support System

The 2018 – 2024 National Coral Reef Monitoring Program (NCRMP) reef fish visual censuses (RVC) data were summarized to assess parrotfish density, biomass, and length frequency between three regions along Florida's Coral Reef. Since the largest scrapers and excavators are known to hinder coral restoration, spatiotemporal patterns of the density and biomass of parrotfish in the above data were utilized, with a focus on the the KJCAP to examine historical predation pressure patterns and relationships in the restoration data. This data were analyzed using ArcGIS, and associated GIS layers were incorporated into Florida's Decision Support System, or Sea Sketch (https://www.seasketch.org/crdss/app).

2.7.1. Data Download and Procurement

The RVC package was downloaded into RStudio from https://github.com/jeremiaheb/rvc 2018 – 2024 data from the Southeast Florida (SEFCRI) and Florida Keys (FLA KEYS) regions were downloaded. Each RVC survey site was given a unique identifier called "SAMPLE_ID". Abundance values were calculated at each SAMPLE_ID for the following groups:

- Blue Parrotfish (*Scarus coeruleus*) individuals >15cm
- Midnight Parrotfish (*Scarus coelestinus*) individuals >15cm

- Rainbow Parrotfish (*Scarus guacamaia*) individuals >15cm
- Queen Parrotfish (*Scarus vetula*) individuals >15cm
- Princess Parrotfish (*Scarus taeniopterus*) individuals >15cm
- Stoplight Parrotfish (*Sparisoma viride*) individuals >15cm
- Redband Parrotfish (*Sparisoma aurofrenatum*) individuals >15cm
- Redtail Parrotfish (*Sparisoma chrysopterum*) individuals >15cm
- Yellowtail Parrotfish (*Sparisoma rubripinne*) individuals >15cm
- Corallivorous Parrotfish individuals >15cm (sum of all spp. listed above)

Data was exported as a CSV and imported into ArcGIS Pro. Each RVC survey was represented as a point and projected into the NAD 1983 (2011) UTM Zone 17N projected coordinate system. The extent of the data was clipped to only include RVC surveys in the Biscayne and Broward-Miami ecoregions. Additionally, 2021 RVC surveys were not conducted south of Government Cut in Miami due to the COVID-19 pandemic. For this reason, 2021 data have a smaller extent, ranging from Government Cut to the northern boundary of the Broward-Miami ecoregion. The following methods were conducted for all survey years despite this; however, the 2021 export maps are separated from the 2018, 2022, and 2024 comparison as the hot spots and interpolations may be misleading due to the variation of extent.

2.7.2. Hot Spot Analysis

RVC survey points were separated into different point layers by each individual year (2018, 2021, 2022, and 2024), as well as one point layer that kept all survey points combined (2018 – 2024). Using ArcGIS Pro's Optimized Hot Spot Analysis, an optimized version of Getis-Ord Gi* hot spot analysis was run for each year's respective point layer using the individual species of interest as the Analysis Field. The output was an individual optimized Getis-Ord Gi* hot spot data layer for each time and species constraint. These layers show hot and cold spots with the corresponding confidence level.

2.7.3. Interpolated Raster Analysis

To produce interpolated surfaces to underlay the hot spot analyses, the ArcGIS Pro's Empirical Bayesian Kriging tool was used with the following settings:

- Input features: RVC survey points for targeted year
- Z value field: targeted species field
- Output cell size: default (width of extent divided by 250)
- Data transformation type: None
- Semivariogram model type: Power

This produced an individual geospatial analysis layer showing the interpolated abundance of an individual species in the selected year. ArcGIS Pro's GA Layer to Rasters tool was used to convert each GA layer to a new raster. The output is a raster layer of predictions derived from the Empirical Bayesian Kriging geospatial analysis layer. A clipping polygon was created using the RVC survey points from each year and ArcGIS Pro's Minimum Bounding Geometry tool. The geometry type was Convex Hull, which creates the smallest polygon that encloses all points. Additional manual clipping was

conducted to remove parts of the intracoastal waterway that are unnecessary for the extent of the rasters. ArcGIS Pro's Extract by Mask tool was then used to clip each raster by the extent of the polygon, leaving each species and year combination with an interpolated prediction raster that covers the relevant extent of the Florida Reef Tract. To standardize the color scheme of these interpolated rasters and allow for visual year-to-year comparisons, the symbology of each raster for each species was changed to a custom stretch type with identical max and min values in all years. The max and min values were selected by the absolute maximum and minimum interpolated value for an individual species across all survey years, which can be found in Table 2.

3. RESULTS

3.1. Literature Review

Across the 33 studies included in the literature review (22 publications, 10 master's theses, and 1 report) spanning 1998–2025, publication output has increased markedly in recent years, with 18 studies (55%) published within the last five years (Figure 2; Table 3). Most studies were conducted either in the Florida Keys (16 studies) or Southeast Florida, including Miami-Dade and Broward Counties (17 studies). Methodologically, relatively few studies incorporated direct behavioral observation, with 10 of 33 (30%) using video. Additionally, 10 of 33 (30%) employed exclusion devices to prevent predation. Four studies used both video and exclusion devices. The types of exclusion devices varied, with the most common device used being a mesh cage (Dougan et al. 2020; Rivas 2020), as well as metal spikes, wood skewer "teepees" (Rule 2023), and biodegradable polyhydroxyalkanoate (PHA) tubes, or "Coral Castles" (Pisano 2023).

When determining whether predation occurred, most studies relied on assessments of grazing scars via fish bites (tissue removed and skeleton exposed/abraded). In terms of coral taxa, *Orbicella faveolata* was the most frequently studied species (18 studies), followed by *Montastraea cavernosa* (11) and *Porites* spp. combined (9, including *P. porites*, *P. astreoides*, and *P. divaricata*). Other commonly represented taxa included and *Pseudodiploria strigosa* (7), *Pseudodiploria clivosa* and *Diploria labyrinthiformis* (6 each), and *Acropora cervicornis* (5).

3.2. Meta-Analysis

3.2.1. Nova Southeastern University Outplanting Projects

Across the NSU studies, predation was substantial but differed greatly across projects, sites, and species. Across 5 studies, 20 sites, and 10 species, 2,475 corals were monitored with overall predation prevalence by the end of the study of 30.1%. At the study level (across sites and species), the highest predation prevalence was in Alfirevich 2023-2025 (50.8%). At the site level (across studies and species), JA12 had the highest predation prevalence (85.7%). At the species level (across sites and studies), *Colpophyllia natans* had the highest overall predation prevalence (76.0%).

Predation prevalence varied significantly by coral species. Relative to *Montastraea cavernosa* (pooled predation risk: 12.4%), *Colpophyllia natans* and *Siderastrea siderea* had the highest odds of predation (OR = 8.95, 95% CI: 5.33–15.22, $p < 0.001$; OR = 8.70, 95% CI: 6.40–12.00, $p < 0.001$, respectively), followed by *Pseudodiploria strigosa* (OR = 4.24, 95% CI: 2.62–6.86, $p < 0.001$) and *Pseudodiploria clivosa* (OR = 2.91, 95% CI: 2.02–

4.21, $p < 0.001$). *Diploria labyrinthiformis* had significantly lower odds of predation than *M. cavernosa* (OR = 0.63, 95% CI: 0.40–0.97, $p = 0.040$), as did *Solenastrea bournoni* (OR = 0.50, 95% CI: 0.27–0.88, $p = 0.021$). *Agaricia agaricites*, *Porites astreoides*, and *Orbicella faveolata* did not differ significantly from *M. cavernosa*.

Predation prevalence also varied substantially across reef sites. Relative to JA6 (predation risk: 45.2%), JA12 had the highest odds of predation (OR = 31.64, 95% CI: 12.69–91.91, $p < 0.001$), followed by JA11 (OR = 13.18, 95% CI: 6.02–30.65, $p < 0.001$), South Spawning Hub (OR = 11.60, 95% CI: 5.88–23.84, $p < 0.001$), and JA10 (OR = 10.55, 95% CI: 4.90–23.80, $p < 0.001$). Experimental 4 had significantly lower odds of predation than JA6 (OR = 0.064, 95% CI: 0.004–0.316, $p = 0.008$), representing the lowest predation pressure among all sites in the dataset. All other sites in the top ten showed significantly elevated odds relative to JA6 ($p < 0.001$). Because site and outplant year were fully confounded across the dataset, with only one site (Core 2) sampled in more than one year, the relative contributions of reef location versus temporal variation to predation pressure could not be formally partitioned, and patterns observed across sites reflect a combination of site-specific and study-specific factors.

Across all studies, cumulative predation prevalence increased rapidly within the first monitoring interval at several sites. All five NSU studies included monitoring visits within the first week post-outplanting. Three studies monitored at two weeks post-outplanting, while only one study captured predation within the first 1–3 days. 80% conducted monitoring at 1–2 months post-outplanting. All five studies included a three-month monitoring visit, representing the most consistently captured later timepoint across the dataset. The most striking early predation was observed at South Spawning Hub (Feb 2025 cohort), where *Pseudodiploria strigosa* and *Colpophyllia natans* reached 80.0% and 69.2% cumulative predation prevalence, respectively, within just 8 days of outplanting. Similarly, *Montastraea cavernosa* at North Spawning Hub (Jan 2025 cohort) reached 50.0% prevalence within 7 days, and *Colpophyllia natans* at South Spawning Hub (Nov 2025 cohort) reached 50.0% within 10 days. At Core 2 (Apr 2021), *Pseudodiploria strigosa* reached 100% cumulative predation prevalence and *Pseudodiploria clivosa* reached 50.0% within 30 days of outplanting, and *Siderastrea siderea* at Experimental 5 reached 74.1% within 14 days. At sites where rapid early predation was not observed, prevalence either increased gradually across monitoring intervals or remained low throughout the study period. These patterns suggest that the window of highest predation risk occurs immediately following outplanting, which agrees with existing literature.

3.2.2. University of Miami Outplanting Projects

Across 15 projects, cumulative predation prevalence varied substantially by coral species, reef site, and study. Overall, 2,367 of 3,667 monitored outplants (64.5%) experienced predation across the study period. Predation prevalence was highest for *Porites astreoides* (95.5%; n = 44, 1 study), *Solenastrea bournoni* (94.7%; n = 19, 1 study), *Diploria labyrinthiformis* (87.3%; n = 354, 6 studies), *Porites porites* (86.2%; n = 29, 2 studies), and *Colpophyllia natans* (86.0%; n = 129, 3 studies). Moderate predation prevalence was observed for *Dichocoenia stokesii* (75.0%; n = 20), *Orbicella annularis*

(69.7%; n = 33), *Pseudodiploria clivosa* (64.9%; n = 701), *Pseudodiploria strigosa* (62.2%; n = 400), *Orbicella faveolata* (61.8%; n = 1,255), *Montastraea cavernosa* (61.4%; n = 568), and *Siderastrea radians* (60.0%; n = 25).

A binomial logistic regression restricted to species with at least 50 monitored individuals across at least two studies identified significant differences in predation odds relative to *Orbicella faveolata*. *Diploria labyrinthiformis* (OR = 4.25, 95% CI: 3.08–6.00, $p < 0.001$) and *Colpophyllia natans* (OR = 3.82, 95% CI: 2.35–6.57, $p < 0.001$) had significantly higher odds of predation than the reference species. *Montastraea cavernosa*, *Pseudodiploria clivosa*, and *Pseudodiploria strigosa* did not differ significantly from *Orbicella faveolata* (all $p > 0.16$).

Predation pressure varied significantly across reef sites in the Miami-Dade region. Relative to Key Biscayne (KB NOAA), which exhibited the lowest observed predation prevalence among Miami-Dade sites (8.8%), all other sites with sufficient sample sizes showed substantially elevated odds of predation. Yung's Reef had the highest odds of predation (OR = 477.89, 95% CI: 95.66–8,716.02, $p < 0.001$), followed by Rainbow Reef (OR = 29.09, 95% CI: 12.61–72.50, $p < 0.001$), Paradise Reef (OR = 26.37, 95% CI: 16.48–44.86, $p < 0.001$), and Flamingo Reef (OR = 14.80, 95% CI: 9.20–25.27, $p < 0.001$). Miami Beach (MB NOAA) did not differ significantly from Key Biscayne (OR = 0.247, 95% CI: 0.014–1.251, $p = 0.180$), consistent with both sites showing low overall predation prevalence. The extremely wide confidence interval for Yung's Reef reflects a small sample size at that site and should be interpreted with caution. BNP was excluded from the model due to complete separation (100% predation prevalence).

The majority of UM studies (86.7%) included at least one monitoring visit within the first 4–5 weeks post-outplanting, and 73.3% conducted monitoring within the first week. Half of all studies (53.3%) monitored corals at 1–2 months post-outplanting. Fewer studies captured very early post-outplanting predation, with only 26.7% conducting within 1–3 day monitoring visits. Across UM outplanting projects with early monitoring data, predation prevalence was consistently high within the first weeks post-outplanting. Within the first 1–3 days, 42.4% of monitored corals (n = 1,231, 4 studies) showed evidence of predation, and 40.5% within the first week (n = 1,952, 11 studies). Predation prevalence remained elevated through the first month, ranging from 33.2% to 41.8% across studies monitoring at weeks 2–5 (n = 734–2,651 corals per bin). These patterns are consistent with the broader literature documenting the first week post-outplanting as the period of highest predation risk for massive coral species on Florida's Coral Reef (Koval et al. 2020; Smith et al. 2021; Little 2024). Because different studies and sites contributed to each monitoring timepoint bin, prevalence values at later timepoints are not directly comparable and are not reported here as a temporal trend.

3.3. C-BRUVS Construction

The construction of five new C-BRUVS was successfully completed by April 2026. The project required extended time for constructing and testing the camera housings to ensure there was no flooding at depth. The design has been tested to 25 feet. Additionally, optimizing the design of the PVC frame was crucial to capturing on video the entire coral

tray while also identifying the species of fish in view. One limitation of this camera design compared to a conventional underwater video camera like a GoPro is the need for a computer and a virtual network computing (VNC) client to access the live view. This means the camera's angle and view cannot be adjusted during a SCUBA dive and must be set and secured in advance. Furthermore, water refraction causes the camera's field of view to shrink compared to when tested on land. To address this, we conducted a pool test to determine the optimal distance and height of the camera. However, we decided to shorten the height to improve the camera's weight distribution and balance with the coral tray during field deployment, especially in the event of strong underwater surges (Figure 1D).

3.4. Coral Propagation and Husbandry

A total of 771 fragments were created for this project in the *ex-situ* laboratory through the micro-fragmentation approach or were received as coral juveniles from an *ex-situ* spawning event. This includes 502 fragments/juveniles of *Porites astreoides*, with 13 unique genotypes as micro-fragments and 89 unique genotypes from the coral juveniles. One genotype of *P. divaricata* was fragmented into 127 fragments, and 5 genotypes of *P. porites* were fragmented into 142 total fragments.

3.5. C-BRUVS Deployments

3.5.1. Number of Deployments Conducted During Project

Several different deployments were conducted during this project in an effort to evaluate the use of real-time video and the use of the coral genus, *Porites*, as a proxy species to understand corallivorous fish predation without sacrificing microfragments of target reef-building coral species used for restoration efforts (i.e., *Montastraea cavernosa, Colpophyllia natans, Orbicella faveolata*). The majority of deployments included the use of *Porites*, but one field outplanting project was also utilized in this study (2026-03-24, Paradise Reef), as it coincided with the development of the new C-BRUVS and involved the use of target reef-building coral species often predated upon by corallivorous fish. Additionally, the first deployment for this project utilized the Coral City Camera (CCC), an underwater camera system with 24-hour real-time footage continuously captured and uploaded for the public in order to evaluate both the efficacy of the CCC for predation trials and to characterize the coral predation pressures in the unique environment of the Port of Miami. Table 1 summarizes all of the deployments conducted during this project, with image examples in Figure 3 and a map of all deployments in Figure 12.

3.5.2. Video Analysis Completed

Video analysis was completed for three deployments which had recorded predation events on the corals during the post-deployment coral survey; this included CCC (2025-10-23), Paradise (2026-03-24), and Oasis (2026-05-01). The CCC deployment had 6 videos for a total of 12 hours of footage, with three hours captured in the first day in the afternoon (14:25–17:25), then three hours two days later in the morning (7:00–12:55). The Paradise deployment included 73 videos and 39 hours of footage. On the day of deployment, footage ranged from 11:26–20:20, and days one and two, the range was 7:00–20:00. On the last day, before the C-BRUVS camera battery died, footage was collected from 7:00–11:07. Lastly, for the Oasis deployment, on the day of deployment, footage

started at 9:13 and ran until 19:13. On days one and two, footage was recorded between 7:00–20:20. The last day, only one video was recorded for 13 minutes.

Notably, for the Oasis deployment, we tracked the amount of time the analysis took to watch 46 videos, and this required 6 staff and 7 students to watch 37 hours of footage. Anecdotally, video analysis for 1 one-hour video ranged from 30 minutes to 4 hours, depending on the time of day of video capture and amount of fish activity. Thus, this project would greatly benefit from the use of machine learning or artificial intelligence video analysis tools, several of which already exist for BRUVS in fisheries contexts (Connolly et al. 2021; Marrable et al. 2022; Baletaud et al. 2025).

3.5.3. Composition of Coral Predators and Periods of Activity

3.5.3.1. All Deployments Combined

Predation was dominated by parrotfishes across all three deployments, and species composition differed notably by deployment. At CCC, stoplight parrotfish overwhelmingly dominated, accounting for 808 of 1,029 bites (78.5%). At Oasis, stoplight parrotfish remained the top predator (963 bites; 48.4%) but was followed more closely by redband parrotfish (407 bites; 20.5%), foureye butterflyfish (190 bites; 9.5%), and striped parrotfish (181 bites; 9.1%). The Paradise deployment showed a strikingly different pattern, where foureye butterflyfish was the dominant fish (334 bites; 62.2%), with stoplight parrotfish second (154 bites; 28.7%) (Figure 13).

Predation was not uniformly distributed across deployment days; however, this was likely due to a difference in video footage captured between CCC (where a C-BRUVS was not used) versus Paradise and Oasis. At CCC, 92.7% of bites (954 of 1,029) occurred on the first day of deployment (when the majority of video was collected), with far fewer recorded on days 2–4. At Paradise, bites peaked on day 2 (294 bites on 2026-03-25) relative to 228 on day 1 and only 15 on day 3. At Oasis, predation was highest on day 3 (570 bites), moderate on day 1 (286) and day 2 (122), and negligible by day 4 (2 bites – although footage on day 4 stopped after 12 minutes).

3.5.3.2. CCC Deployment

At CCC, bite activity began within the first minutes of each 60-minute video session and was dominated by stoplight parrotfish throughout, with slippery dick wrasses and striped parrotfish contributing minimally in comparison. As *P. astreoides* was the only coral species used in this deployment, across fragments, 1,029 total number of bites were recorded.

3.5.3.3. Paradise Deployment

Nine corals were tracked during this C-BRUVS deployment, three per species. These nine corals were originally reared as sexual recruits from The Florida Aquarium (TFA). During this deployment, *P. clivosa* were highly targeted, experiencing 91.4% of predation events (equating to 491 total bites) compared to 6.5% for *D. labyrinthiformis* and 2% for *P. strigosa.*

3.5.3.4. Oasis Deployment

A total of 48 coral fragments were tracked across the Oasis deployment (May 1–4, 2026), comprising 33 *P. astreoides* and 15 *P. porites* fragments. ImageJ-based live tissue area (LTA) measurements revealed that 38 of 48 fragments (79.2%) experienced an overall net tissue loss over the 4-day deployment, with a mean ΔLTA of -0.948 cm² (standard deviation = 1.607; Figure 15). Across all videos, parrotfish dominated corallivory, accounting for 89.6% of all predator bites. 47 of 48 fragments had at least one recorded bite event. In comparing predation events of *P. astreoides* and *P. porites* from parrotfish and butterflyfish, *P. astreoides* experienced 53.3% of predation events (979 bites) compared to *P. porites* (46.6%, 856 bites).

The correlation between total number of fish bites and ΔLTA was statistically significant and negative (Pearson r = -0.606, p = < 0.001; Spearman ρ = -0.561, p = < 0.001), indicating that fragments receiving more fish bites tended to show greater tissue loss (Figure 16). When the two greatest fish taxa were examined separately, parrotfish bites were significantly negatively correlated with ΔLTA (Pearson r = -0.592, p = < 0.001), while butterflyfish bites showed a weaker relationship (Pearson r = -0.275, p = 0.0582) (Figure 17). Because parrotfish and butterflyfish were often recorded biting the same corals, these separate correlations cannot be directly compared (although an assumption can be made that the grazing scars on the coral fragments would be attributed to parrotfish rather than butterflyfish, based on previous literature). To formally test whether each fish taxa had an independent effect after accounting for the other, a multiple linear regression was fit with both predictors simultaneously. The model explained 0.325 of variance in ΔLTA (adjusted R^2 = 0.34; $F(2, 45)$ = 13.13, p < 0.001). Parrotfish bites remained a significant negative predictor (β = -0.0360, p < 0.001), while butterflyfish bites were not significant (β = -0.0486, p = 0.2623), consistent with their role as polyp-feeders rather than tissue-removing scrapers.

In terms of fish species exhibiting greatest pressures, stoplight parrotfish across the Oasis deployment contributed 963 total bites of 44 corals, followed by redband parrotfish (387 bites, 46 corals), striped parrotfish (174 bites, 34 corals), and princess parrotfish (98 bites, 23 corals). However, as 83% of corals were bit by multiple parrotfish species, the greatest contributors to total tissue removal across fragments could not be determined.

Species-level differences in corals experiencing predation were apparent (with the caveat that the sample sizes were skewed towards *P. astreoides* fragments). *P. porites* fragments experienced substantially greater mean tissue loss (mean ΔLTA = -2.178 cm²) compared to *P. astreoides* (mean ΔLTA = -0.388 cm²), despite comparable video-recorded bite counts. This suggests that *P. porites* may be more susceptible to tissue damage per bite event, or that parrotfish remove a larger portion of tissue when predating on this species. The differing morphologies of the two species was apparent for several fragments on the coral tray deployed at Oasis. A majority of the *P. porites* fragments had at least one branch with greater vertical extension compared to the mounding *P. astreoides* fragments. This may also explain the differing predation pressures documented.

3.6. Predation Pressure Index

The PPI ranged from 0.02 to 1.00 across 56 site-year combinations spanning 33 reef sites in Miami-Dade County (n = 34 site-years, UM dataset) and Broward County (n = 22 site-years, NSU dataset), with data collected between 2018 and 2025 (Figure 10). Miami-Dade County sites exhibited consistently higher PPI values (median = 0.849) than Broward County sites (median = 0.214), and this regional difference was the strongest predictor of predation prevalence across all analyses.

Model comparison identified the full interaction model (Species × Region + Year) as the best-supported candidate model (AIC = 824.79). The additive model including species, region, and year as main effects (AIC = 827.61, ΔAIC = 2.82) was also competitive, while models excluding year (ΔAIC = 25.59) or including species alone (ΔAIC = 48.51) were substantially less supported, confirming the independent contributions of region, year, and species identity to predation prevalence.

Odds ratios extracted from the best model indicated that outplants in Miami-Dade County had 167-fold higher odds of predation occurrence compared to those in Broward County (OR = 167.14, 95% CI: 16.15–1730.23, $p < 0.001$). Year was a significant positive predictor of predation prevalence, with each additional year associated with a 55% increase in the odds of predation occurrence (OR = 1.55, 95% CI: 1.26–1.92, $p < 0.001$). Among coral species, *Colpophyllia natans* (OR = 13.62, 95% CI: 1.55–119.96, $p = 0.019$) and *Pseudodiploria clivosa* (OR = 9.56, 95% CI: 1.08–84.73, $p = 0.043$) exhibited significantly elevated odds of predation relative to the reference species *Orbicella faveolata* in Broward County. No other species reached statistical significance as independent predictors, and none of the species × region interaction terms were significant, though *Pseudodiploria clivosa* in Miami-Dade County showed a marginally non-significant negative interaction (OR = 0.11, 95% CI: 0.01–1.05, $p = 0.056$), suggesting that the elevated susceptibility of this species relative to *O. faveolata* may be attenuated at Miami-Dade sites where overall predation prevalence is already high across species.

Random forest analysis of site-year-level PPI values confirmed region as the most important predictor variable (% IncMSE = 55.2%), followed by year (% IncMSE = 32.6%) and coral species identity (% IncMSE = 19.5%). The relative importance rankings were consistent with the logistic regression results and with the visual pattern in the species-by-region prevalence comparisons (Figure 11), in which Miami-Dade County prevalence exceeded Broward County prevalence for all eight species represented in both datasets.

Paradise Reef (Miami-Dade County) had PPI values of 0.879, 1.00, 0.742, and 1.00 in 2022, 2023, 2024, and 2025 respectively, confirming it as a persistently high-predation site across years and species. The lowest PPI values in Miami-Dade County were observed at Miami Beach (PPI = 0.023 in 2018) and Key Biscayne (PPI = 0.088 in 2018), though these observations are early in the time series and the significant positive year trend cautions against assuming these sites remain low-risk. In Broward County, Experimental 4 (PPI = 0.016 in 2021), Experimental 6 (PPI = 0.061 in 2021), and Core 2 (PPI = 0.032 in 2022) represented the lowest-predation sites across the full dataset.

3.7. GIS Integration and Decision Support System

3.7.1. Spatial Trends

Overall trends from the cumulative survey period indicated that the hot spots for all but two species are found in the southernmost area of Biscayne National Park. The exceptions were the redband and princess parrotfish, with these hot spots being more prominent in the northern stretches of the Broward-Miami ecoregion. Species such as the blue parrotfish and queen parrotfish showed hot spots corresponding with deeper reef habitats. Both stoplight and princess parrotfish hot spots appeared to shift gradually northward over time.

3.7.2. Output Maps

The combined data show optimized hot spot analyses overlaid on interpolated rasters generated from all RVC survey points collected between 2018 and 2024 for the Broward-Miami ecoregion to the southern boundary of Biscayne National Park (918A). The majority of hot spots occurred near Key Largo in southern Biscayne National Park. Species-specific analyses revealed clear spatial differences. Stoplight parrotfish largely drive the hot spot patterns in the combined data (Figure 19D), whereas redband (Figure 19B) and princess parrotfish (Figure 19C) show more localized clustering at reef sites in the Broward-Miami ecoregion near Miami Beach and Fort Lauderdale.

The temporal maps present optimized hot spot analyses overlaid on interpolated rasters for individual survey years displayed side by side. Consistent and unique patterns were present across survey years for all corallivorous parrotfish larger than 15 cm (Figure 20). Parrotfish density and hotspots were consistently high in southern Biscayne National Park. In 2018, there was a notable hotspot cluster near Fowey Rocks, but that cluster was not evident in 2022 or 2024. Hotspots varied in the Broward-Miami ecoregion with one around Fort Lauderdale in 2018, none in 2022, and two in 2024: Pompano and Hollywood. In 2018, redband parrotfish hot spots occurred in the south and north, but only in the north thereafter, with the highest abundances occurring in Broward by 2024 (Figure 21). Stoplight parrotfish had the most hotspots and abundances in Biscayne National Park throughout. However, a concentrated hot spot also emerged off North Miami Beach in 2024 (Figure 22).

4. DISCUSSION AND MANAGEMENT RECOMMENDATIONS

4.1. Summary of Current Understanding in the Field

4.1.1. Predation Rates and Temporal Patterns

Across studies conducted on FCR, fish predation on coral outplants is consistently documented as high and temporally concentrated in the period immediately following outplanting. Koval et al. (2020) reported that 73% of massive coral outplants experienced predation initially in Miami-Dade County, while Page et al. (2018) found that 40% of micro-fragmented outplants experienced predation at Florida Keys sites. Little (2024) similarly reported that the majority of predation occurred within the first week post-outplanting for juveniles of six massive coral species, and Alfirevich (2023) found that predation was highest one-week post-outplanting across 12 sites in Broward County, with most sites showing declining predation intensity by one month. Wagner (2024) tracked large *O. faveolata* colonies across Broward that were cored for sampling collections and found that over 90% of all predation events occurred within the first month post-sampling.

Smith et al. (2021) in the Florida Keys found that, while predation was highest in the first week, overall survival remained at 96%, suggesting that while early predation is intense, it does not necessarily translate to long-term mortality, particularly for larger fragments. Taken together, these studies indicate that the first week following outplanting represents a critical vulnerability window, and that survival outcomes depend heavily on the size and condition of outplants at the time of deployment.

Spatial variation in predation intensity has also been documented. Ladd et al. (2024) tracked bite occurrence and complete fragment removal of *D. labyrinthiformis* across Broward, Miami-Dade, and Monroe Counties, finding the highest predation intensity in Miami-Dade. Similarly, a two-year outplanting and monitoring study across a majority of the FCR using three species (*O. faveolata, M. cavernosa, P. clivosa)* found Miami-Dade and Broward reefs had the highest incidences of predation (Sharp et al. 2024). Alfirevich (2023) found that predation was present at all sites surveyed in Broward County, though one site had a notably higher predation prevalence than others (>75%), consistent with the broader finding that predation pressure is spatially heterogeneous.

4.1.2. Corallivorous Fish Taxa

Parrotfishes are the dominant corallivores documented across Florida studies. Burkepile (2012) identified stoplight parrotfish and redband parrotfish as the primary coral predators in the Florida Keys, and Miller and Hay (1998) confirmed these same two species as responsible for coral predation via video, with over half of uncaged transplants completely consumed within 48 hours. Burkepile et al. (2019) used 94 hours of video in the Florida Keys to characterize species-specific predation, finding that corallivory is concentrated among a subset of parrotfish species. However, the authors found that corallivory only represented <2% of the total number of bites recorded on video. The most frequent coral predators were rainbow parrotfish, princess parrotfish, and stoplight parrotfish. Rainbow parrotfish were the most likely to feed on coral. Parrotfish genera demonstrated prey preferences which seem to be distinguished by skeletal types, where *Sparisoma* species primarily targeted *C. natans* and *O. faveolata*, while *Scarus* species were more likely to target *Porites* or *Siderastrea.* Shantz and Ladd (2024) replicated the Burkepile (2012) survey methods in the Upper Florida Keys between 2010 and 2022 and excluded parrotfish smaller than 15 cm total length, consistent with evidence that small parrotfishes rarely feed on corals or leave bite scars (Bonaldo and Rotjan 2018; Burkepile et al. 2019). Zaneveld et al. (2016) also recorded parrotfish grazing scars and invertebrate corallivores on multiple coral species in a three-year exclusion experiment in the Florida Keys. Shantz et al. (2020) showed that large parrotfishes (>20 cm in total length) caused significant damage to mounding coral species through excavation, while the exclusion of large parrotfishes unexpectedly impaired mounding coral growth through algal competition. Shantz and Ladd (2024) also noted that redtail and blue parrotfish do not feed on corals, further narrowing the relevant corallivorous taxa. Fitt (2025) hypothesized that the population of rainbow parrotfish in an area of Florida Bay may be contributing to seasonal physiological stress of *Siderastrea radians* based on grazing scars observed, suggesting this species may also contribute to predation impacts under certain conditions. Beyond parrotfishes, Koval et al. (2020) documented butterflyfish and damselfish as

additional contributors to predation on massive coral outplants, with butterflyfish in particular recorded biting multiple coral species.

4.1.3. Coral Species Preferences and Susceptibility

A consistent pattern across Florida studies is that *Porites* spp. experience disproportionately high predation relative to other coral taxa. Burkepile (2012) found that *P. porites* and *P. astreoides* were the most frequently predated species in the Florida Keys, and that predation intensity on these species increased significantly as overall coral cover declined. Miller and Hay (1998) documented *P. divaricata* as the most heavily targeted species in comparative video assays, with *P. porites* also present in the area but less targeted than *P. divaricata*. Alfirevich (2023) found that wild colonies during rapid assessment surveys of *P. porites* and *P. astreoides* had the highest prevalence of predation scars across Broward County sites, while *Siderastrea siderea* had moderate predation. Zaneveld et al. (2016) found that in nutrient-enriched conditions, *Porites* survival plummeted, with 92% losing tissue through predation, leading to 62% mortality, highlighting an interaction between local stressors and predation susceptibility.

Among restoration target coral species, *Pseudodiploria* spp. emerge as particularly vulnerable. Koval et al. (2020) reported that *Pseudodiploria* species had the highest prevalence of fish bites among the four massive coral species studied in Miami-Dade County. Little (2024) found 100% predation occurrence on *Pseudodiploria clivosa* and *Orbicella faveolata* juvenile outplants in Broward County, compared to 40% on *Montastraea cavernosa*, though predation on *M. cavernosa* was also documented across multiple other studies (Page et al. 2018; Alfirevich 2023). For their outplanted corals, Alfirevich (2023) documented *S. siderea* with the highest predation prevalence, followed by *P. clivosa* and *M. cavernosa*. Lirman et al. (2022) documented parrotfish targeting large, relocated colonies of *P. clivosa*, *P. strigosa*, and *D. labyrinthiformis*, causing up to 100% tissue loss in some cases. Rempel et al. (2024), in a multi-region study that included Florida sites, found that corallivory was concentrated across regions on *Orbicella* spp., *Porites* spp., and *Stephanocoenia intersepta*. Importantly, not all massive coral species are targeted uniformly. Lustic et al. (2020) found no significant partial mortality on medium-sized colonies (40–130 cm²) of *O. faveolata* and *M. cavernosa* outplanted in the Florida Keys, indicating a potential size threshold below which predation risk is substantially reduced for these species.

4.1.4. Coral Size, Genotype, and Conditioning

Colony size at the time of outplanting is a significant predictor of predation risk and outcome. Rivas et al. (2021) found that larger colonies (25 cm²) showed higher resistance to predation than smaller fragments and that certain genotypes with lower lipid and protein content experienced greater predation mortality among *O. faveolata* outplants in Miami-Dade County. Little (2024) found that, while smaller corals were less likely to experience predation events, if predated upon, they had lower survival, suggesting that fragment size mediates both exposure and vulnerability to predation. Ladd et al. (2024) found that *ex situ* conditioning before outplanting reduced the probability of complete fragment removal by predators compared to *in situ* conditioning (likely due to the increased coral growth rates

ex situ leading to larger coral fragments in comparison), implying that nursery history influences post-outplant susceptibility.

4.1.5. Predation Deterrent Strategies

Physical exclusion cages have been the most consistently effective deterrent documented in Florida studies, but their benefits appear to be temporary. Rivas et al. (2021) found that cages provided 100% protection from predation while in place, but that benefits were entirely lost upon cage removal. Brownlee (2010) documented that partial caging of *S. siderea* reduced but did not eliminate predation and reported 94% predation on uncaged small transplants via grazing scars within one week. Evans et al. (2013) used *Porites astreoides* recruits in the Florida Keys but did not detect survivorship differences between caged and uncaged treatments after one month. Pisano (2023) evaluated biodegradable polyhydroxyalkanoate (PHA) tube barriers ("Coral Castles") combined with concrete bases and found that these prototype structures significantly reduced bite incidence and dislodgement for both *P. astreoides* and *O. faveolata*, improving survivorship while offering a degradable alternative to permanent caging. Harrell and Lirman (2023) tested a chemical deterrent approach, finding that pre-outplanting feeding of powdered *Dictyota* algae to *O. faveolata* conferred some unpalatability and reduced predation mortality, though this method was limited to corals held in *ex situ* nurseries. Co-outplanting *Acropora cervicornis* with massive coral species has been explored as a deterrent strategy (Rivas et al. 2021), though outcomes were variable.

4.1.6. Ecological Context and Implications for Restoration

Broader ecological context shapes corallivore predation pressure which is relevant for site selection and restoration planning. Burkepile (2012) found that parrotfish predation intensity on corals increased significantly as overall coral cover declined, suggesting that degraded reefs may expose outplants to disproportionately high predation. Rempel et al. (2024) found that, across multiple Atlantic sites including Florida, parrotfish density and biomass were positively correlated with relative coral area preyed upon, and that corallivory was concentrated on reefs with higher coral diversity and proportional cover of frequently targeted taxa. Dougan et al. (2020) found a trend toward greater and larger parrotfish wounds on nutrient-enriched *P. porites*, suggesting that water quality stress compounds predation risk. Zaneveld et al. (2016) similarly demonstrated that nutrient pollution interacting with corallivory led to severe *Porites* mortality, emphasizing that predation does not operate in isolation from other stressors on FCR. Lindholm et al. (2006) documented strong site fidelity and limited movement of blue parrotfish and princess parrotfish at Conch Reef in the northern Florida Keys, with most movement from core use areas lasting less than 30 minutes. This supports the notion that resident parrotfish communities can impose sustained and predictable predation pressure at specific reef locations rather than transient exposure, however, further studies of corallivorous parrotfish species should be conducted.

4.2. Biscayne-Miami Predation Prevalence Patterns

The combined UM and NSU datasets provide the most comprehensive characterization of coral outplant predation in the Biscayne-Miami region to date, spanning nearly a decade

of monitoring across 61 reef sites and representing over 6,200 individual coral outplants from 16 coral species. Despite differences in study design, coral species, outplanting methods, and monitoring duration across projects, several consistent patterns emerge from this synthesis.

Predation prevalence was broadly high across both datasets, with UM studies recording an overall cumulative prevalence of 64.5% and NSU studies recording 30.1%. This discrepancy likely reflects differences in the composition of coral species outplanted, as UM studies purposefully included a higher proportion of species with documented high predation susceptibility, such as *Diploria labyrinthiformis*, *Colpophyllia natans*, and *Porites* spp. Additionally, there were inherent differences in monitoring duration and reef site characteristics across the two institutions. Despite this, both datasets are consistent with the broader literature documenting high and temporally concentrated predation on massive coral outplants in Florida (Koval et al. 2020; Smith et al. 2021; Ladd et al. 2024).

Across both datasets, species identity was a significant predictor of predation risk. *Colpophyllia natans* emerged as a high-risk species in both the NSU (OR = 8.95 relative to *M. cavernosa*) and UM (OR = 3.82 relative to *O. faveolata*) analyses. *Diploria labyrinthiformis* similarly showed elevated predation risk in the UM dataset (OR = 4.25), and *Siderastrea siderea* was among the highest-prevalence species in the NSU dataset (56.8%). In contrast, *Montastraea cavernosa* showed relatively lower predation odds in both datasets. Practitioners outplanting high-susceptibility species like *C. natans* and *D. labyrinthiformis* should anticipate elevated predation risk and prioritize protective measures that minimize exposure to large corallivorous parrotfish during the critical post-outplant period.

Geographic patterns in predation risk were also consistent with the spatial hot spot analyses of NCRMP reef fish survey data. Among NSU sites in Broward County, the highest predation odds were broadly consistent with sites with the highest parrotfish densities documented in the RVC-derived hot spot analyses. Among UM Miami-Dade sites, Yung's Reef, Rainbow Reef, and Paradise Reef showed markedly higher predation odds than Key Biscayne and Miami Beach, however, the NCRMP data demonstrated a critical gap in this region where there were less surveys conducted in the area across years. Additionally, future work should aim to expand this framework into the Florida Keys, where the longer history of parrotfish research and larger existing datasets would substantially strengthen the predictive power of predation risk across the full extent of Florida's Coral Reef.

4.3. Predation Pressure Index and Management Implications

The prevalence-based PPI developed here captures the most consistently recorded dimension of predation impact across the meta-analysis dataset and provides an interpretable metric for restoration site planning. The finding that region explains more variance in PPI than either coral species identity or year, which was determined by both logistic regression and random forest analyses, has an important management implication that the existing literature does not fully emphasize. Predation risk in these regions is

primarily driven by location rather than species, although more studies where the same coral species are outplanted at the same time and monitored across regions are imperative to further support this finding. Miami-Dade County sites showed consistently high predation prevalence across all eight coral species represented in the dataset, ranging from 62% (*O. faveolata*) to 95% (*Solenastrea bournoni* and *Porites astreoides*). This means that species-specific susceptibility differences, while statistically detectable, are of limited practical utility for site selection within that region, as nearly all species at nearly all Miami-Dade sites experienced high predation prevalence regardless of species identity.

The significant positive temporal trend (OR = 1.55 per year, $p < 0.001$) is consistent with documented increases in corallivorous parrotfish abundance and body size in southeast Florida over the study period (Graff 2024) and with the broader pattern of tropicalization driving range expansions of large-bodied parrotfishes northward along Florida's Coral Reef (Walker et al. 2024). Critically, this trend means that PPI values calculated from earlier years in the dataset (particularly 2018–2019) likely underestimate current predation pressure at the same sites. Restoration practitioners should treat historical PPI values as conservative lower bounds.

Among coral species, the significantly elevated susceptibility of *Colpophyllia natans* (OR = 13.6 vs. *O. faveolata*) and *Pseudodiploria clivosa* (OR = 9.6 vs. *O. faveolata*) warrants attention for outplanting programs prioritizing these species. However, the wide confidence intervals on these odds ratios (reflecting limited within-species sample sizes at Broward County sites) indicate that species-specific recommendations are not definitive at this time. Tissue loss data removed by predation collected consistently across future deployments would substantially improve the precision of species-level risk estimates and is identified as a priority for subsequent monitoring efforts.

A key limitation of the prevalence-based PPI is that it quantifies observed predation outcomes on coral outplants rather than predicted predation pressure from the fish community. Formal integration of fish abundance and size-class data with coral predation occurrence data was not achievable for the majority of studies in the meta-analysis dataset, because paired fish survey data were not collected concurrently with predation monitoring at most sites. This limitation reflects a broader gap in the Florida corallivory literature, which is that coral predation studies and fish community surveys have generally been conducted separately, using different spatial extents, sampling intervals, and levels of taxonomic resolution. As a result, linking these datasets quantitatively remains methodologically challenging without a larger, congruent dataset.

In the absence of sufficient paired data to derive a fish-informed PPI, the optimized hot spot analysis of large parrotfish density and biomass from the 2012–2024 NCRMP dataset serves as the best currently available fish-based predictor of spatial predation risk across Florida's Coral Reef. This approach is supported from prior work demonstrating that large-bodied scrapers and excavators (i.e., stoplight and redband parrotfish) are the primary drivers of coral tissue removal in the Caribbean and in southeast Florida specifically (Burkepile 2012; Burkepile et al. 2019; Koval et al. 2020). Spatiotemporal patterns in parrotfish population structure across Florida's Coral Reef documented by Graff (2024)

and the habitat-stratified assemblage differences characterized by Walker et al. (2024) provide the distributional context within which hot spot clusters should be interpreted. Together, the coral-predation-derived PPI and the parrotfish hot spot layer contribute two components to a growing predation risk framework. The PPI reflects site-specific predation history at locations where outplanting has already occurred, while the hot spot layer extends spatial prediction to unstudied reef areas where fish community data are available but outplanting records are not.

4.4. Spatial Distribution of Parrotfish Species and Relationship to Predation Observations

The majority of Florida's parrotfish populations are dominated by small scrapers and browsers that are not capable of excavating (Graff 2024). However, species-specific differences in bite morphology, feeding behavior, and the size threshold at which excavation becomes damaging, may result in different patterns of tissue loss across coral species and outplant sizes depending on which predator assemblage dominates at a given site (Burkepile 2012; Shantz et al. 2020; van der Steeg et al. 2025). The hot spot analyses of corallivorous parrotfish (> 15 cm) from RVC-derived data collected between 2018–2024 provides important regional context for interpreting the predation pressure patterns identified in this project. With all species combined, the strongest hot spots occurred in Biscayne National Park, particularly near Key Largo. This pattern was driven by the abundance of stoplight parrotfish. Stoplight parrotfish are known as one of the primary excavators among Caribbean parrotfishes (Burkepile et al. 2019). Their occurrence on reef habitats in Florida is driven by the proximity to mangrove and seagrass habitats (Shideler et al 2017), which could help explain the observed hotspot distributions near more undeveloped coastline. This is broadly consistent with Graff (2024), who found that the density and biomass of parrotfish in southeast Florida were significantly lower than the Florida Keys and Dry Tortugas. In contrast, corallivorous redband and princess parrotfish showed more localized and variable hotspot clustering in more developed coastal habitats in the Broward-Miami ecoregion, including areas near Miami Beach and Fort Lauderdale. The co-occurrence of multiple corallivorous parrotfish species hotspots in the Broward-Miami ecoregion suggests that predation pressure in this region, while perhaps lower in overall magnitude than further south, is driven by a more diverse assemblage.

Temporal patterns in parrotfish distributions revealed variation with potential implications for the predation pressure index. Parrotfish density near Fowey Rocks was highest in 2018 but was not evident as a distinct cluster in 2022 or 2024. Redband parrotfish hot spots shifted from south to north over the survey period, with the highest abundances occurring in the Broward region by 2024. Stoplight parrotfish showed a similar, though weaker, northward trend, with a concentrated hot spot emerging off north Miami Beach in 2024 while southern Biscayne abundances remained comparatively high. These temporal dynamics highlight a potential limitation for restoration planning, as the distribution of large corallivorous parrotfish can shift substantially over multi-year timescales. Regional temporal shifts in parrotfish densities and biomass following Hurricane Irma have been documented, with populations failing to recover to pre-disturbance levels by 2022 (Graff 2024). However, local acoustic telemetry studies of individual parrotfish have documented

strong site fidelity and limited movement across reef areas (Lindholm et al. 2006), suggesting that the population-level distributional shifts detected here may not reflect the localized predation pressure experienced at individual restoration sites. This distinction underscores an important methodological consideration that broad-scale annual diver-based survey data capture community-level patterns, whereas restoration practitioners may be more concerned with the dynamic behavior of resident individuals over shorter timeframes.

The significant influence of ecoregion, habitat depth, habitat type, and relief on reef fish assemblage structure documented by Walker et al. (2024) provides important context for interpreting the spatial patterns in parrotfish hot spots identified here. The distinct predation pressures observed between the Broward-Miami and Biscayne ecoregions are likely shaped in part by these habitat-level factors, which drive differences in fish community composition. Incorporating ecoregion stratification into the RVC-based hot spot analyses conducted in this project is essential for detecting spatial and temporal changes in reef fish assemblages. Additionally, the phenomenon of tropicalization, or the expansion of tropical fish species northward as ocean temperatures rise, may increase corallivorous parrotfish densities in northern ecoregions (Walker et al. 2024). However, this movement may not be linear due to cold-water events and upwelling north of the Bahamas Fracture Zone (near Palm Beach) interacting with the Florida Current.

4.5. Efficacy of C-BRUVS Deployments for Capturing Fish Predation

4.5.1. Oasis Deployment Comparison of Video and Tissue Area

The results from the Oasis deployment provide empirical support for the hypothesis that observed parrotfish biting activity is a meaningful predictor of coral tissue loss over short deployment periods. The significant negative correlation between total predator bites and change in live tissue area (LTA) is consistent with controlled studies demonstrating that parrotfish are major drivers of Caribbean coral tissue removal (Bellwood & Choat 1990; Bonaldo & Bellwood 2009; Rempel et al. 2020). Importantly, this correlation was driven largely by parrotfish rather than butterflyfish, as parrotfish bite rates were significantly correlated with loss of LTA, while butterflyfish biting showed a weaker association. This is biologically consistent with the feeding mechanics of these taxa, as several parrotfish species use scraping and excavating bites that remove both skeleton and living tissue simultaneously (Bellwood & Choat 1990; Bonaldo & Bellwood 2009), whereas butterflyfish feed on individual polyps and may cause more localized damage per bite (Alwany et al. 2003; Cole et al. 2008).

We were unable to determine the contribution of individual parrotfish species on tissue loss from this deployment, as 83% of coral fragments were bit by two or more parrotfish species over the deployment, and 58% bitten by three or more parrotfish species. Seven species were observed to predate on *P. astreoides* and *P. porites* during this deployment at one reef site. The differential vulnerability of the two coral species was surprising. *P. porites* fragments displayed markedly greater tissue loss relative to *P. astreoides*, despite receiving similar numbers of bites. Several mechanisms may explain this pattern. First, *P. porites* may be more palatable to parrotfish. Second, the live tissue area measuring using ImageJ was two-dimensional, meaning that predation scars may have

been underreported for both species for fragments that had greater three-dimensional growth.

4.5.2. Limitations and Challenges

Anecdotally, most of the interactions with the C-BRUVS were with the coral trays and plug edges due to the presence of macroalgae. It was not always feasible to ascertain from video whether a fish targeted live coral tissue or the algae growing on the plug or tray surface. Bites recorded may therefore encompass a portion of algae-grazing events, potentially inflating apparent predation pressure on coral tissue. For all future deployments, when possible, the coral trays will be cleaned or freshly bleached prior to deployments, and plugs will be cleaned as best as possible (while avoiding coral tissue) to reduce the number of interactions that are non-coral.

The analysis of video footage presents significant challenges in distinguishing unique fish. Fish frequently exited the frame of view, and while some appeared identical upon re-entry, definitive identification was near impossible. To address this, expanding the time frame for a unique fish (identical size and pattern) to encompass 1 minute, 5 minutes, 10 minutes, or even a 50–66-minute video could be considered. Similarly, if a fish reappears hours later, this information would be valuable, however, it would be very difficult to determine through visual identification alone. One suggestion is to deploy a second camera further away with a broader view of the C-BRUVS setup to observe fish swimming in and out of the area. However, this would necessitate additional video analysis, which is already time-consuming.

Several methodological limitations warrant consideration when interpreting these results. Firstly, the camera's smaller field of view for the Oasis deployment prevented accurate size estimation for the largest parrotfish. This prevented a direct correlation between fish body size or biomass and predation pressure. However, age/phase data could provide partial support for this analysis, as terminal phase individuals, likely the largest in the population, were disproportionately responsible for the most damaging bite events, notably at Paradise, where terminal phase stoplight parrotfish accounted for 71.4% of that species' bites.

Deployments where camera batteries died before the collection of the cameras and *in-situ* coral surveys resulted in the loss of monitored predation activity. This occurred when weather limited collections beyond the 3-day window of the camera deployment and battery life. Interactions occurring after camera failure cannot be linked to the final coral tissue survey, leading to a conservative underestimate of total predation pressure for those deployments.

Finally, practical logistical constraints limit the scalability of this approach. Video footage from multi-day deployments requires substantial computer and cloud storage, and manual video analysis is time intensive. Additionally, variability in footage quality due to lighting conditions, water clarity, and camera angle impeded accurate species identification for some observed fish, resulting in some unresolved or low-confidence identifications within the dataset.

4.6. Conclusions

This project provides one of the most comprehensive syntheses to date of corallivorous fish impacts on coral restoration in Florida, integrating long-term field datasets, experimental observations, and spatial analyses across Miami-Dade and Broward Counties. The PPI, derived from a meta-analysis of 56 site-year combinations spanning 2018–2026, demonstrates that predation risk is strongly driven by reef location and has increased over time, with Miami-Dade County sites showing consistently higher predation prevalence than Broward County sites across all coral species examined. C-BRUVS deployments confirmed that parrotfish, particularly stoplight parrotfish (*Sparisoma viride*), are the primary drivers of coral tissue loss on Miami's reefs and provided direct behavioral observations of predator identity and feeding activity that are not captured by existing surveys. The coral predation PPI and the parrotfish hot spot analysis derived from the 2012–2024 NCRMP dataset are presented as two complementary but independently derived spatial indicators of predation risk. Together, these layers give restoration practitioners a more informed basis for site selection. The PPI reflects documented predation history at outplanting sites, while the hot spot analysis extends spatial prediction to reef areas where fish community data are available but outplanting records are not. Closing the gap between these two data streams through consistent concurrent collection of fish abundance and coral predation data at the same sites is the most important priority identified by this project for improving the predictive power of future PPI iterations. Additionally, targeted integrations of different coral life stages (e.g., recruits and juveniles) would strengthen this index.

5. TABLES AND FIGURES

Table 1. Description of each deployment conducted, including site information, number of coral species fragments used, and number of cameras used to conduct video analysis of corallivorous fish predation.

Deployment Description	No. of C-BRUVS	Date of Deployment	Site Name	Coral Species (No. Fragments)
Porites to CCC	0 (used Coral City Camera)	10/23/25	Coral City Camera/Port of Miami	*P. astreoides* (21)
Porites to Paradise Nursery	1	3/12/26	Paradise Nursery	*P. astreoides* (33), *P. porites* (15)
Deployment of 1 camera to Rowan Anderson MPS project during predation protection removal for outplant monoculture vs polyculture experiment	1	3/24/26	Paradise Reef PR2	*D. labyrinthiformis* (3), *P. clivosa* (3), *P. strigosa* (3)

Deployment of 1 camera to Oasis and tray of *Porites*	1	5/1/26	Oasis	*P. astreoides* (33), *P. porites* (15)
Deployment of 3 cameras + *Porites* mixed species and genotypes to Paradise Reef	3	5/13/26	Paradise Reef PR1	*P. astreoides* (40), *P. porites* (19), *P. divaricata* (28)

Table 2. Maximum and minimum interpolated prediction values used to standardize raster symbology across all survey years for each species, derived from the absolute range of interpolated values across Florida's Coral Reef.

Parrotfish Species	**Min. Interpolated Value**	**Max. Interpolated Value**
All corallivorous species	0.1	19.8
Blue (*Sc. coeruleus*)	0.0	4.8
Midnight (*Sc. coelestinus*)	0.0	2.3
Rainbow (*Sc. guacamaia*)	0.0	2.2
Princess (*Sc. taeniopterus*)	0.0	4.7
Queen (*Sc. vetula*)	0.0	1.3
Stoplight (*Sp. viride*)	0.0	3.4
Redband (*Sp. aurofrenatum*)	0.0	12.0
Redtail (*Sp. chrysopterum*)	0.0	12.5
Yellowtail (*Sp. rubrippine*)	0.0	4.4

Table 3. Number of reports, theses, and publications which met our criteria for coral predation studies. Data sources were manually scanned to determine whether they were relevant for our literature review.

Repository	**Number of Relevant Coral Predation Sources**
Web of Science	33
UM Digital Thesis Repository	5 (Weisman 2024, Rule 2023, Rivas 2020, Tobias 2024, Harrell 2022)
NSU Digital Thesis Repository	6 (Brownlee 2010, Hannigan 2022, Pisano 2023, Alfirevich 2023, Little 2024, Wagner 2024)
Florida Department of Environmental Protection (DEP)'s report archive website	3 (Sharp et al. 2024, Lirman et al. 2023, 2024)
National Oceanic and Atmospheric Administration (NOAA) report repository	4 (Data associated with the publications: Rempel et al. 2020, Rempel et al. 2024, Ladd et al. 2024, Koval et al. 2020)

Table 4. Summary of C-BRUVS video captured at each deployment.

Site	**Deployment Date**	**No. Video Files (Total File Size)**	**Earliest Video Start (h:m:s)**	**Latest Video End (h:m:s)**	**Total Video Duration (h)**

CCC	10/23/25	6 (34 GB)	7:00:00	17:25:00	12.28
Paradise Nursery	3/12/26	73 (80 GB)	7:00:00	20:59:20	39.38
Paradise Reef	3/24/26	42 (103 GB)	7:00:00	20:20:00	25.39
Oasis	5/1/26	46 (115 GB)	7:00:00	20:59:20	36.92

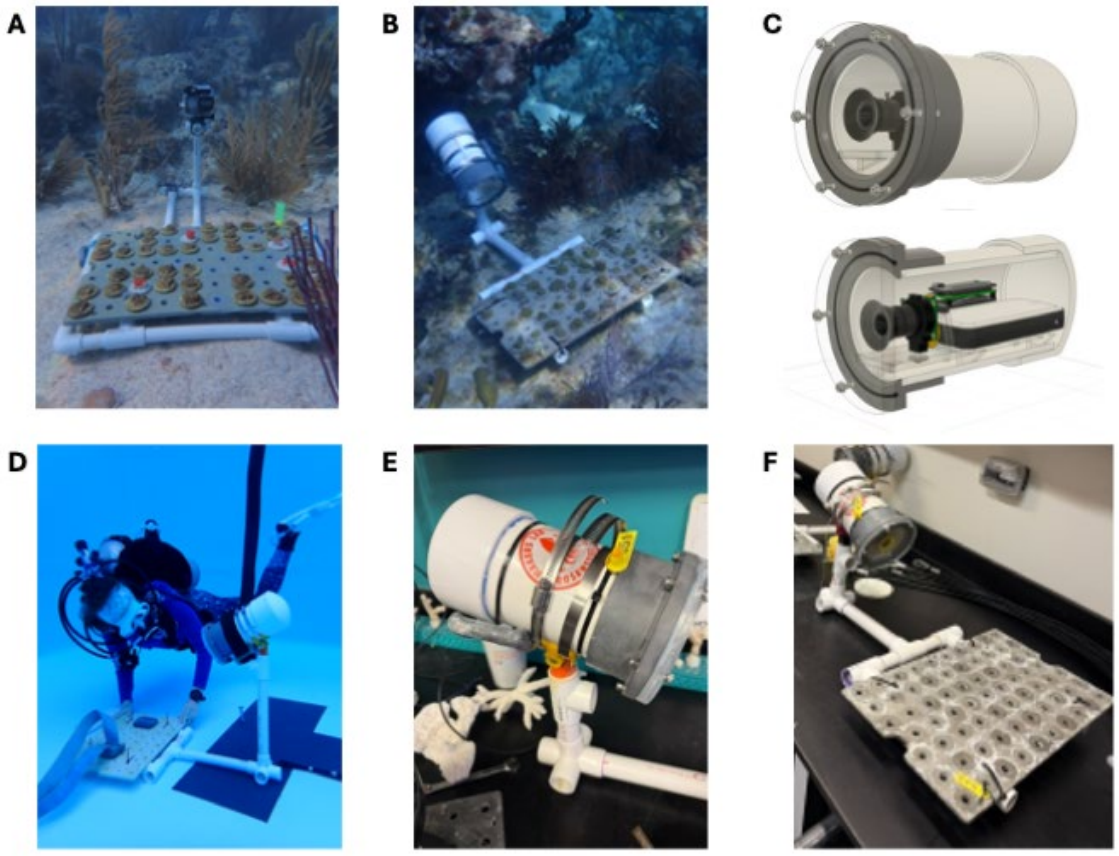


Figure 1. A) First iteration of C-BRUVS developed in 2024 by Dr. Lirman's research group, which included a GoPro attached to PVC frame. **B**) Second iteration of C-BRUVS developed during this FY25-26 project, which has a new PVC frame design and a new camera system. **C**) Rendering of 3D model of the new camera design, showing PVC housing and internal parts (Raspberry Pi computer, external battery pack, and camera lens). **D)** Pool testing of camera housing and new PVC frame. **E)** Attachment of camera to PVC frame using hose clamps. **F**) New camera system mounted onto PVC frame design with Coral Lok tray, which is designed to hold coral plugs.

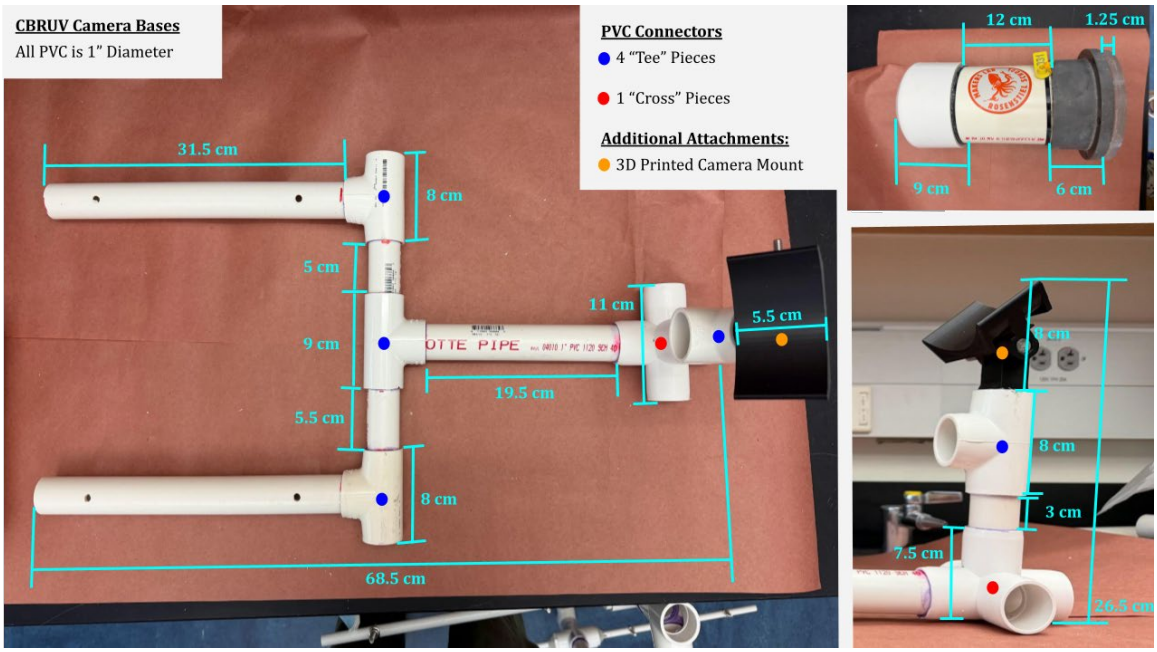


Figure 2. Dimensions of PVC stand, camera housing, and camera holder.

Figure 3. Summary of the Florida corallivory literature review (n = 33 studies). A) Number of studies in which each coral species was represented as a focal study species; species appearing in only one study are excluded. B) Summary of metrics used to report predation across studies. C) Number of studies published from 1998 to 2025.

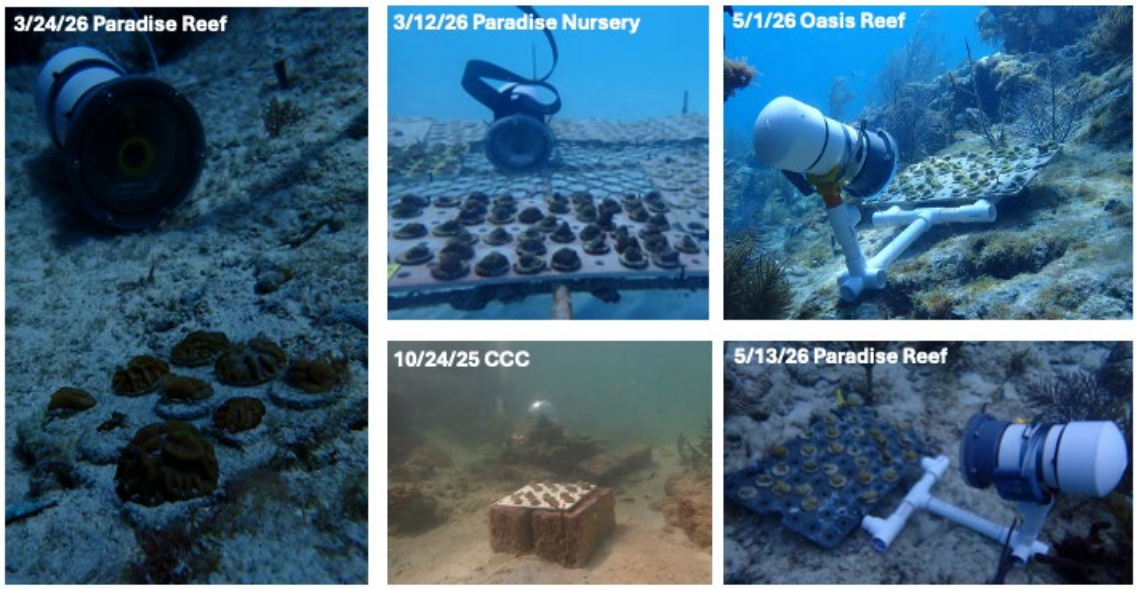


Figure 4. Image from each deployment (with corresponding date and reef site name) highlighting the different approaches and coral species tested across the project.

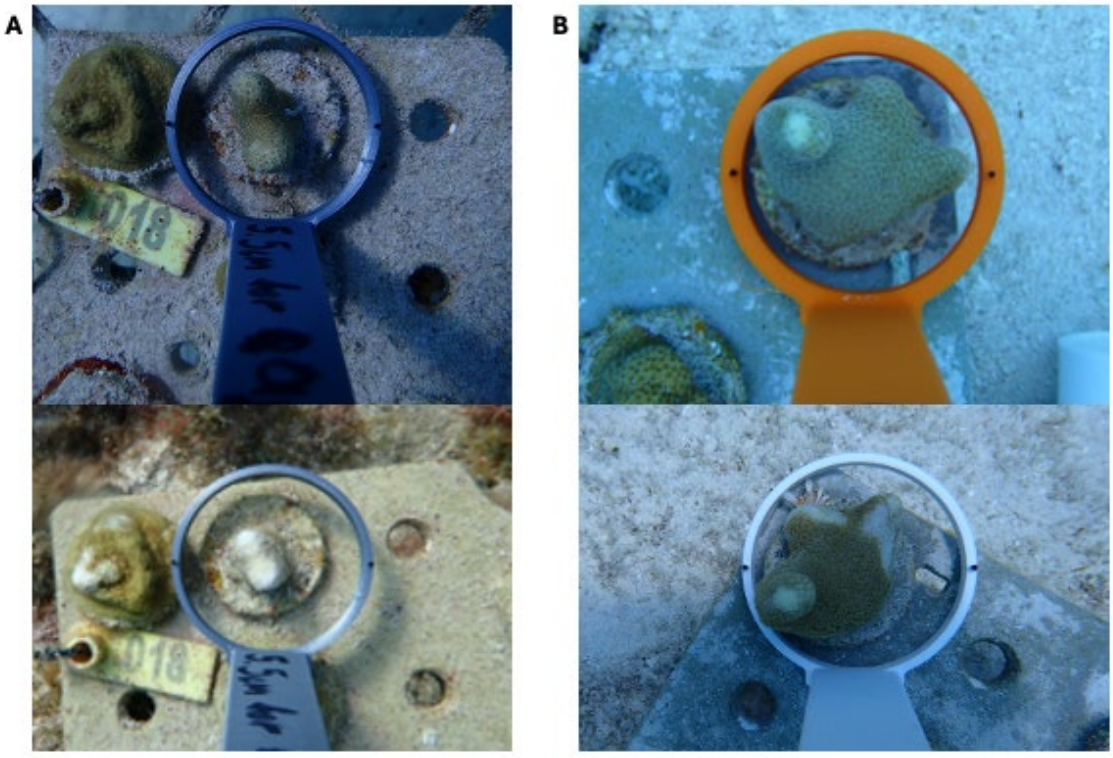
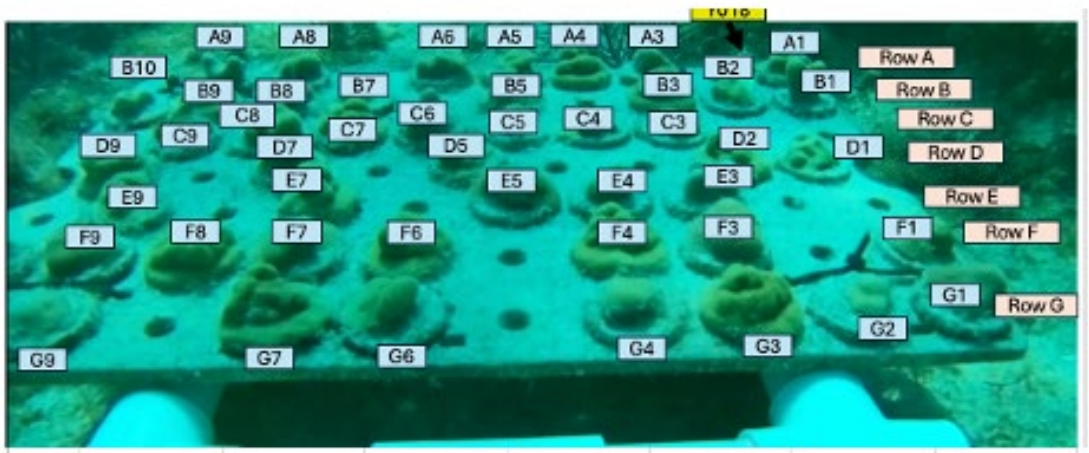


Figure 5. Images of coral tray pre- (top row) and post- (bottom row) deployment with a representative individual coral from two separate trials; **A)** Oasis Reef, **B)** Paradise Reef. Images were taken with a standardized scale bar for measuring live tissue area in ImageJ during post-processing.



Pos	A	B	C	D	E	F	G
1	PAST03	PPOR01		PAST03		PAST03	PPOR01
2		PAST03		PAST03			PAST02
3	PPOR01	PAST01	PAST03		PPOR01	PPOR01	PAST03
4	PAST02		PAST03		PAST03	PAST03	PAST02
5	PAST03	PPOR01	PAST02	PAST03	PPOR01		
6	PPOR01		PPOR01			PAST03	PAST03
7		PAST02	PAST03	PPOR01	PAST03	PPOR01	PAST03
8	PAST02	PPOR01	PAST02			PAST03	
9	PPOR01	PAST03	PPOR01	PAST03	PAST03	PPOR01	PAST02
10		PAST02				PAST02	PAST01

Figure 6. An example data sheet and annotated tray map from a live image of a C-BRUVS deployment (2026-05-01 Oasis Reef) to demonstrate the unique tracking of individual coral plugs during video analysis.

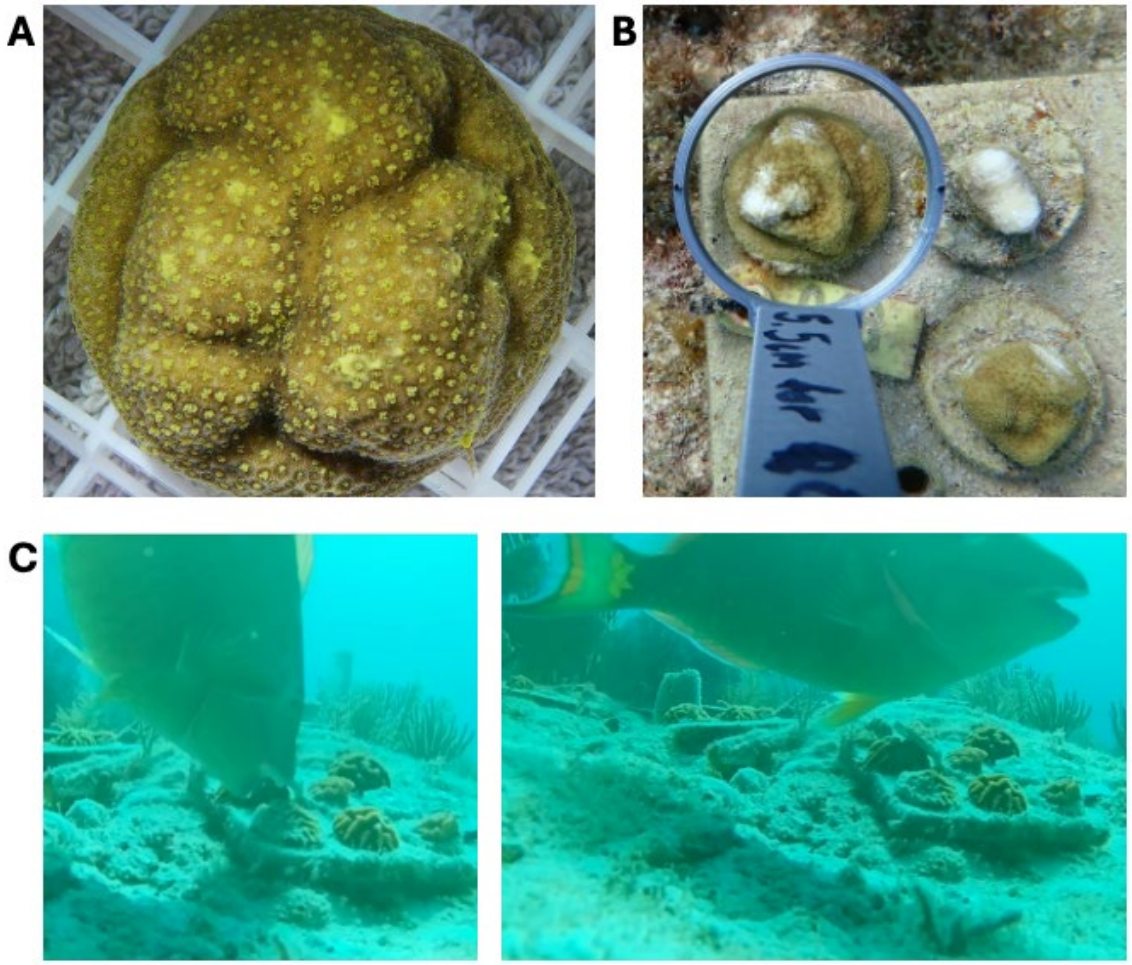


Figure 7. A) Example of a *P. astreoides* fragment from the Coral City Camera (CCC) deployment, showing the yellow surface markings on the coral tissue which may correspond to surface tissue bites. **B)** Grazing scars on *P. astreoides* and *P. porites* consistent with scraping and excavating by parrotfish. **C)** Example images during video analysis from the Paradise deployment of *P. clivosa* selectively targeted and tissue removal by parrotfish, with a stoplight parrotfish depicted in these photos. One coral plug (also a *P. clivosa* fragment) was completely removed from the tile (seen on top of a *D. labyrinthiformis* fragment in the image on the right).

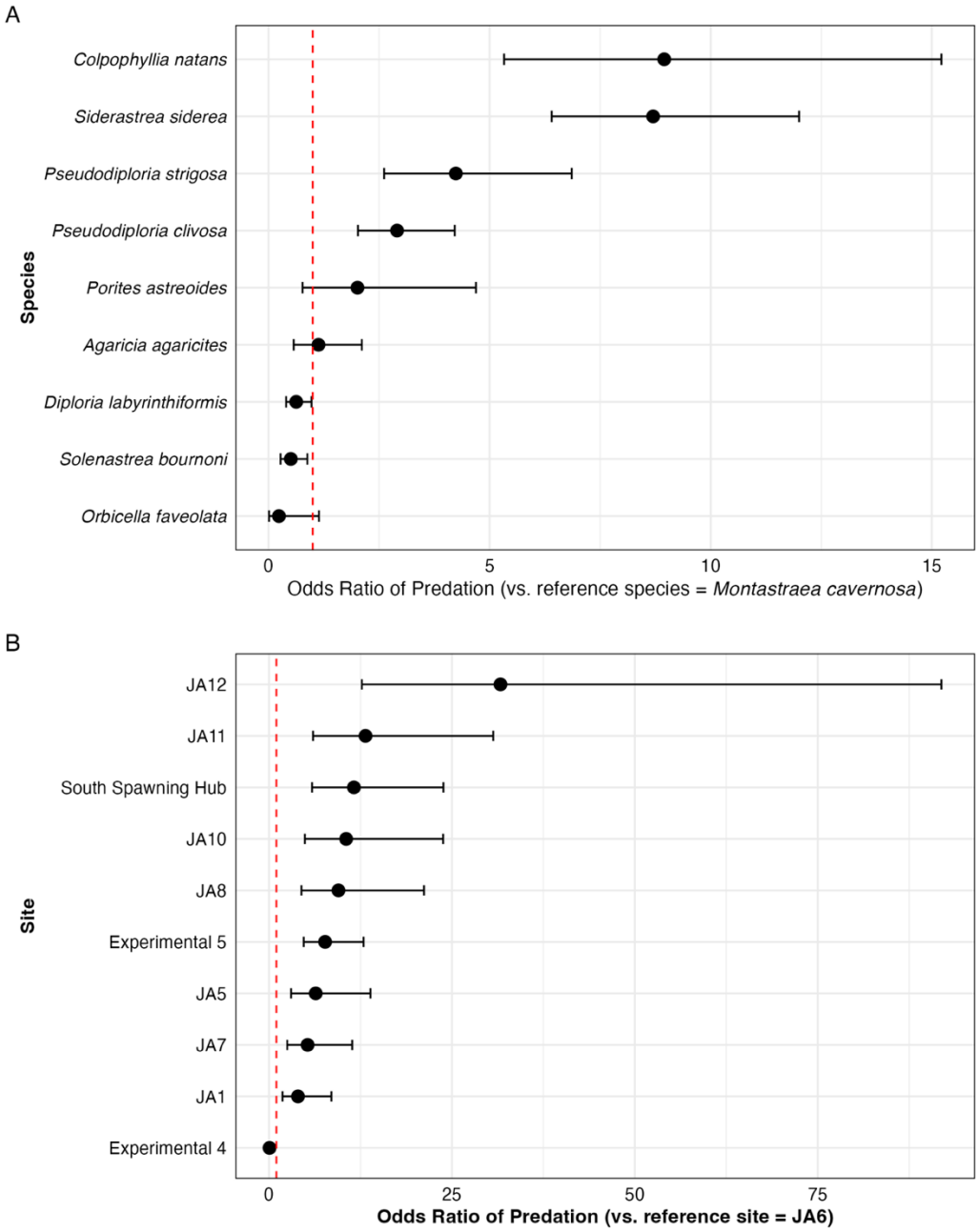


Figure 8. NSU outplanted coral odds ratios for cumulative predation prevalence for (A) coral species and (B) reef sites, derived from binomial logistic regression models. Reference category for species comparisons is *Montastraea cavernosa*, and reference site is JA6. Points indicate odds ratio estimates and horizontal bars indicate 95% confidence intervals. Dashed vertical line at OR = 1 denotes no difference from the reference. Species or sites with OR > 1 had higher odds of predation than the reference, and those with OR < 1 had lower odds. Only the ten sites with the greatest deviation from the reference are shown in panel B.

Figure 9. Odds ratios of cumulative predation prevalence for (A) coral species and (B) reef sites, derived from binomial logistic regression models. Reference category for species comparisons is *Orbicella faveolata*; reference site is Key Biscayne. Points indicate odds ratio estimates and horizontal bars indicate 95% confidence intervals. Dashed vertical line at OR = 1 denotes no difference from the reference. Species model is restricted to species with $n \geq 50$ corals monitored across at least two studies.

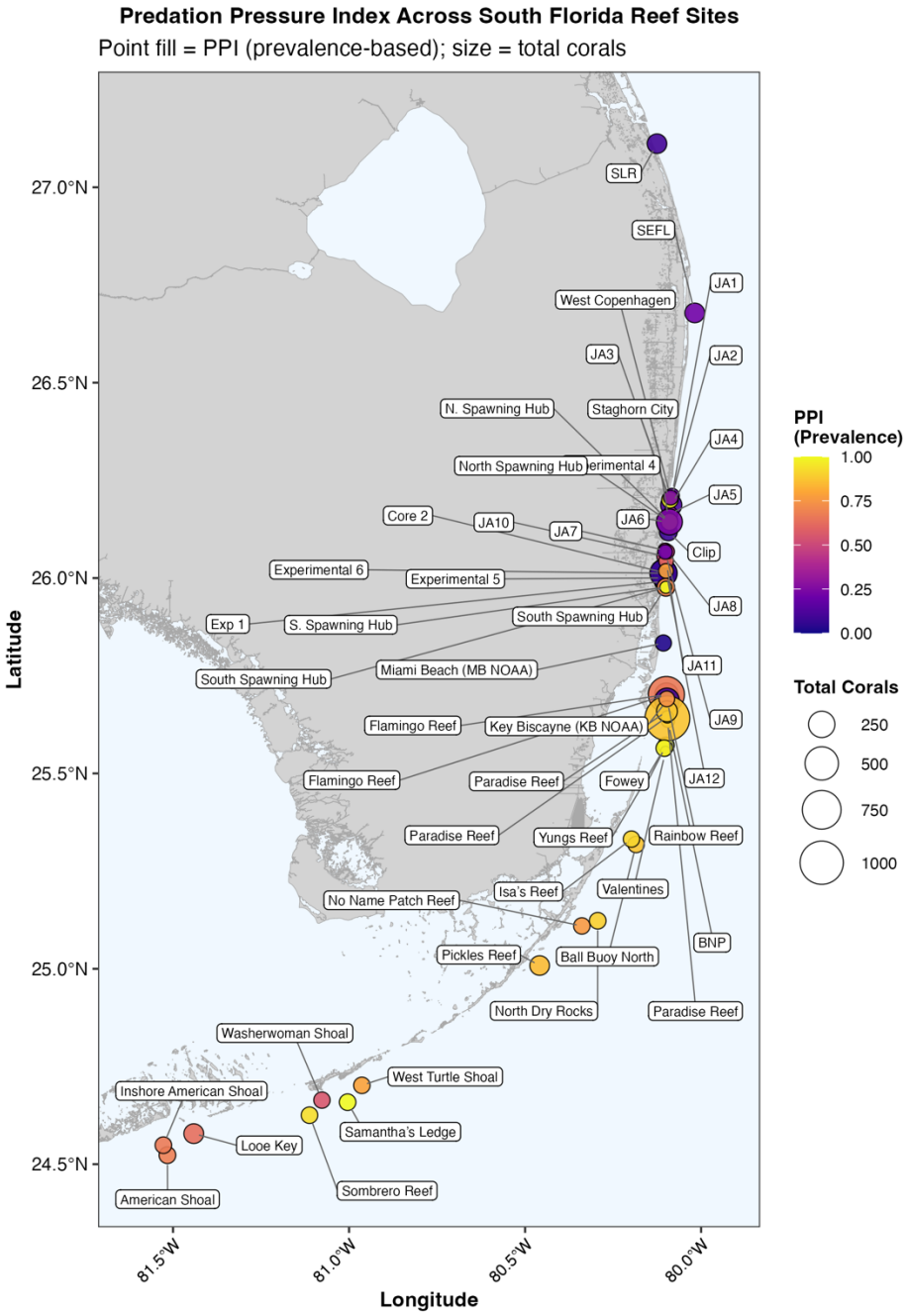


Figure 10. Spatial distribution of the prevalence-based Predation Pressure Index (PPI) across south Florida reef sites. Point fill color encodes PPI value on a continuous scale from dark purple (PPI = 0.00) to yellow (PPI = 1.00). Point size scales with total number of coral outplants monitored at that site across all years. Site labels are shown for all locations.

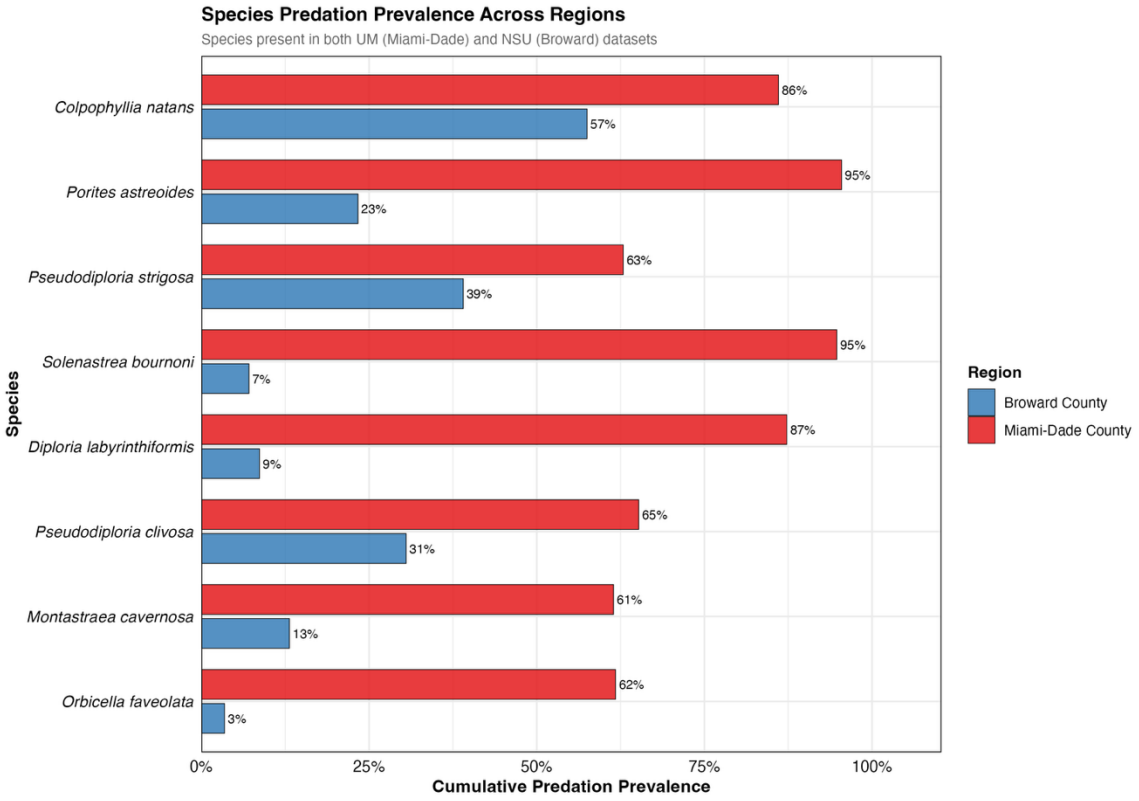


Figure 11. Cumulative predation prevalence by coral species and region for the eight species represented in both the University of Miami (Miami-Dade County, red bars) and Nova Southeastern University (Broward County, blue bars) datasets. Prevalence is the proportion of monitored outplants that experienced at least one predation event across the full monitoring period for each study, aggregated across all sites and years within each region. Miami-Dade County prevalence exceeded Broward County prevalence for all eight species.

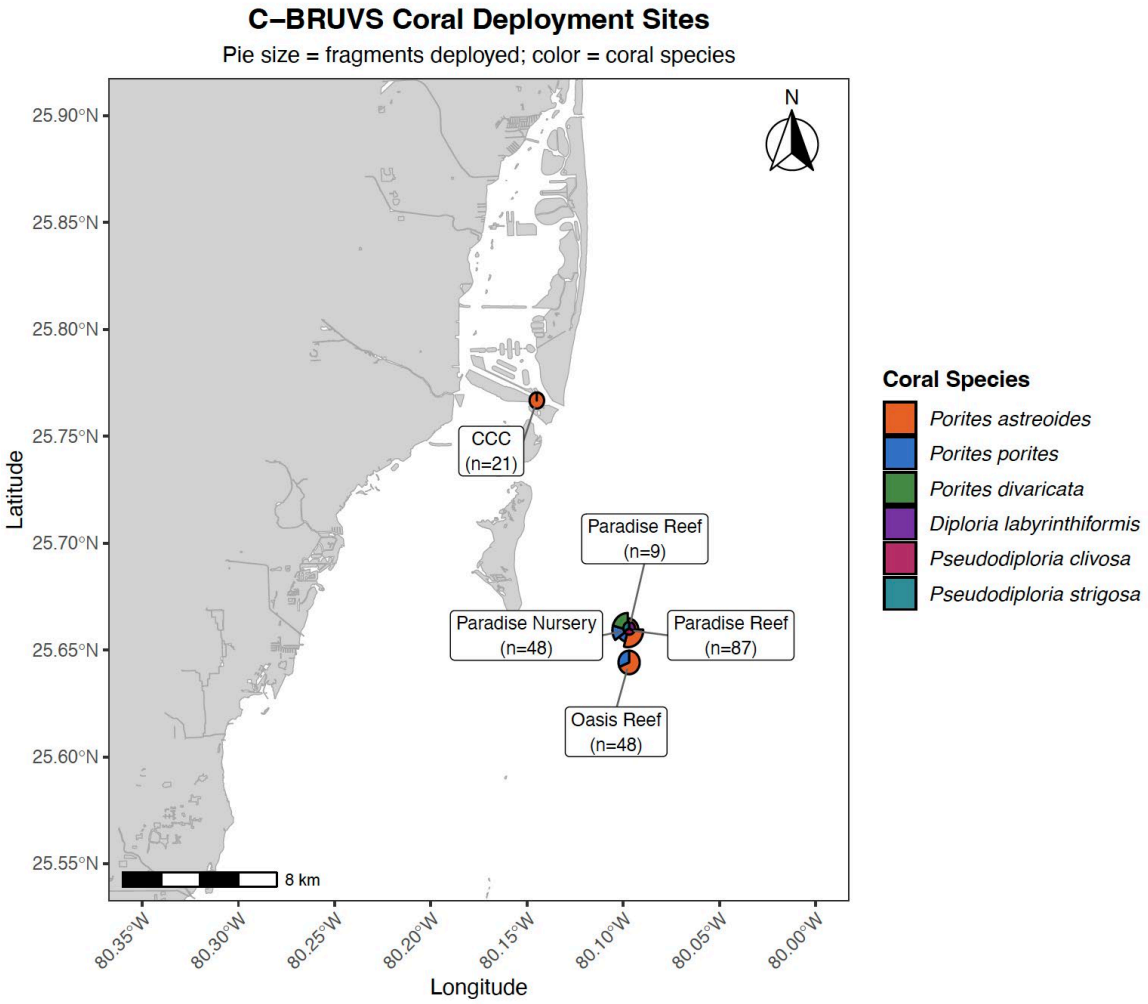


Figure 12. Map of C-BRUVS deployment sites, labeled with reef site name, and total number of corals used in that deployment.

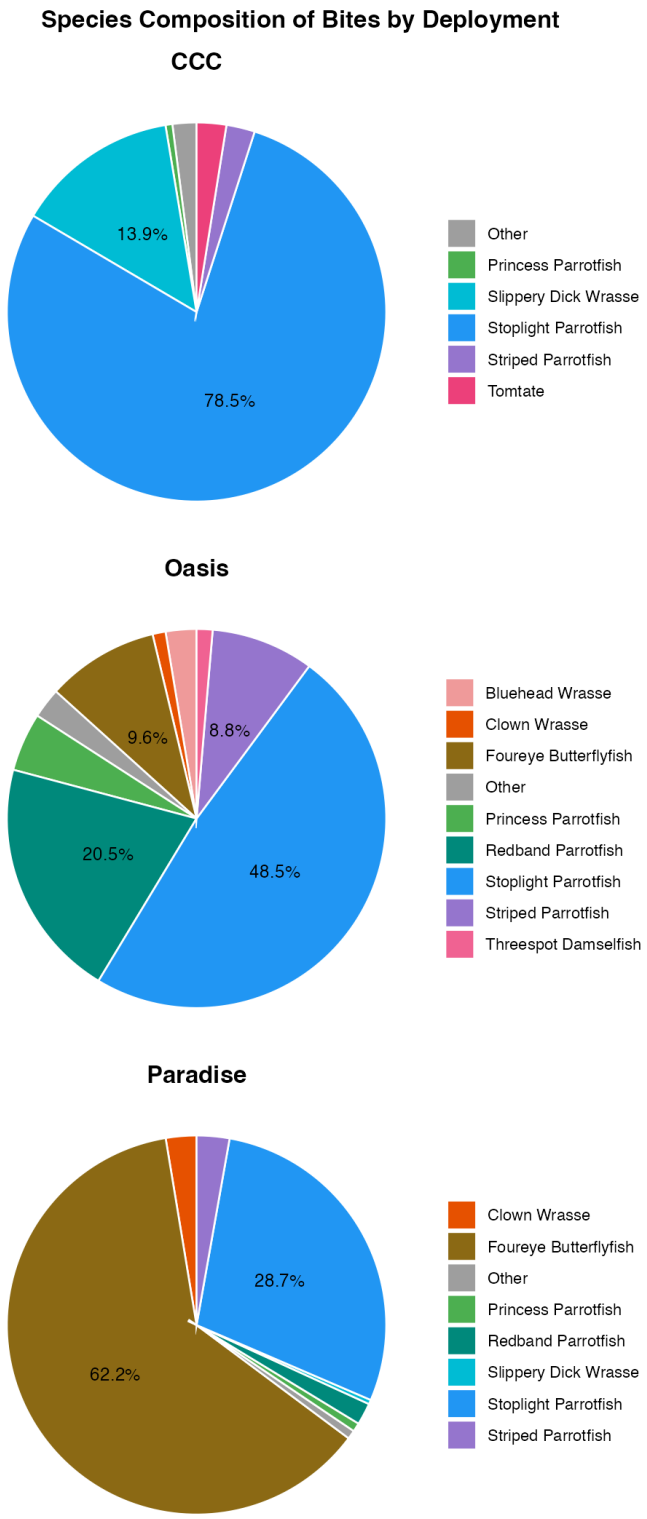


Figure 13. Proportion of fish species contributing greatest number of bites on coral, by deployment site (CCC=Coral City Camera, Oasis Reef, Paradise Reef).

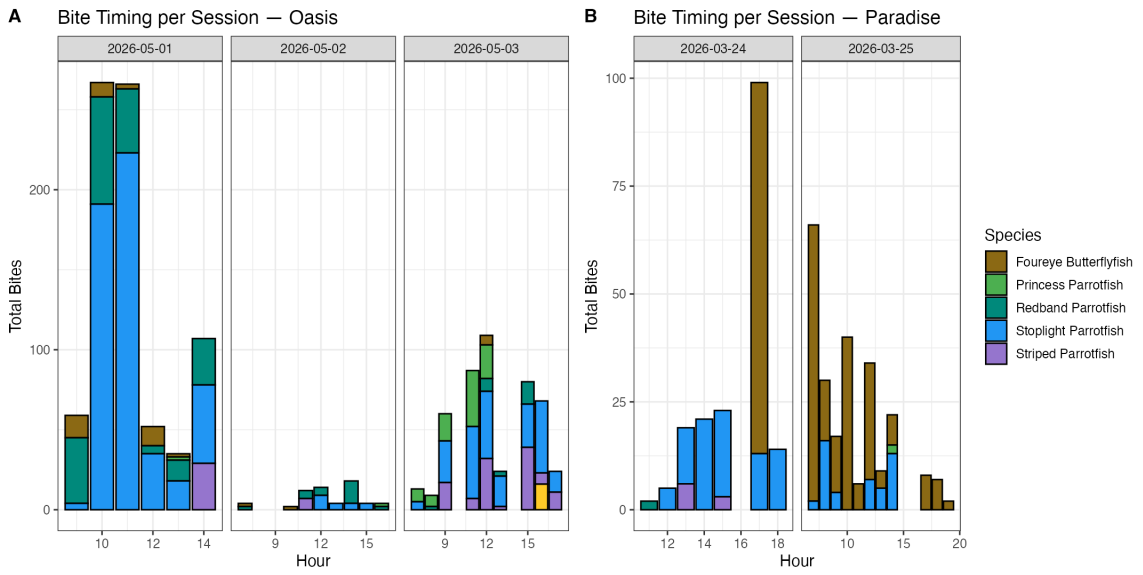


Figure 14. Hour of bites recorded per fish species (butterflyfish and parrotfish only) for A) Oasis and B) Paradise deployments and separated by each day of video.

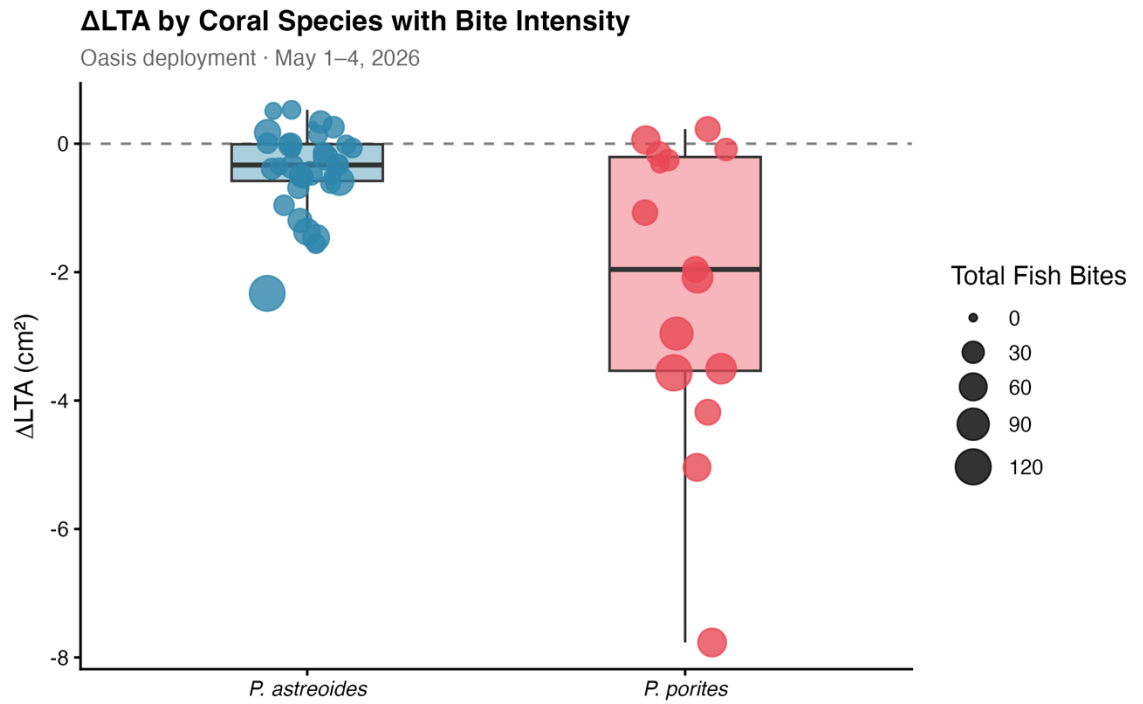


Figure 15. Boxplots of change in live tissue area by coral species. Points represent individual coral fragments measured pre- and post-deployment, and points are sized by total fish bite count across the four days.

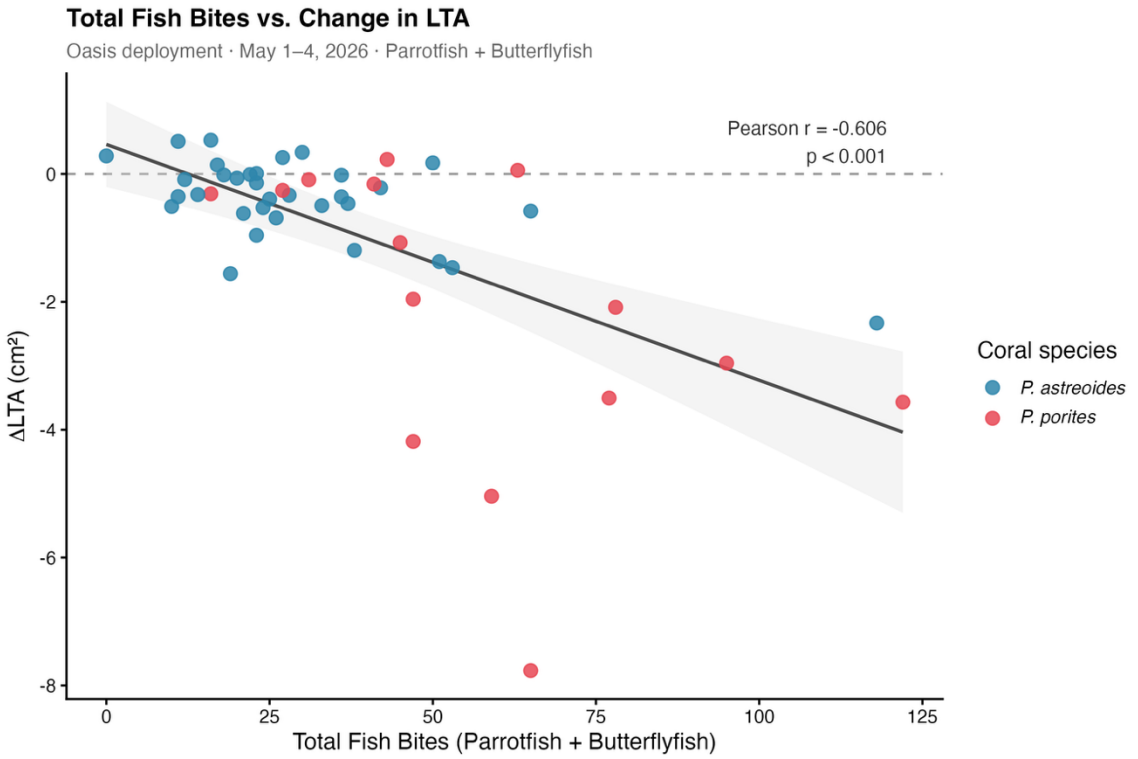


Figure 16. Correlation between total number of fish bites exhibited by parrotfish and butterflyfish, and the change in live tissue area (ΔLTA) calculated over the four-day C-BRUVS deployment at Oasis Reef. Each point represents one coral fragment, and each genotype code is labeled with its position on the Coral Lok tray. Regression line with 95% confidence interval is shown in grey. The Pearson correlation coefficient (*r*) and *p*-value are shown. Negative ΔLTA indicates tissue loss, assuming to be attributed to corallivorous fish predation.

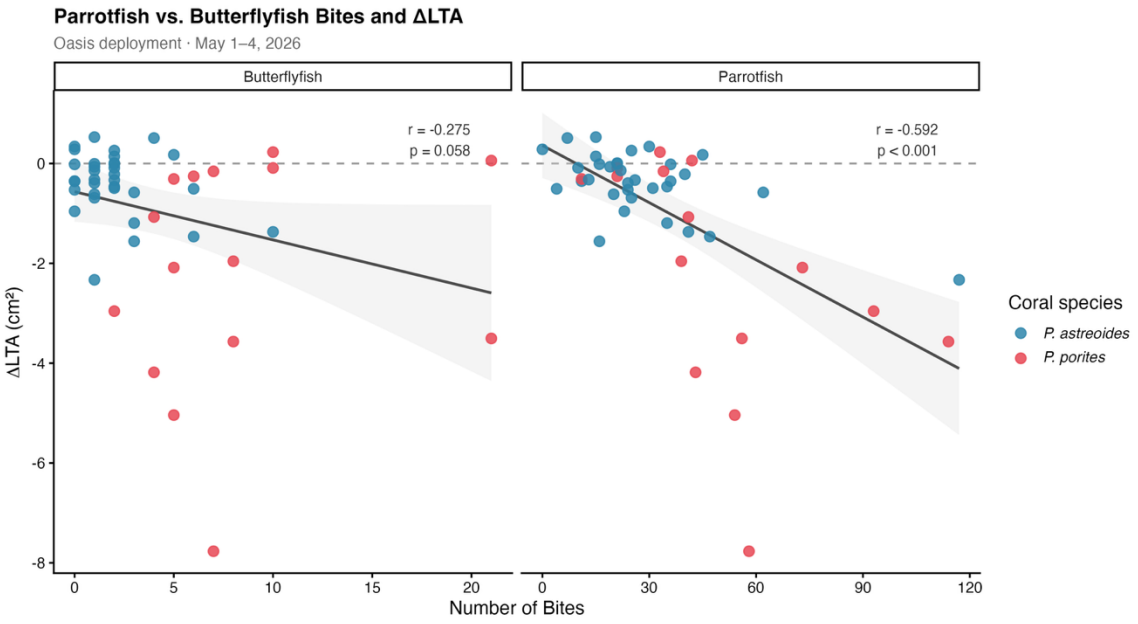


Figure 17. Number of fish bites on individual coral fragments by butterflyfish (left) and parrotfish (right) separately plotted against the individual coral's change in LTA. The Pearson correlation coefficient (*r*) and *p*-value are shown in both panels. Regression line with 95% confidence interval is shown in grey.

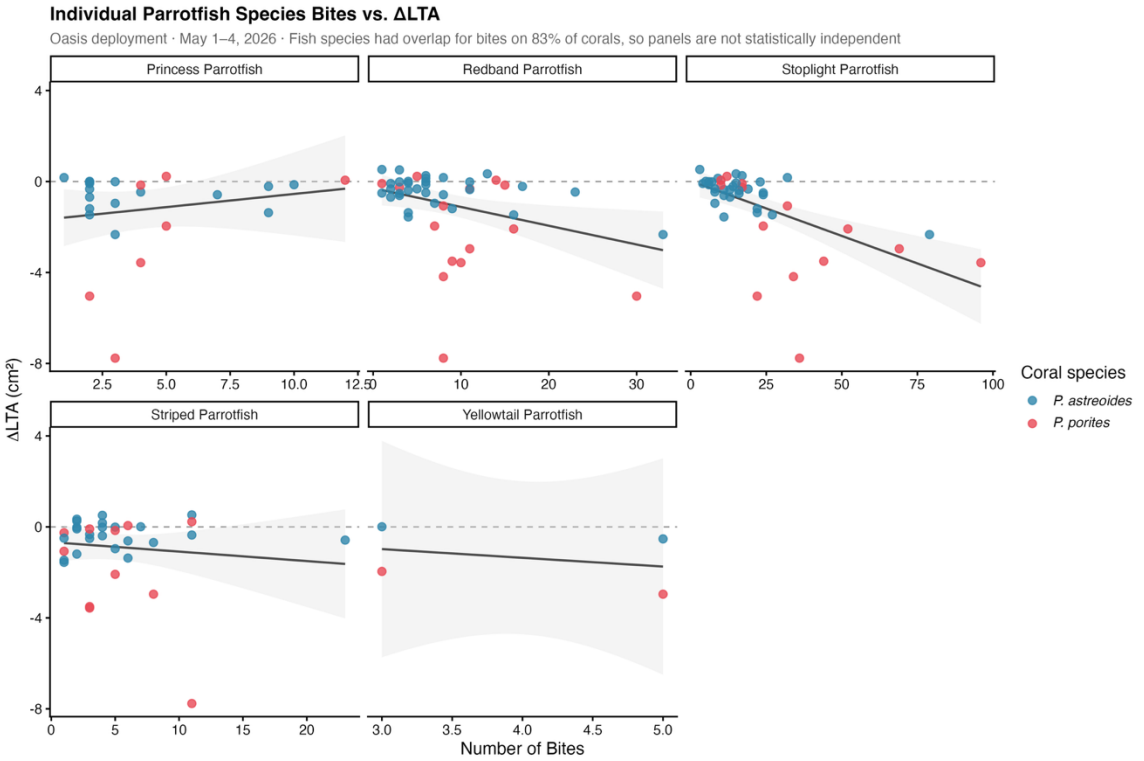


Figure 18. Bites per individual parrotfish species seen predating on individual coral fragments in C-BRUVS video analysis plotted against ΔLTA. Only fish species with ≥ 4 bites on corals are shown, and only coral fragments that received bites (bites > 0) are plotted to explore the potential relationship of number of bites to tissue loss reported. Note that 83% of coral fragments were bitten by ≥ 2 parrotfish species in the C-BRUVS video, so panels are not independent and tissue loss cannot be definitively linked to an individual parrotfish species. Data plotted for exploratory purposes. Regression line with 95% confidence interval is shown in grey.

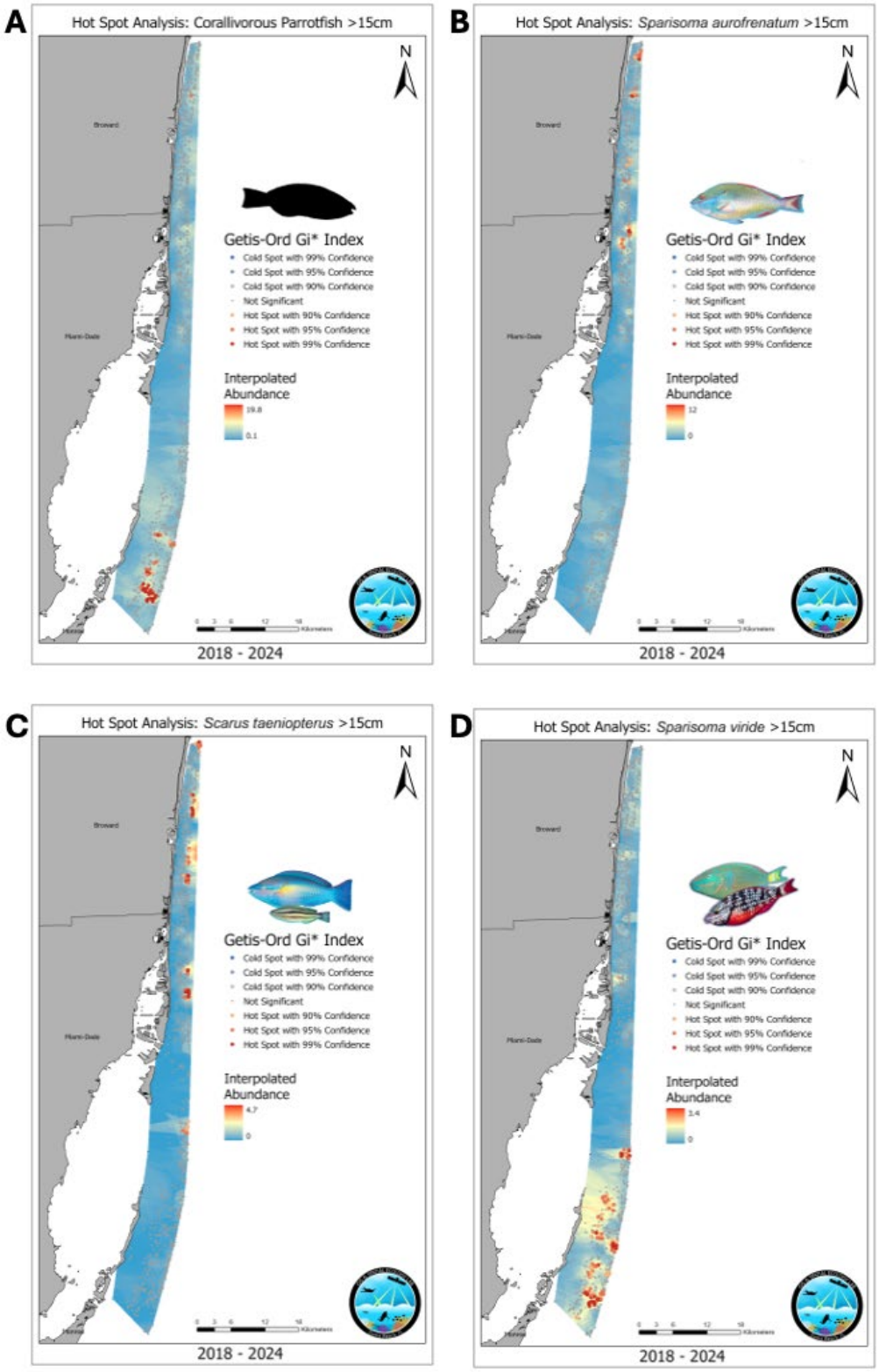


Figure 19. Hot Spot maps of all NCRMP survey years used in this analysis (2018–2024) combined for **A)** All corallivorous parrotfish, **B)** redband parrotfish, **C)** princess parrotfish, **D)** stoplight parrotfish.

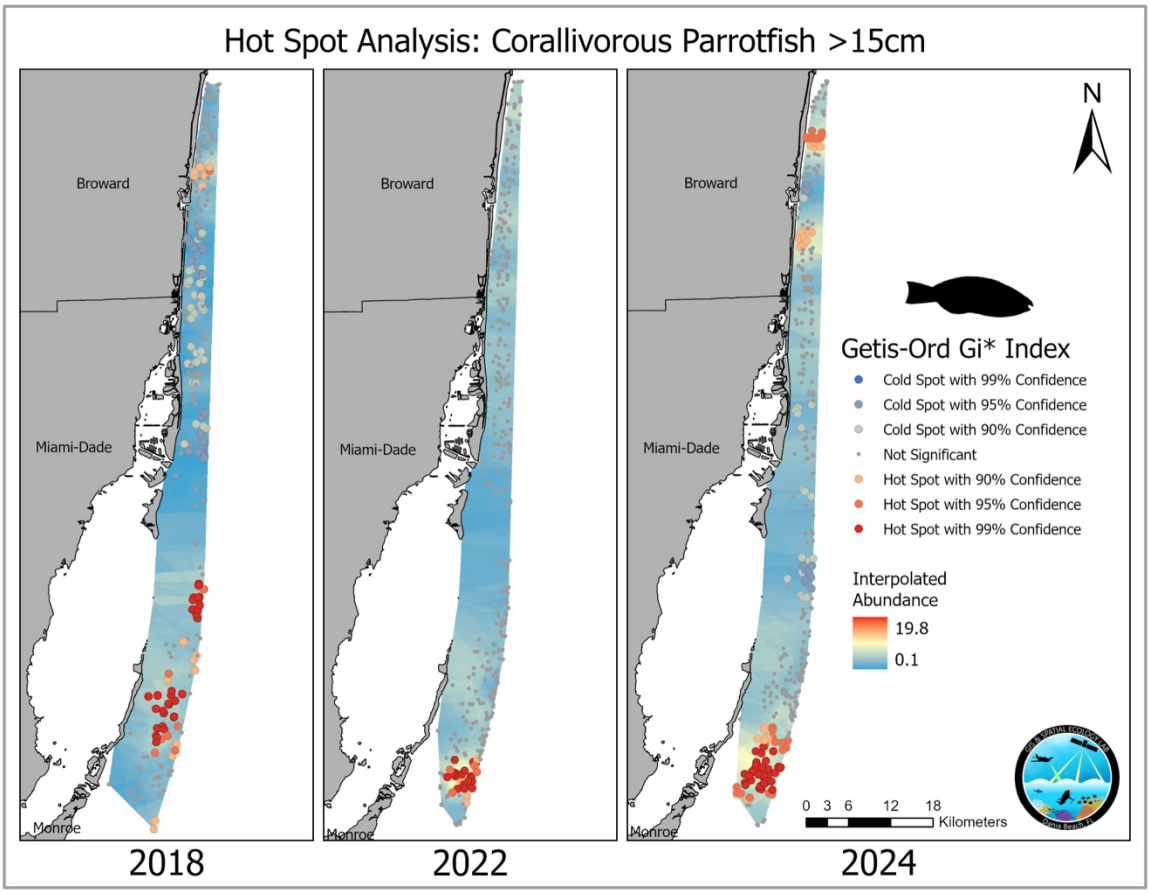


Figure 20. Hot Spot maps for all corallivorous parrotfish (>15 cm) separated by survey year.

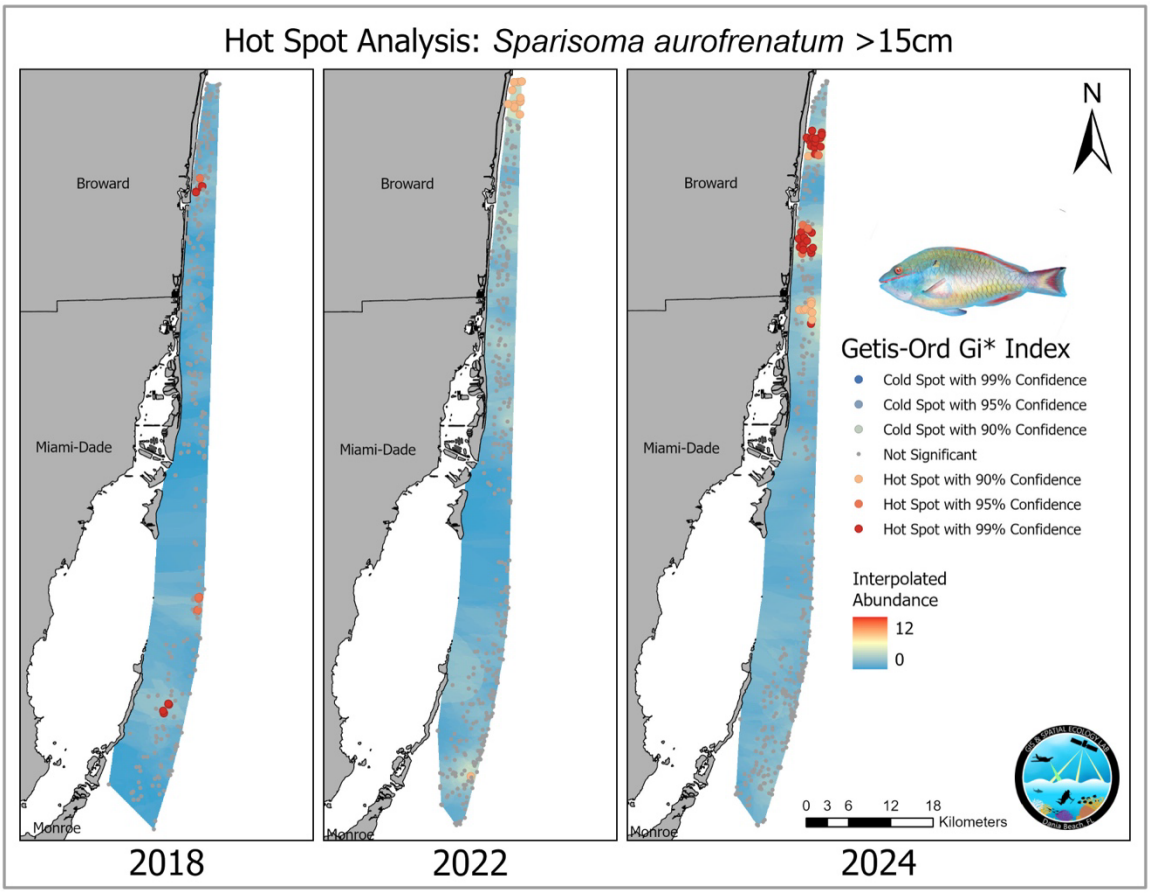


Figure 21. Hot Spot maps for redband parrotfish (>15 cm) separated by survey year.

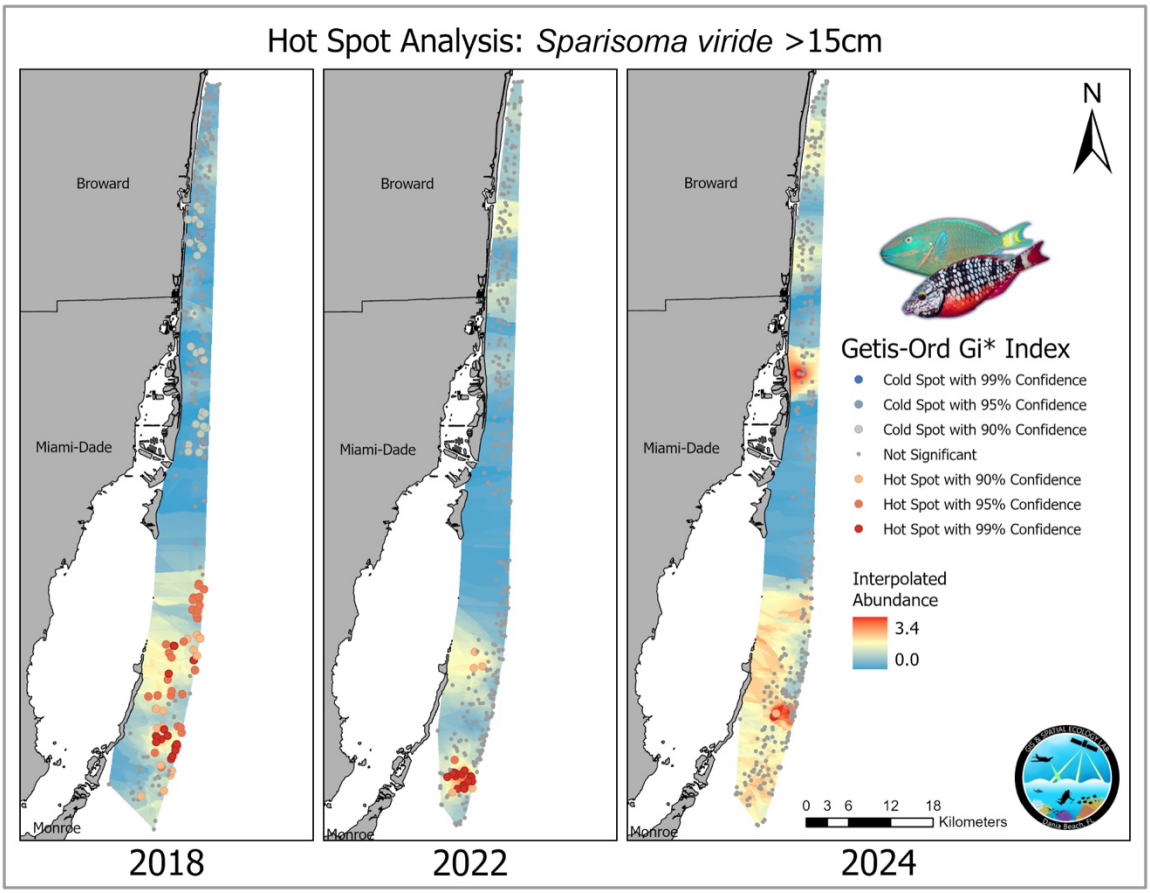


Figure 22. Hot Spot maps for stoplight parrotfish (>15 cm) separated by survey year.

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