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Monitoring Faunal Responses to Biodegradable Oyster Reef Restoration Materials with Camera Traps

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MONITORING FAUNAL RESPONSES TO BIODEGRADABLE OYSTER REEF RESTORATION MATERIALS WITH CAMERA TRAPS

by

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ABSTRACT

Restoration of the oyster reefs has become increasingly crucial due to great population declines around the globe. Intertidal oyster reefs provide essential foraging and loafing grounds to many faunal species, including several threatened/endangered wading bird species. Biodegradable oyster reef restoration materials have been introduced to avoid potential plastic pollution from traditional materials. Studies have shown success regarding oyster recruitment rates to these materials. However, their impacts on fauna using restored oyster reefs are unknown. This project aims to evaluate oyster reef restoration using biodegradable materials to increase faunal diversity, abundance, and foraging behaviors. Camera traps were deployed to observe fauna on reefs of the eastern oyster (*Crassostrea virginica*) in summer 2021, winter 2022, and summer 2022 in Mosquito Lagoon, FL. Treatments included Biodegradable EcoSystem Engineering Elements (BESE) shell mats, cement-jute tiles, and cement-jute rings. Unrestored, live reefs were used as positive controls, and unrestored, dead reefs (piles of disarticulated shell) were used as negative controls with three replicates of each treatment. A total of 11,458 vertebrates were observed out of 82,261 video clips. These comprised 44 species, including seven species of birds listed as threatened in the state of FL. There was a significant interaction between timeframe and treatment for non-foraging behaviors, such as loafing, grooming, and walking. Restoration materials did not decrease counts of foraging. However, foraging counts significantly varied over time, based on bird migratory patterns and time since restoration. This research provides essential information on the faunal use of restored and unrestored oyster reefs and highlights the importance of a mosaic of oyster reef types in estuarine systems.

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INTRODUCTION

Habitat loss has been identified as one of the most significant threats to global biodiversity (Brooks et al. 2002, Hoekstra et al. 2005, Hanski 2011). Anthropogenic habitat destruction and degradation have been linked to the extinction or severe threat of extinction of many terrestrial and aquatic species (Brooks et al. 2002, Hanski 2011). These threats of habitat loss and biodiversity declines have highlighted the need for habitat restoration, which has been espoused by many scientists and restoration practitioners (Jordan et al. 1988). Habitat restoration projects widely vary in their aims and methods. These projects may target the habitat of a specific species or encompass a large group of species found in an area (Miller and Hobbs 2007). For example, tree planting projects have been implemented in areas such as tropical rainforests where overall biodiversity declines have been associated with deforestation (Catterall et al. 2012). In freshwater systems, recovery efforts for Pacific salmon (*Oncorhynchus spp*.) have included restoring floodplains, increasing fish passage, and improving riparian habitat (Roni et al. 2014).

Estuaries are often threatened by habitat loss and degradation caused by anthropogenic stressors such as shoreline development, pollution, and dredging (Cicchetti and Greening 2011). Temperate and subtropical estuarine systems contain a great diversity of distinct habitat types, including mangrove stands, seagrass beds, saltmarsh, mudflats, and oyster reefs, which all provide important ecosystem services (Pihl et al. 2007). Extensive restoration projects have occurred in many of these habitats (Elliot et al. 2007). Work to restore mangrove habitat often includes planting and stabilizing the shoreline (Gedan et al. 2011, Su et al. 2020). Seagrass restoration projects commonly transplant rhizome fragments or seeds (van Katwijk et al. 2016).

Oyster reefs are another vital habitat type in estuaries that is commonly the focus of restoration projects.

The eastern oyster, *Crassostrea virginica*, is widely regarded as an ecosystem engineer and a keystone species due to its numerous benefits to estuarine ecosystems (Jones et al. 1994, Coen et al. 2007). The range of *C. virginica* extends approximately 8,000 km from eastern Canada to Brazil (Carriker and Gaffney 1996). These protandric hermaphrodites generally function as males upon sexual maturity, then undergo a sex change as they grow larger (Thompson et al. 1996). Macroscale environmental changes stimulate spawning, and planktotrophic larvae develop in and disperse through the water column before settling gregariously on hard substrates (Thompson et al. 1996). Settlement of multiple generations of oysters leads to the development of an oyster reef (Coen and Luckenbach 2000).

Oyster reefs are complex three-dimensional structures that provide a habitat to a myriad of other species (Wells 1961). Many recreationally and commercially valuable fish and crustacean species depend upon oyster reefs for food sources and shelter from predation (Peterson et al. 2003). The infaunal communities associated with oyster reefs are crucial components of estuarine food webs (Wells 1961, Meyer and Townsend 2000, Wong et al. 2011). These transient and resident organisms make oyster reefs rich foraging grounds for large vertebrates, such as wading birds and mammals (Wong et al. 2011, Copertino et al. 2022).

As well as being a food source and habitat for other organisms, oysters provide many other ecosystem services (Grabowski et al. 2012). As filter feeders, oysters remove excess nutrients, sediments, and phytoplankton from the water column (Grabowski and Peterson 2007). This biofiltration improves water quality and benefits submerged aquatic vegetation including seagrass by allowing deeper light penetration (Wall et al. 2008). Oyster biodeposition through feces and pseudofeces helps sequester carbon and nitrogen, which decreases the effects of eutrophication (Newell et al. 2005). Oyster reefs can also help prevent shoreline erosion by acting as breakwaters to stabilize sediment and reduce wind- and boat-generated wave energy (Meyer et al. 1997). The annual economic value of oyster reefs averages between $10,325 and $99,421 per hectare (Grabowski et al. 2012). This is estimated from oysters' ecological services and the income from commercial harvesting of oysters and reef-associated fishes and crabs.

It is estimated that 85% of shellfish reefs have been lost globally over the past century (Beck et al. 2011). Due to their role as an ecosystem engineer and keystone species, the loss of oyster populations has detrimental ecosystem-wide impacts. This significant decline is attributed to threats such as habitat loss and degradation caused by pollution, dredging, trawling, disease, competition with non-native species, and overharvesting (Lenihan and Peterson 1998, Lenihan et al. 1999, Wall et al. 2005, Walters et al. 2021). Boat wakes commonly displace oyster clusters and push them onto the reef platform, making them vulnerable to desiccation (Walters et al., 2021). Furthermore, oyster spat survival has been negatively correlated with boat wakes (Wall et al. 2005). Oyster reef degradation can lead to reef fragmentation, which decreases total reef area and increases vulnerability to disturbance (Harwell et al. 2011, Benson et al. 2023).

Oyster reef restoration has become increasingly crucial due to the significant decline in oyster populations and the loss of the ecosystem services they provide. While oyster reef restoration has traditionally emphasized the enhancement of fishery stock, restoration projects have since shifted to include the goal of restoring their ecological functions (Luckenbach et al. 2005). Restoration materials and methods vary greatly to target the local cause of population declines (Nitsch et al. 2021). Oyster cultch (disarticulated shell) has historically been chosen for restoration projects (Walters et al. 2022). In areas with large-scale restoration projects, such as

Chesapeake Bay, limitations on cultch have motivated the use of alternative materials (Schulte et al. 2009, Theuerkauf et al. 2015).

Many alternatives to oyster cultch are still being evaluated. Metal-based restoration materials include crab traps, steel gabion cages, and steel oyster mats (Johnson et al. 2019, Gilby et al. 2021, Hunsucker et al. 2021, Grizzle et al. 2023). Prefabricated, concrete-based materials include Reef Balls™ and Oyster Castles (Theuerkauf et al. 2015, Grizzle et al. 2023). Jute, hemp, and coconut coir are natural fiber-based materials that have been used for restoration alone or infused with a mineral hardener such as cement (Soucy 2020, Walters et al. 2022). Plastic-based materials, such as Naltex™ mesh bags, Vexar™ aquaculture mesh, and other polythene materials, have frequently been used to hold disarticulated oyster shells in place for reef restoration (Hadley et al. 2010, Anderson et al. 2019, Walters et al. 2021)

As the impacts of plastic pollution are becoming better understood, there is a growing concern that plastic-based restoration materials will break down and introduce harmful plastics into the environment (Walters et al. 2022). Microplastics (0.001 - 5 mm in length) are globally prevalent and are easily ingested by many organisms (Hale et al. 2020). Microplastic ingestion has been documented in over 800 animal species, including fish, turtles, marine birds, and mammals (Gouin 2020). Filter-feeding species such as oysters are especially vulnerable to microplastic ingestion (Van Cauwenberghe and Janssen 2014). In Mosquito Lagoon, FL, an average of 16.5 microplastics were found per *C. virginica* individual (Waite et al. 2018). Once consumed, plastics bioaccumulate and can transfer from one trophic level to the next (Farrell and Nelson 2013). Top predators, such as raptors, have been documented to have large numbers of microplastics in their gastrointestinal tract tissue, likely from the prey they consume (Carlin et al.

2020). Efforts to avoid contributing to plastic pollution have pushed the implementation of many biodegradable restoration materials (Walters et al. 2022).

With great diversity in oyster reef restoration methods, techniques for evaluating restoration success vary based on the scale and goals of the restoration project. Oyster metrics (reef dimensions, reef height, oyster density, shell height) and environmental variables (water temperature, salinity, dissolved oxygen) are often used in monitoring reefs post-restoration (Baggett et al. 2015). These universal metrics provide data about the oyster populations and reef structure and allow for a basic assessment of restoration performance (Baggett et al. 2015). Habitat enhancement is the goal of many oyster reef restoration projects (Peterson et al. 2003). Restoration may target a specific species, faunal group, or total biodiversity. The success of these restoration projects should be measured by the density of the selected species or faunal group (Baggett et al. 2015). Lewis et al. (2021) compared the response of reef-associated fishes to universal oyster metrics and found that fish density was comparable to oyster metrics to measure restoration success. Resident crab and mussel populations have also been used as indicators of the habitat value of restored oyster reefs (Hadley et al. 2010).

Birds have frequently been selected as indicators of biodiversity, ecosystem integrity, and restoration success (Temple and Wiens 1989, Weller 1995, Melvin et al. 1999, Gregory et al. 2003, Gregory and Strien 2010). Bird populations provide many ecological services by serving as mobile links between aquatic and terrestrial habitats, depositing nutrients, and playing a pivotal role in many trophic webs (Sekercioglu 2006). Many species of waterbirds (waterfowl, shorebirds, seabirds) depend on intertidal oyster reefs for foraging, loafing, and nesting habitat and, therefore, are impacted by the loss and degradation of reef structures (Copertino et al. 2022). Birds are often located high on the trophic web in aquatic systems, making them sensitive to changes in the trophic structure (Gregory and Strein 2010). An increase in bird diversity and foraging in an area post-restoration is often correlated with improved habitat quality (Melvin et al. 1999, Shaffer et al. 2019, Copertino et al. 2022). Shaffer et al. (2019) found a significantly higher frequency of foraging on live (natural) and restored oyster reefs than on damaged oyster reefs by birds that probe the sediment for prey. Live, restored, and degraded oyster reefs were all utilized by birds, but bird abundance and foraging behaviors have been found to be the most similar between restored and live reefs (Shaffer et al. 2019). This indicates that successful oyster reef restoration will provide additional foraging habitats for coastal birds (Copertino et al. 2022).

Camera traps are automatically triggered cameras used to take photographs or videos of wildlife (Rovero et al. 2013). For over a century, camera traps have been used to record animal behavior, monitor populations, and study ecological interactions (Trolliet et al. 2014). Camera traps are frequently used to monitor vertebrate populations as they allow for continuous sampling and work well in otherwise difficult-to-access areas (O'Connell et al. 2011, McCallum 2013, Trolliet et al. 2014). They have been successfully used to study nocturnal and highly elusive species (Troillet et al. 2014). While in-person sampling methods may frighten away some animals or cause their behaviors to be altered, camera traps are minimally invasive (O'Connell et al. 2011).

Camera traps have been used to study many large animals, including primates, elephants, giraffes, canines, and felids (Wang and Macdonald 2009, Head et al. 2012, Canu et al. 2017, McCarthy et al. 2018, Muneza et al. 2019). Smaller species are often more challenging to detect and identify with camera traps; however, camera trap design and setup can improve results. In one study, Bushnell trophy cameras detected over 80% of brown rats running along the ground at distances up to 2 m (Ortmann and Johnson 2020). Studies of small vertebrate species have used camera traps to monitor rodents, tortoises, snakes, and small birds (Melidonis and Peter 2015, Ballouard et al. 2016, Fontúrbel et al. 2020, Guise 2022).

STUDY AIMS AND HYPOTHESES

The overall goal of this study is to evaluate oyster reef restoration that included multiple biodegradable materials to determine if restored reefs alter faunal diversity, abundance, and frequency of foraging. To achieve this goal, this study aimed to 1) determine if faunal diversity and abundance differed between unrestored, live oyster reefs; unrestored, dead oyster reefs; and reefs restored with different biodegradable materials; and 2) determine if the frequency of foraging differed between reef types. I hypothesize that faunal diversity and abundance on restored reefs will be more similar to live reefs than dead reefs.

H_0: Faunal diversity and abundance do not differ between reef types.

H_a: Faunal diversity and abundance are similar between restored reefs and live reefs.

H_b: Faunal diversity and abundance are similar between restored reefs and dead reefs.

This study also aims to test the hypothesis that the frequency of foraging will be higher on restored and live reefs than on dead reefs.

H_0: Counts of vertebrates foraging do not differ between reef types.

H_a: Counts of vertebrates foraging are similar between restored reefs and live reefs.

H_b: Counts of vertebrates foraging are similar between restored reefs and dead reefs.

RESEARCH METHODS

Site Description

The Indian River Lagoon (IRL) is an estuary system on Florida's Atlantic coast. It covers 40% of the coastline, is 251 km long, and encompasses three connected lagoons: Mosquito Lagoon, Banana River Lagoon, and the Indian River Lagoon (ECFRPC 2016). The IRL system is a transitional zone between temperate and subtropical biomes (Lapointe et al. 2015). This and the diverse array of habitat types including mangrove stands, oyster reefs, seagrass beds, and saltmarsh allow for a high level of biodiversity (Gilmore 1995). The Indian River Lagoon National Estuary Program (2019) estimated that the IRL is inhabited by over 2,000 species of plants, 600 species of fish, and 300 species of birds, including 50 threatened or endangered species during some portion of their lifecycles (IRLNEP 2019). The IRL is also an essential stop for many migratory bird species as it is located along the 'Atlantic Flyway,' a migration route used by many waterbirds (IRLNEP 2019). This system has been threatened by the effects of eutrophication and harmful algal blooms (Lapointe et al. 2015). Agricultural and urban fertilizer sources are the most significant inputs of nitrogen and phosphorus into the water (Badruzzaman et al. 2012). An excess of nutrients has negatively affected water quality and threatened biodiversity in the IRL (Lapointe et al. 2015).

Mosquito Lagoon is the northernmost section of the IRL. Oyster reef coverage in Mosquito Lagoon decreased by 63% between 1943 and 2021 (Benson et al. 2023). Boat strikes and boat wakes cause significant damage to oyster reefs in this region which can transform a live reef into a large pile of disarticulated shell (Walters et al. 2021). For this study, these large piles of disarticulated oyster shell, approximately one meter above water level, are referred to as dead oyster reefs. Over a dozen bird species that rely on the habitat of Mosquito Lagoon are listed as threatened by the state of Florida, including *Egretta rufescens* (reddish egret), *Egretta caerulea* (little blue heron), *Egretta tricolor* (tricolored heron), *Haematopus palliates* (American oystercatcher), and *Sternula antillarum* (least tern) (FFWCC 2022).

Restoration Materials

Oyster restoration in Mosquito Lagoon is vital to returning oyster populations to their historical levels and maintaining habitats for the many species that rely on them. Plastic-free restoration materials have been deployed in Mosquito Lagoon since 2019. These materials have included cement-jute structures and Biodegradable Ecosystem Engineering materials (BESE). BESE material comprises 98% organic matter, making it almost entirely biodegradable (Nitsch et al. 2021). BESE-shell mats are constructed by attaching 36 disarticulate oyster shells to the BESE mat using stainless steel cable ties. BESE-shell mats are placed on dead oyster reefs that have been raked down to water level and connected with steel cable ties to cement irrigation weights. This method reduces the number of mats dislodged by waves due to storms and boat wakes.

Cement-jute rings and cement-jute tiles are two other biodegradable oyster reef restoration materials utilized in Mosquito Lagoon. These materials are made of jute (a natural fibrous cord) soaked in cement and shaped into a circle (rings) or a large square (tiles). Cement-jute rings and cement-jute tiles are deployed using similar methods of BESE-shell mats by placing them in a quilt-like pattern covering a raked-down dead oyster reef. These materials have shown great success in terms of oyster larvae recruitment, with 1,000 live oysters/m^2 after three years (Sailor-Tynes et al. 2023). Restoration work on selected oyster reefs took place in June

2021. Unrestored, live reefs were used as positive controls, and unrestored, dead reefs as negative controls with three replicates of each treatment.

Figure 1: Restoration materials deployed in Mosquito Lagoon. A: BESE-shell mats; B: Cement-jute tiles; C: Cement-jute rings.

<u>Camera Trap Deployment</u>

Camera traps (Bushnell Trophy Cameras, Model 119876) were deployed on intertidal oyster reefs in Mosquito Lagoon, FL, to document and analyze faunal presence and behavior. Fifteen oyster reefs were selected, with one camera trap mounted on a pole, sign, or tree near each reef. These camera traps were set to record a 10-second video when triggered by motion. Approximately every two weeks, camera batteries and SD cards were exchanged, and cameras were repositioned as needed if moved by animals, storms, or blocked by vegetation. Camera traps were deployed on reefs for three periods: 26 May through 18 August 2021, 6 January through 22 February 2022, and 19 July through 30 August 2022 for a total of 156 days. Pre-restoration data was 39 days from 26 May 2021 to 4 July 2021, and post-restoration data was 117 days.

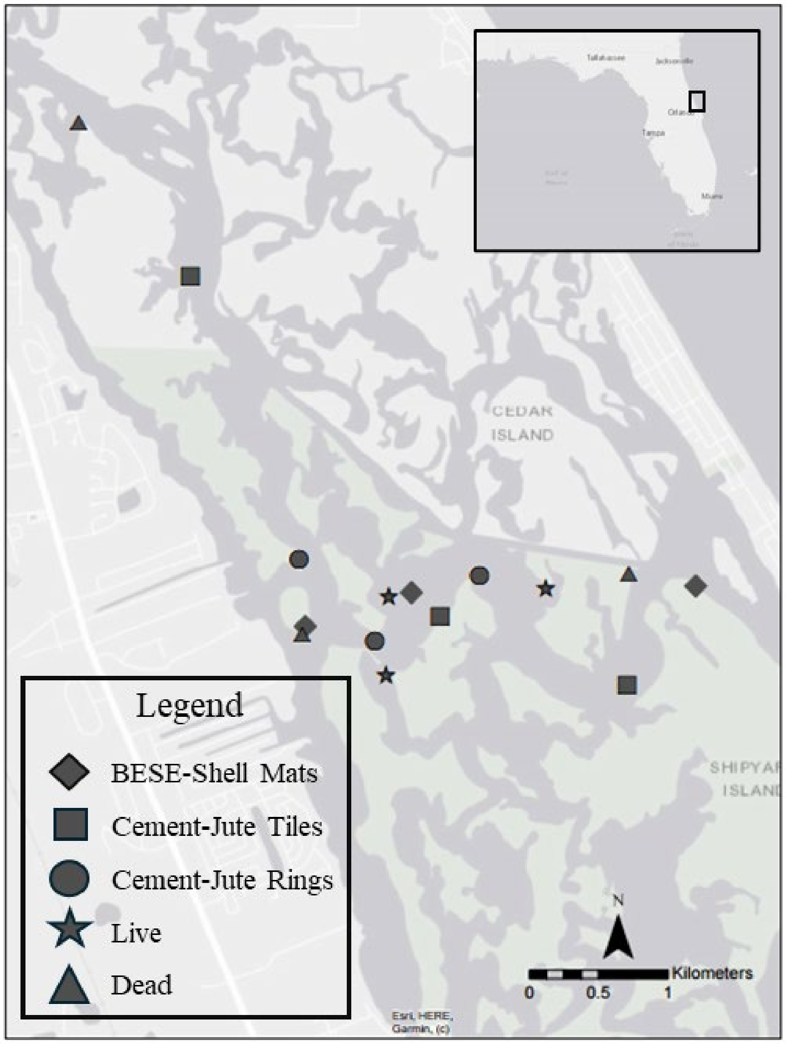

Figure 2: Map of the 15 oyster reef sites where camera traps were deployed in Mosquito Lagoon, located along the east coast of central Florida.

Video Processing

Clips from camera traps were watched one at a time, and fauna on oyster reefs or within one meter of the reef in the water were recorded. As the full extent of each reef could only be seen at low tide, videos taken at mid or high tide were compared side-by-side to a video of the same reef at low tide to visually approximate whether an individual was within the one-meter buffer of the reef. The fauna recorded included birds and mammals such as raccoons, river otters, and small rodents. Fish, insects, crustaceans, manatees, dolphins, and sea turtles observed in clips were not recorded or included in analyses. For each vertebrate seen, its behavior was documented. Behaviors included: solo foraging (active), solo foraging (stationary), group foraging (active), group foraging (stationary), consumption (solo), consumption (group), parent-assisted consumption, loafing (solo), loafing (group), courting, mating, grooming (solo), grooming (group), investigating camera, flying, walking, swimming, collecting nesting materials, and unidentified.

Vertebrates were identified to the species level. Due to lighting or distance from the camera, some individual's species or behavior was not identifiable. In these situations, the individual was still recorded but was classified as "unidentified" for species and behavior if necessary. Because the vertebrates recorded were not individually identifiable, a ten-minute rule used by Rifenberg et al. (2021) was used to determine whether a vertebrate was the same individual or a new one. If a break of less than ten minutes passed between clips containing a vertebrate of the single species, the clips were considered to be of the same individual. If a break of more than ten minutes passed between clips containing a vertebrate of the species, this vertebrate was considered a new individual.

Data Analysis

To assess differences in faunal communities based on reef types, non-metric multidimensional scaling (NMDS) was used. Fauna species were grouped into families to reduce the number of zeros in this ordination plot. Fifty runs were conducted with real data, and 50 runs were conducted with randomized data, and a two-dimensional solution was recommended.

PERMANOVA tests were used to determine if there were significant differences in faunal communities between reef types. Pre-restoration data was excluded from all analyses as it was too short of a timeframe to accurately represent a pre-restoration baseline. Species observed $\leq$ five times in the post-restoration data were excluded from the NMDS and PERMANOVA. Species richness was calculated for each reef, and a 1-way ANOVA was used to determine differences. Tukey HSD post-hoc testing was conducted to make pairwise comparisons of each reef type.

To compare behavior data between restoration materials and between timeframes, total faunal counts and counts of each behavior on each reef were divided by the number of days in each timeframe. This was done to account for the difference in number of days in each timeframe. Repeated measures ANOVAs were used to compare behavior data between restoration materials and between timeframes. For this analysis, all behaviors were grouped into two categories: foraging and non-foraging.

RESULTS

A total of 11,458 vertebrates were observed from 82,261 video clips. These were made up of 44 species, including six Florida state-designated threatened species of birds and one federally threatened species (wood stork). The most observed species on BESE-shell mat reefs was *Ardea herodias* (great blue heron), which comprised 15.2% of observations on this reef type. On cement-jute tile reefs, cement-jute ring reefs, and live reefs, the most observed species was *Pelecanus erythrorhynchos* (American white pelican), which made up 23.0%, 21.2%, and 18.8% of observations, respectively. The most observed species on dead reefs was *Thalasseus maximus* (Royal tern), which made up 22.9% of the observations.

Nine species which made up 28.7% of all observations were seen only in the winter timeframe, many of which are migratory species. Observations during the winter timeframe made up 79.5% of all observations, with an average of 298.2 faunal observations per day. In comparison, the summer 2021 timeframe had an average of 28.3 observations per day, and the summer 2022 timeframe had an average of 22.1 observations per day.

Table 1: Total counts and percentage of observations for each bird species per reef type from post-restoration data.

Species	Total Count	Percent of Observations				
		BESE-Shell Mat Reefs	Cement-Jute Tile Reefs	Cement-Jute Ring Reefs	Live Reefs	Dead Reefs
Actitis macularius (Spotted sandpiper)	36	0.30	0.09	0.38	0.51	0.35
Anas fulvigula (Mottled duck)	13	0.00	0.19	0.50	0.64	0.03
Anhinga anhinga (Anhinga)	37	1.20	0.94	0.38	0.89	0.07
Ardea alba (Great egret)	189	3.29	2.82	1.51	6.26	0.90
Ardea herodias (Great Blue heron)	841	12.77	6.95	14.00	4.98	6.76
Arenaria interpres (Ruddy turnstone)	16	0.00	0.00	0.25	0.00	0.19
Bubo virginianus (Great-horned owl)	1	0.00	0.00	0.00	0.00	0.01
Butorides virescens (Green heron)	16	0.30	0.09	0.00	0.38	0.12
Calidris alba (Sanderling)	1,003	0.10	0.19	3.28	0.00	13.47
Calidris mauri (Western sandpiper)	32	0.00	0.00	0.00	0.00	0.44
Cathartes aura (Turkey vulture)	1	0.00	0.09	0.00	0.00	0.00
Charadrius semipalmatus (Semi-palmated plover)	259	0.00	0.00	0.00	0.00	3.58
Corvus ossifragus (Fish crow)	5	0.00	0.00	0.50	0.00	0.01
Coragyps atratus (Black vulture)	3	0.10	0.00	0.00	0.00	0.03
**Egretta caerulea* (Little Blue heron)	80	1.90	1.13	0.00	2.43	0.41
**Egretta rufescens* (Reddish egret)	13	0.50	0.09	0.00	0.00	0.10
Egretta thula (Snowy egret)	50	1.00	0.28	0.50	3.70	0.06
**Egretta tricolor* (Tri-colored heron)	30	1.90	0.28	0.00	0.77	0.03
Eudocimus albus (White ibis)	400	9.68	9.02	2.52	13.03	1.18
**Haematopus palliatus* (American oystercatcher)	173	7.78	3.38	3.40	0.00	0.44
Larus delawarensis (Ring-billed gull)	750	7.88	5.73	8.83	6.39	6.78
Larus marinus (Great black-backed gull)	3	0.00	0.00	0.00	0.00	0.04
Leucophaeus atricilla (Laughing gull)	753	1.50	0.56	2.27	0.26	9.85
Lophodytes cucullatus (Hooded merganser)	32	0.10	0.94	0.00	2.68	0.00
Megaceryle alcyon (Belted kingfisher)	66	3.29	1.79	1.39	0.26	0.01
Mergus serrator (Red-breasted merganser)	550	2.10	20.39	14.50	19.03	0.66
***Mycteria americana* (Wood stork)	3	0.30	0.00	0.00	0.00	0.00

Species	Total Count	Percent of Observations BESE-Shell Mat Reefs	Cement-Jute Tile Reefs	Cement-Jute Ring Reefs	Live Reefs	Dead Reefs
Nyctanassa violacea (Yellow-crowned night heron)	204	2.20	4.98	0.76	2.68	1.41
Nycticorax nycticorax (Black-crowned night heron)	125	2.30	1.32	4.79	2.43	0.43
Pandion haliaetus (Osprey)	78	2.30	2.54	1.64	0.89	0.11
Pelecanus occidentalis (Brown pelican)	963	9.88	4.79	7.94	2.30	10.12
Phalacrocorax auritus (Double-crested cormorant)	168	4.59	2.07	4.67	1.79	0.68
Plegadis falcinellus (Glossy ibis)	1	0.10	0.00	0.00	0.00	0.00
Pluvialis squatarola (Black-bellied plover)	80	0.40	0.00	0.13	0.00	1.04
Podilymbus podiceps (Pied-billed grebe)	7	0.00	0.09	0.13	0.64	0.00
**Sternula antillarum* (Least tern)	33	0.00	0.00	0.00	0.00	0.46
Thalasseus maximus (Royal tern)	1,688	0.20	0.00	0.25	0.00	23.29
Thalasseus sandvicensis (Sandwich tern)	113	0.00	0.00	0.00	0.00	1.56
Tringa semipalmata (Willet)	166	4.99	2.26	0.50	2.81	0.91
Zenaida macroura (Mourning dove)	1	0.00	0.00	0.00	0.00	0.01
Unidentified bird	463	6.19	1.32	2.52	2.94	4.76
Total Birds	10,810	9.33	9.85	7.36	7.27	66.93

* Florida state-designated threatened species. ** Federally threatened species.

Table 2: Total counts and percentage of observations for each mammal species per reef type from post-restoration data.

Species	Total Count	Percent of Observations BESE-Shell Mat Reefs	Cement-Jute Tile Reefs	Cement-Jute Ring Reefs	Live Reefs	Dead Reefs
Lontra canadensis (River otter)	3	0.30	0.00	0.00	0.00	0.00
Procyon lotor (Raccoon)	50	3.49	0.38	0.00	0.51	0.10
Unidentified rodent	9	0.10	0.19	0.13	0.13	0.06
Total Mammals	62	62.90	9.68	1.61	8.06	17.74

When assessing faunal families observed on each reef type (Figure 3), no significant differences were found ($p = 0.392$). The NMDS ordination plot shows variation within dead reef sites. The stress value for this ordination plot (8.143) falls between the good and fair ratings using Kruskal's rule of thumb.

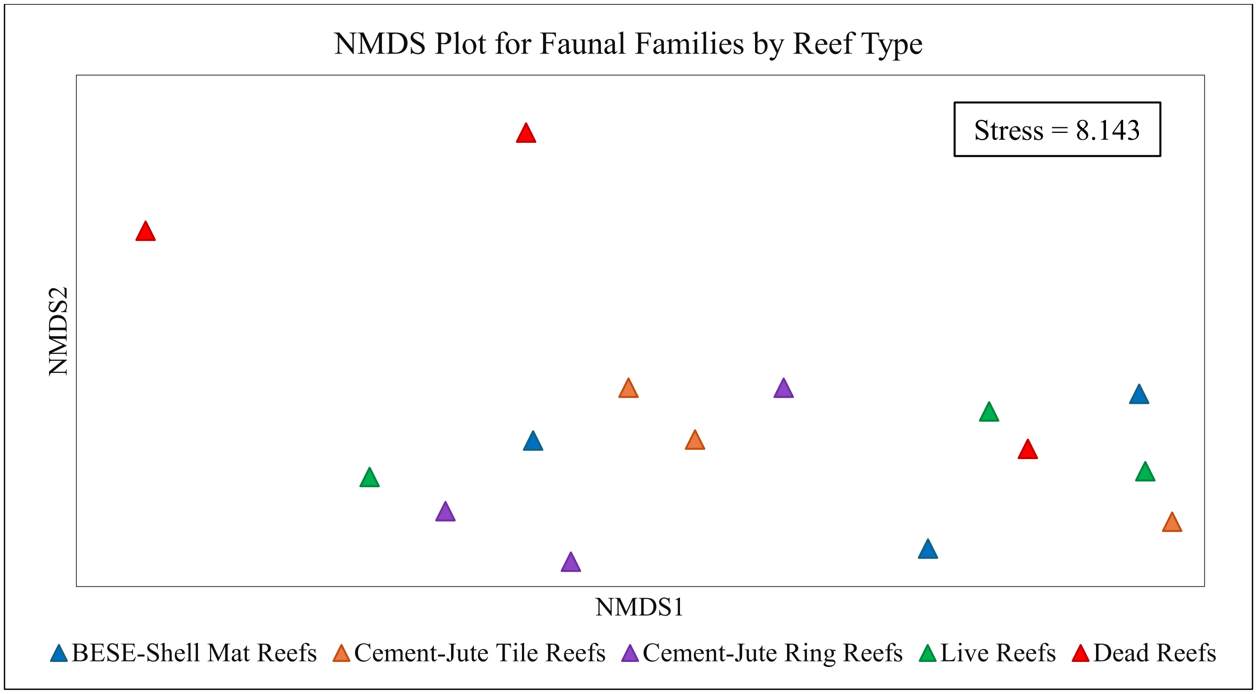

Figure 3: NMDS ordination plot for faunal families.

Species richness (Figure 4) was significantly different between reef types ($p = 0.047$). Pairwise comparisons of species richness determined that cement-jute ring reefs and dead reefs were driving this difference ($p = 0.038$), with richness on dead reefs significantly higher than cement-jute ring reefs. All other reef types were not significantly different from each other.

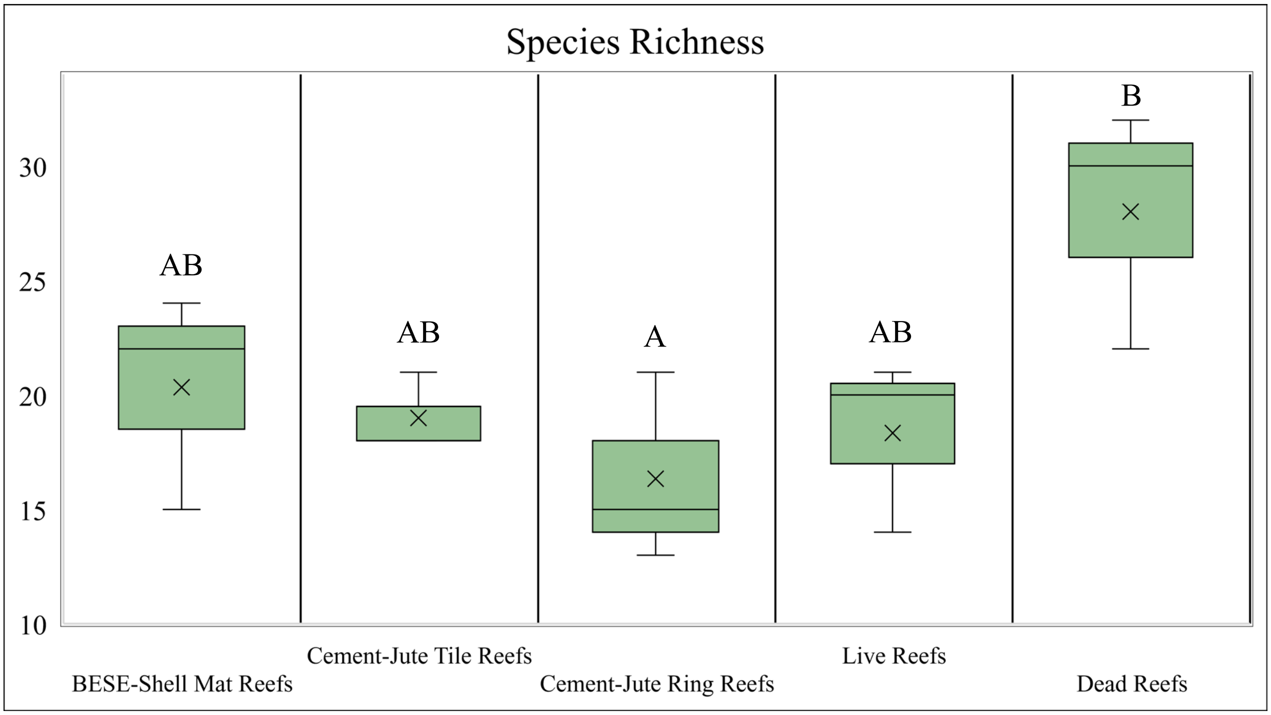

Figure 4: Species Richness by Reef Type.

Repeated measures ANOVAs determined that there was a significant interaction between timeframe and treatment for the frequency of non-foraging behaviors ($p = 0.035$). Frequencies of foraging observations (Figure 5) significantly varied over time but were not significantly impacted by treatment ($p = 0.0001$). These results support the hypotheses faunal diversity and abundance does not differ between reef types and that the counts of vertebrates foraging do not differ between reef types.

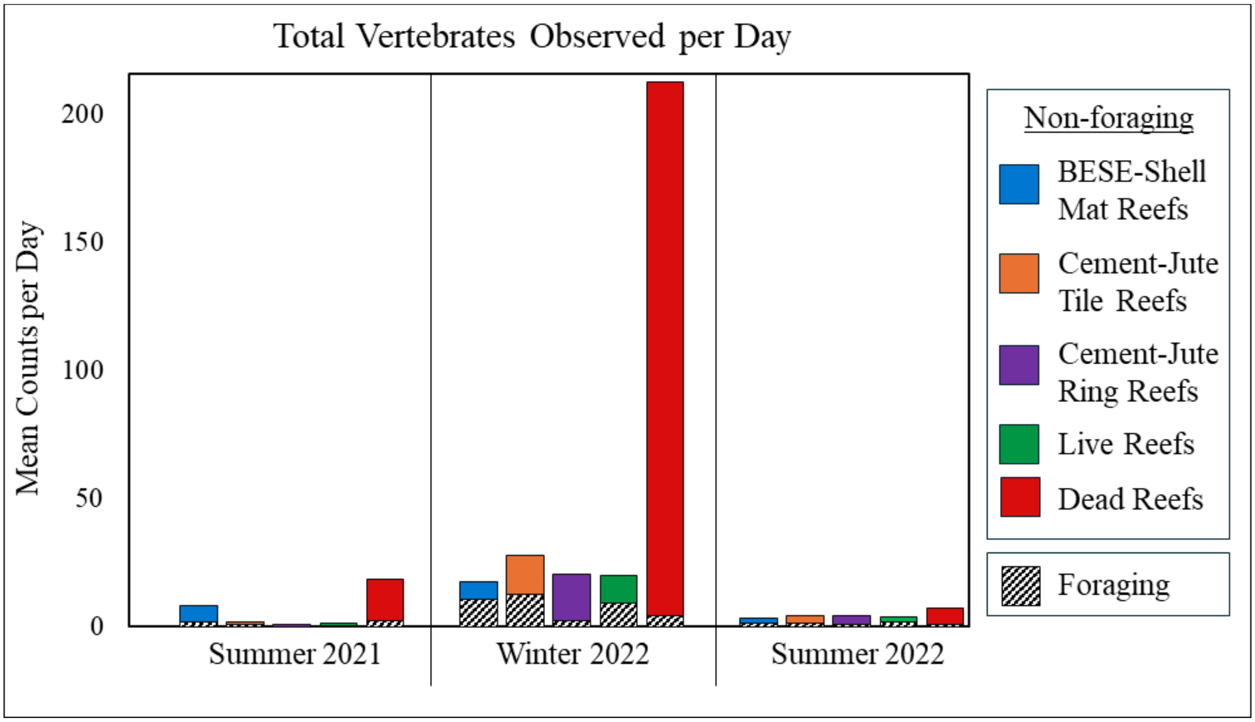

Figure 5: Mean counts of foraging and non-foraging behaviors observed per day.

DISCUSSION

With significant declines in global oyster populations, there is a great need for successful oyster reef restoration methods. Universal oyster metrics (reef dimensions, reef height, oyster density, shell height) and environmental variables (water temperature, salinity, dissolved oxygen) provide useful information on restoration (Baggett et al. 2015). However, as the aim of many restoration projects is to enhance habitat and increase biodiversity, the success of these projects should also be evaluated by monitoring the targeted faunal group or groups (Baggett et al. 2015). Large faunal species such as birds have been used as indicators of restoration success, and examining their behaviors and communities may provide information on the habitat quality of an oyster reef post-restoration (Melvin et al. 1999, Shaffer et al. 2019, Copertino et al. 2022).

Birds are useful indicators of oyster reef restoration success due to their high trophic level (Gregory and Strein 2010). Monitoring bird populations and behavior on oyster reefs provides vital information about the trophic web dynamics as they are sensitive to changes (Melvin et al. 1999). Higher densities of prey items such as invertebrates often support higher densities of birds (Goss-Custard 1996). This study provides vital information about the impacts of biodegradable oyster reef restoration materials on the many faunal species that utilize oyster reefs for foraging and loafing.

Many biodegradable oyster reef restoration materials have been found to be successful in terms of oyster recruitment rates (Sailor-Tynes et al. 2023). Studies have worked to evaluate the biogeochemical properties, durability, and rates of oyster recruitment of biodegradable materials (Nitsch et al. 2021, Walters et al. 2022, Sailor-Tynes et al. 2023). However, as the widescale use of these materials has only been implemented more recently, with many projects less than three years old, continuous monitoring is important (Walters et al. 2022).

Camera traps are a useful method for monitoring faunal behavior and populations as they provide continuous monitoring and work well in areas that are difficult to frequently access (O'Connell et al. 2011, McCallum 2013, Trolliet et al. 2014). Camera traps are a minimally invasive alternative to in-person monitoring. During this study, fauna rarely interacted with the camera traps with only 0.1% (13) of recorded birds and mammals directly contacting the cameras by touching or pecking at them. These camera traps were successful in recording fauna of a variety of sizes and during the night. Only 4.6% (522) of recorded fauna was unidentifiable due to lighting or distance from the camera.

Certain dead reefs with steeper slopes, higher elevations, more vegetative cover, and less live oyster coverage can provide nesting habitat to several bird species (Copertino et al. 2022). *Sternula antillarum* (least tern) and *Haematopus palliates* (American oystercatcher) are Florida state-designated threatened species that have been documented to nest on dead reefs in Mosquito Lagoon (Copertino et al. 2022). Both species were documented during this study with *Haematopus palliates* observed on all reef types except for live reefs and *Sternula antillarum* observed only on one dead reef site. While nesting was not observed in the camera trap video clips, reproductive behaviors including courtship and mating were observed and birds were seen collecting nesting materials from oyster reefs. Circular display flight was observed in one pair of *Ardea alba* (great egret) which is a courtship display (Mock 1980). One *Butorides virescens* (green heron) pair was seen mating. The birds seen collecting nesting materials were all on dead reef sites and included 10 *Zenaida macroura* (mourning dove) individuals, and one *Ardea herodias* (great blue heron).

Overall faunal abundance was greatest during the winter, which was likely due to the many migratory species observed, including *Pelecanus erythrorhynchos* (American white pelican), *Lophodytes cucullatus* (Hooded merganser), and *Mergus serrator* (Red-breasted merganser). Faunal communities were similar on all reefs, however, two of the three dead reefs were spatially separated from the rest of the sites in the NMDS plot. These two dead reef sites had the greatest species abundance, richness, and difference in faunal communities between seasons. During the winter season, these two dead reefs were consistently covered with large flocks of birds. Family *Laridae* (gulls, terns) was most closely associated with one of these dead reefs. Members of this family, such as *Leucophaeus atricilla* (Laughing gull), *Larus delawarensis* (Ring-billed gull), and *Thalasseus maximus* (Royal tern), were commonly observed loafing in large interspecific groups, including other gull and tern species. Families *Scolopacidae* (sandpipers) and *Charadriidae* (plovers) were closely associated with the other dead reef. Dead oyster reefs are approximately one meter higher than the mean high-water level (Wall et al. 2005). This means these reefs are always exposed and available for faunal use, even when live and restored reefs are submerged at high tide. The results of this study indicate that certain dead reefs may provide critical habitat for migratory species. This should be considered by restoration managers when determining which dead reefs should be selected for restoration projects. Future studies are recommended to evaluate specific dead reefs to maintain as habitat for nesting, threatened, and migratory birds.

The results of this study show that faunal diversity, abundance, and behaviors are similar between restored and unrestored oyster reefs in Mosquito Lagoon. This suggests that biodegradable oyster reef restoration materials do not negatively impact faunal communities and may be suitable alternatives for plastic-based restoration materials. Decisions on restoration materials should instead focus on which material will be best suited to the habitat and the local causes of oyster population declines.

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