

THE ROLE OF ARTIFICIAL REEFS AND NATURAL BANKS IN THE DISTRIBUTION
AND ABUNDANCE OF ECONOMICALLY IMPORTANT FISHES

A Dissertation

by

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KELSEY LYNN MARTIN

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ABSTRACT

In the northern Gulf of Mexico, artificial reefs have become an important resource enhancement tool. These structures, along with natural banks, act as fish aggregation sites on a relatively barren continental shelf region that contains little to no vertical structure for fishes to seek refuge. Generally, a lack of fishery-independent studies comparing artificial reefs to their natural counterparts has led to uncertainty surrounding the role of artificial reefs. Furthermore, with increasing legislative regulations on the types of materials permitted for artificial reefs, it is important to examine which materials provide the most benefit. Thus, the goal of this dissertation was to present stakeholders and managers with the tools to understand how artificial reef design, placement, and proximity to other reefs, which are difficult to ascertain, affect population dynamics to guide future reefing projects.

In Chapter 2, I used a combination of remotely operated vehicle and hydroacoustic surveys to characterize fish communities at artificial reefs and natural banks and provide absolute abundance estimates of four economically important fish species: Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper. My results show high abundance of reef fish, particularly Red Snapper, Vermilion Snapper, and Greater Amberjack. Additionally, age 2+ Greater Amberjack extrapolated estimates were 221% of the most recent stock assessment for this species and identify a need for reevaluation of the habitat use and abundance estimates for this species.

In Chapter 3, I conducted surveys using stereo-remote underwater video surveys to address the effect of prefabricated pyramid reefing location and orientation on fish community structure. Although reefing location nor orientation had a significant effect on species richness, diversity, and evenness, I identified differing communities between both variables (reefing

location and orientation) with Red Snapper, Greater Amberjack and Gray Triggerfish among the main contributing species to these differences. Fish measurements identified primarily juvenile individuals which emphasizes the importance of lower relief reefs as habitat for younger cohorts. Absolute abundance estimates were derived from combining the video surveys and hydroacoustic surveys and were used to calculate two types of cost-benefit ratios based on overall fish density and the densities of economically important species such as Red Snapper, Greater Amberjack, Gray Triggerfish, Vermilion Snapper, and Almaco Jack. Port O'Connor nearshore reef was considered the least cost-effective reefing location in terms of overall fish density, but not for economically important species density. Port O'Connor was configured with science in mind and produced the most consistent results compared to the other reefing locations, which highlights the importance of developing a reef with science in mind to evaluate reef benefits more competently.

In Chapter 4, I used the same methods from Chapter 2 to address the effect of artificial reefing material (prefabricated pyramids and railroad ties) on fish community structure and cost-benefit. Species richness was found to increase with increasing tonnage of railroad ties and is likely due to differences in structural complexity. In contrast, evaluation of the two different types of cost-benefit ratios identified a massive 4000-ton railroad tie pile as the least cost-effective in terms of overall fish density and economically important species density. As a result, I do not recommend considerably large piles of reefing material if the goal of the reef is to provide habitat for economically important species, but rather to spread this material in smaller and widely dispersed groupings. Importantly, this study identifies the need for specifying clear objectives when designing artificial reefs to maximize efficiency.

In Chapter 5, I developed an individual-based model to evaluate Red Snapper, Greater Amberjack, and four competing species, population responses to changing artificial reefs in more complex, but realistic, habitat scenarios like those observed on the Texas continental shelf. My results identify the importance of artificial reefs in the growth and overall biomass of Red Snapper and Greater Amberjack, particularly in areas, like the Gulf of Mexico, where structured habitats are limited. This model also validated age- or size-based transitions in habitat for Red Snapper that have been observed in other studies, suggesting a transition from habitat-centric to forage-centric movement for this species. This model presents resource managers and stakeholders with a tool to guide future reefing projects to discern the impact of placement and density of artificial reefs on population dynamics of economically important fish species.

Collectively, my findings suggest that artificial reefs in the western Gulf of Mexico play an important role in shaping fish communities and in the population dynamics of economically important species. Furthermore, I developed a unique approach to quantifying the efficacy of artificial reefs and revealed two types of reefs that were significantly less cost-effective than the others explored in this study. Future artificial reefing projects should avoid large, highly dense reefs and should consider proximity to other artificial or natural habitats. I also present a tool for resource managers and stakeholders to guide future reefing projects to discern the impact of differing artificial reef designs on populations of economically important species. This dissertation improves the scientific understanding of the role of natural and artificial habitat in fish communities and provides fisheries managers with key abundance estimates needed for stock assessment models for several economically important species on the Texas coast.

DEDICATION

In dedication to my husband, whose faith and support in me never waivers, and to my mom, dad, brother, and sister, for igniting my love of the ocean and encouraging me to pursue my dreams.

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CHAPTER I: INTRODUCTION

Gulf of Mexico Habitats

Habitat plays an important role in governing fish community distribution and abundance (Shumway et al. 2007, Kovalenko et al. 2012, Gregor & Anderson 2016). Habitats provide an opportunity for increased survival and are distinguished by factors such as predation risk, competition, growth, and reproduction (Steele 1998, Chalfoun & Martin 2009, Gregor & Anderson 2012). Fish habitat preference is a series of trade-offs and interactions among each of these factors (Steele 1998, Chalfoun & Martin 2009). For example, habitats optimize reproduction and minimize predation risk, while increasing competition (Lindberg et al. 1990, Steele 1998, Almany 2004, Keller 2019). Each factor, particularly predation risk, is often associated with habitat complexity, where more structurally complex habitats provide the most refuge for prey species and reduce predator foraging efficiency (Crowder & Cooper 1982, Ryer et al. 2004, Scharf et al. 2006, Trussell et al. 2006, Grabowski et al. 2008, Kolar et al. 2019).

Habitat complexity, or the heterogeneous arrangement of physical structure in the habitat, can be difficult to quantify, but is thought to affect species richness, diversity, and abundance of fish communities (MacArthur & MacArthur 1961, Luckhurst & Luckhurst 1978, Menge et al. 1985, Dean & Connell 1987, Bell et al. 1991, Shumway et al. 2007, Kovalenko et al. 2012, to name a few). Typically, more complex habitats harbor greater species diversity and abundance than less complex habitats (Potts & Hurlbert 1994, Spieler et al. 2001, Lingo & Szedlmayer 2006, Broughton 2012, Ajemian et al. 2015a,b, Streich et al. 2017a). Therefore, habitats that differ in complexity, often harbor different fish communities (Wilson et al. 2003, Streich et al. 2017a, Downey et al. 2018).

Two types of habitats that vary greatly in complexity are natural and artificial reefs. On the ~59,000 km² of the Texas continental shelf, there are myriad of natural and artificial structures where fishes are known to aggregate (Bohnsack & Sutherland 1985, Dennis & Bright 1988). Natural reefs in the Gulf of Mexico (GOM), which can be categorized as either rock, coral, or sponge habitat, have long been considered to act as fish aggregation sites (Carson 1944). They typically harbor extensive reef fish, and even tropical reef biota (Bright 1977, Dennis & Bright 1988). Natural reefs on high relief banks occur in the western GOM regions as conspicuous areas among a vast expanse of continental shelf sediment characterized by sand and mud bottom. These structures provide complex habitat in an otherwise featureless bottom that are thought to attract fish communities by providing refuge and escape routes from predators (Lingo & Szedlmayer 2006).

Unlike natural reefs, artificial reefs are manmade structures, including permitted artificial reefs, oil and gas platforms, shipwrecks/obstructions, and other anthropogenic materials of a wide variety of composition (e.g., tires, reef balls, bridge spans, military equipment). Prior to the 21st century, relatively little was known about the biology and ecology of artificial reef faunal communities (Bohnsack & Sutherland 1985). Today, artificial reefs are recognized to serve as essential fish habitat (Bohnsack & Sutherland 1985, Shipp & Bortone 2009, Ajemian et al. 2015a,b), are important for fisheries management (Patterson et al. 2009), and relieve the restraint on population abundance with additional habitats (Bortone 2008) that can also facilitate range expansion (Rooker et al. 1997, Love & York 2005, Broughton 2012). Current research focuses on the faunal production of artificial reefs compared to natural reefs (Wilson et al. 2003, Streich et al. 2017a,b, Downey et al. 2018). Artificial reef construction in the northern GOM has been shown to increase recruitment of certain species of reef fish, including economically important

Red Snapper (*Lutjanus campechanus*; Streich et al. 2017a) indicating that artificial reefs and offshore standing platforms serve as viable habitat for these species, and likely other economically important fishes.

Offshore oil and gas platforms have been described as potential stepping-stones for the expansion of various marine fish communities (Broughton 2012). Of the ~7,000 oil and gas platforms that have been installed in the GOM, about 1,600 are still in use and 573 have been converted into artificial reefs as part of the Bureau of Safety and Environmental Enforcement “Rigs to Reefs” program (www.bsee.gov; accessed 5 June 2022). In the northern GOM, oil and gas platforms support the majority of the United States Red Snapper harvest (Shipp & Bortone 2009) and provide habitat for reef fish species in areas where they are not normally known to occur (Rooker et al. 1997, Love & York 2005). Although artificial reefs offer new habitat for these economically important fish species, some studies suggest that they may not be positively contributing to fisheries production (Strelcheck et al. 2007, Patterson et al. 2009, Broughton 2012) even though artificial reefs tend to have higher densities of fish relative to natural bottom (Claisse et al. 2014, Ajemian et al. 2015a, Roa-Ureta et al. 2019). Artificial reefs that produce significant new biomass have the potential to increase fishery resources (Bohnsack 1989); however, artificial reefs that act primarily by attraction may increase catchability; and thus, promote overfishing (Bohnsack 1989). Additionally, sites that aggregate species can result in hyperstable abundance estimates that confound true population trends because catch-per-unit-effort (CPUE) can be maintained artificially high at those locations (i.e., population trend may be decreasing but the CPUE trend remains unchanged because fish are aggregating). In contrast, they can also be used to redirect fishing pressure away from natural banks that contain species (e.g., many coral species) that are more sensitive to anthropogenic disturbances. While the

debate has largely recognized that both attraction and production occur in positive synergistic ways, the debate has waned in the most recent decade. However, more data are needed to characterize artificial reefs more effectively for fisheries management, particularly in terms of estimates of absolute abundance. Absolute abundance estimates can enhance the scientific understanding of the role of artificial reefs and the positive or negative effects of their presence. Moreover, both management efforts and artificial reef design need to be addressed to properly manage fish using these habitats (Pickering & Whitmarsh 1997, Broughton 2012).

Characterizing Fish Communities

Artificial reefs and natural banks are commonly exploited fishing sites in the northern GOM (Ditton et al. 1978, Beck & Odaya 2001, Simard et al. 2016). Recreational and commercial fishers in this region are notorious for exploiting known productive fishing sites, which can lead to localized depletion or overexploitation of certain sites (Bradley & Bryan 1975). Effective fisheries management relies on characterizing both the direct and indirect effects of intensive fishing, and the first step is a firm understanding of communities and their population dynamics at these habitats.

To characterize fish communities, several non-extractive approaches have been employed at artificial reefs and natural banks, each with their own strengths and weaknesses. Perhaps one of the simplest and most common methods is underwater visual census (UVC) by SCUBA divers (Estes & Gilbert 1978, Brock 1982, Marsh & Sinclair 1989, Thanopoulou et al. 2018). This method is cost effective, relatively quick, non-destructive, and repeatable (St. John et al. 1990, English et al. 1994, Watson et al. 1995, Thompson & Mapstone 1997, Harvey et al. 2004, Thanopoulou et al. 2018), but is often limited by several biases (Brock 1982, Bohnsack & Bannerot 1986, Thresher & Gunn 1986, Kulbicki 1998, Harvey et al. 2004, 2007, Villon et al.

2018). For example, visual census using divers is reasonable for diurnally active species, but generally underestimates the most common species and typically misses or underestimates the presence of cryptic fish species (Brock 1982). Additionally, the SCUBA diver must decide whether a fish is inside or outside of the sampling unit (Harvey et al. 2004). This leads to biases in the count of fishes within the sample unit and in many cases results in survey designs that do not encompass entire spatial distributions of fish (Bohnsack & Bannerot 1986, Thresher & Gunn 1986, Harvey et al. 2004). The presence of the diver also presents certain biases in fish behavior, which have been shown to vary between sites (Kulbicki 1998). Diver avoidance behaviors have been shown to be more common at highly disturbed areas (where divers frequently chased fish through spearfishing or netting) and may draw erroneous conclusions if comparing marine protected areas with fished areas (Kulbicki 1998). Along with sampling biases, human-based techniques are limited by depth and duration at which a diver can safely operate (Harvey et al. 2007, Villon et al. 2018).

Due to the biases and limitations of using SCUBA divers, video-based surveys have been increasingly employed over the last few decades to characterize fish communities (Ellis & DeMartini 1995, Cappo et al. 2003, Langlois et al. 2010, Mallet & Pelletier 2014, Villon et al. 2018). Video-based methods provide similar estimates of fish abundance and species diversity as traditional UVC methods using SCUBA (Pelletier et al. 2011), while also overcoming the depth or time limitations of human-based surveys (Villon et al. 2018). Video-based methods also provide a permanent record of surveys so that data collection may be impartial, repeatable, and standardized (Cappo et al. 2006). Several video-based methods include, but are not limited to, remotely operated vehicle (ROV) and remote underwater video systems (RUVS). Due to recent reductions in cost and size and increases in maneuverability, ROVs have become more

commonly used by fisheries researchers (Ajemian et al. 2015b). However, more mobile or cryptic species are often missed in ROV surveys and can even cause adverse behavioral effects (e.g., avoidance or attraction) outside the field-of-view of the camera (Stoner et al. 2008). Additional survey biases can occur with the presence of the vehicle, the vehicle motion and water displacement, the speed, the sounds produced by the motors, and electrical or magnetic fields produced by the ROV that may be detectable at close range (Stoner et al. 2008). Although attempts have been made to calibrate ROV surveys for behavioral responses of fish to ROVs, species-specific interactions are difficult to capture, resulting in biased density estimates (Campbell et al. 2021). As a result, more studies are turning to RUVS (Barans et al. 2005, Cappo et al. 2006, Brooks et al. 2011, Bond et al. 2018). Apart from being more time- and cost-efficient than ROVs, RUVS do not cause major disturbances to the community being sampled and observe individuals that are likely to avoid SCUBA divers or ROVs (Cappo et al. 2004, Cappo et al. 2006, Harvey et al. 2007, Bond et al. 2018). However, RUVS are also likely to miss more cryptic species despite being less invasive.

While video-based methods are ideal for evaluating relative abundance and fish community composition, these methods only provide snapshots of the total abundance of the reef and are limited by water clarity presenting a major challenge when sampling in turbid or low visibility waters. As such, hydroacoustics has become a useful tool for precise and accurate estimates of absolute abundance and size distribution, is not limited by poor visibility, and has a greater range than visual methods (Stanley & Wilson 1996). Hydroacoustics can also accurately estimate fish school density (Johnson et al. 2019) and can be used to investigate fish density distributions on large spatial scales providing a more holistic view of community health (Zenone et al. 2017). However, hydroacoustics alone cannot obtain species-specific abundance estimates

and must be used in combination with visual census methods, such as ROV or RUVS surveys. If applied correctly, absolute abundance estimates of economically important species can be obtained from acoustic profiles associated with known species composition. Taken together, these methods can inform stakeholders of the economic benefits from natural habitats compared to their artificial reefs.

Economically Important Species

Perhaps the most sought after and economically important offshore finfish in the Gulf of Mexico is Red Snapper (Shipp & Bortone 2009, Brulé et al. 2010, Keithly Jr & Roberts 2017). Ubiquitous on natural and artificial reefs, Red Snapper support healthy commercial and recreational fisheries (Patterson et al. 2014, Karnauskas et al. 2016, Streich et al. 2017a, Garner et al. 2019). Recent controversy surrounding population estimates of Red Snapper has led to discord between management agencies and fishers (Cowan 2011, Cowan et al. 2011, Simmons et al. 2019, Scyphers et al. 2021, Stunz et al. 2021). To summarize, available data on the status of the Red Snapper stock has led to proposed underestimates of the population; meanwhile, high CPUE suggests to fishers that the population is healthier than the stock assessments indicate (Simmons et al. 2019, Scyphers et al. 2021, Stunz et al. 2021). Because Red Snapper stocks are rebuilding in the Gulf of Mexico (SEDAR 2018), population monitoring is critical for evaluating the status of the stock (Cowan 2011, Stunz et al. 2021).

Other economically important species in the Gulf of Mexico include Vermilion Snapper (*Rhomboplites aurorubens*), Greater Amberjack (*Seriola dumerili*), and Gray Triggerfish (*Balistes capriscus*). Behind Red Snapper, Vermilion Snapper are one of the most targeted species by commercial fishers (Keithly Jr & Roberts 2017). However, despite their significance, population estimate models have historically lacked data describing populations in the

northwestern GOM (Johnson et al. 2010). Greater Amberjack and Gray Triggerfish are also particularly important recreational fishery species (Goodyear & Thompson 1993, Harper & McClellan 1997, Manooch & Potts 1997). They are also both classified as rebuilding, yet evidence of recovery is uncertain due to a lack of reliable data (SEDAR 2013, 2020). In fact, the most recent stock assessment for Gray Triggerfish is delayed after peer reviewers identified concerns over essential data inputs for age and growth (SEDAR 62).

A substantial increase in the number of dedicated artificial reefs in the GOM over the last 50 years, has corresponded to major shifts in harvest locations and areas of economically important species concentrations (Shipp & Bortone 2009). Artificial reef construction has been shown to increase recruitment of certain reef species, including Red Snapper (Streich et al. 2017a). Similar to Red Snapper, Vermilion Snapper, Greater Amberjack, and Gray Triggerfish use artificial reefs as habitat during various stages in their life histories (Manooch & Potts 1997, Simmons 2008, Gallaway et al. 2009, Gallaway et al. 2021). With recent advances in wind energy, fisheries managers must address how offshore wind farms will influence the population dynamics of these economically important fish species. Because wind farms are typically placed on flat soft-bottom habitat, these artificial structures will present new structured habitat for fish in an otherwise structureless region (Langhamer & Wilhelmsson 2009, Glarou et al. 2020). Until then, artificial reefing programs have been introduced and proposed as a method to help maintain stable harvestable populations of economically important fish species with the rapid decommissioning of oil and gas platforms. The Artificial Reef Act of 1989 provided guidance for the development of cost-effective artificial reefs. One of the recommendations of this Act states to arrange materials in as complex a fashion to not jeopardize the structural integrity, but to maximize effectiveness, durability, and stability (Stephan et al. 1990). Due to differing

limitations in fish productivity or recruitment by region, artificial reefs should be designed to maximize abundance based on the needs of the region (Grossman et al. 1997, Strelchek et al. 2005). Furthermore, the materials used can also play a role in the overall effectiveness of the reef. Concrete materials, like prefabricated pyramids, that have biomimetic properties, increase the benefits of reefs to the local environment (Vivier et al. 2021). With increasing legislative regulations on the types of materials permitted for artificial reefs, it is important to examine which materials provide the most benefit (Becker et al. 2018). When the ecological benefit is balanced with the cost associated with artificial reef implementation, return on investments in the project can be maximized (Adams et al. 2006, Graham et al. 2016).

Evaluating the Cost-Benefit of Artificial Reefs

Often not documented nor characterized, ecological or human benefits are the ultimate objective of habitat enhancement (Grove & Wilson 1994). Typically, a successful reef is one that is useful to people (Milon et al. 2000) with a high return on investment. Prior to 2000, a relatively small proportion of published studies documented the human benefits of habitat enhancements such as artificial reefs (Seaman Jr. & Jensen 2000). After 2000, numerous studies focused on evaluating the benefits of artificial reefs (Black 2001, Pitcher et al. 2002, Stanley & Scarborough 2004, Adams et al. 2006, Whitmarsh et al. 2008, Chen et al. 2013, Polak & Shashar 2013, Schuett et al. 2016, to name a few); however, many of these studies fail to address how the ecological benefit(s) contributes to the economy of local communities. Quantifying the economic importance of an artificial reef may help justify future public artificial reef development projects and assist in developing adaptive strategies associated with reef deployment (Adams et al. 2006).

One way in which artificial reef efficacy can be studied is with individual-based models (IBMs). Individual-based models simulate populations and communities by following individuals and their properties (DeAngelis & Grimm 2014). Individual-based models also allow researchers to study how system-level properties affect the behavior of individuals from multiple species (Campbell et al. 2011, Grüss et al. 2016, Campbell et al. 2018). This method of study is particularly useful for fisheries managers (Boyd et al. 2018). Unlike traditional empirical models, IBMs are bottom-up models in which population-level behaviors emerge from the interactions of autonomous individuals and their environment (DeAngelis & Grimm 2014). Aspects such as variability among individuals, local interactions, complete life cycles, and individual behavior adapting to their changing internal and external environments are usually ignored in analytical models (Grimm et al. 2006). Thus, IBMs are thought to better simulate an individual's natural responses to changing scenarios than analytical models (Boyd et al. 2018).

Recent examples of IBMs include PISCATOR and OSMOSE (object-oriented simulator of marine ecosystems exploitation), which were developed to simulate fish community dynamics and trophic interactions (van Nes et al. 1996, Shin & Cury 2001, 2004). Other IBMs have been developed to simulate fish community responses to various environmental or anthropogenic stressors (Rose et al. 1996, McDermot & Rose 2000, Rose et al. 2009, Campbell et al. 2011). One such study developed an IBM to address fish community responses to various artificial reef configurations (Campbell et al. 2011). This novel study integrated movement, bioenergetics, reproduction, and mortality of Red Snapper, and two competitor species, to evaluate their response to changes in oil and gas platform (henceforth, rig) design (Campbell et al. 2011). As a result, IBMs present stakeholders and managers with the tools to understand how artificial reef placement and proximity to other reefs, which are difficult to ascertain, affect population

dynamics to guide future reefing projects (Charbonnel et al. 2002, Strelchek et al. 2005, Campbell et al. 2011, Vivier et al. 2021).

Dissertation Purpose and Chapter Outline

The purpose of this dissertation was to provide fundamental insight and characterize habitat use of both natural and artificial structures in the GOM by fish communities and several economically important fishes. These data not only provide valuable insight on the population dynamics of reef-associated species, but they also inform managers of both the direct and indirect effects of exploited fish aggregation sites. This dissertation also focused on developing a cost-benefit analysis to inform stakeholders on the most ecological and economically beneficial artificial designs along the Texas coast. This dissertation therefore enhances the scientific understanding of the relationship among reef complexity (e.g., density, orientation, and material) and fish community structure complexity (e.g., species richness, diversity, and abundance) at commonly exploited fishing sites.

In this dissertation, I present results from four unique studies designed to address the role of artificial reefs and natural banks in shaping reef fish communities with a focus on economically important fishes:

- (1) Chapter 2 – I address fish community differences between artificial reefs and natural banks. I also incorporate population estimates for Red Snapper, Greater Amberjack, Vermilion Snapper, and Gray Triggerfish at Texas artificial reefs and natural banks.
- (2) Chapter 3 – I evaluate fish communities at three nearshore prefabricated pyramid reefs and quantify their efficacy through a cost-benefit analysis.
- (3) Chapter 4 – I examine fish communities among different reefing materials and quantities and evaluate their efficacy using the same methods from Chapter 2.

(4) Chapter 5 – I built upon an existing IBM from Campbell et al. (2011) to evaluate Red Snapper, Greater Amberjack, and competing species, population responses to changing artificial reefs in more complex, but realistic, habitat scenarios like those observed on the Texas continental shelf.

I conclude with a summary of key findings of each chapter followed by a discussion of potential reefing recommendations and future directions for research.

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CHAPTER II:
FISH COMMUNITY DIFFERENCES AMONG TEXAS ARTIFICIAL REEFS AND
NATURAL BANKS: ABUNDANCE ESTIMATES AND MANAGEMENT IMPLICATIONS
FOR FOUR ECONOMICALLY IMPORTANT SPECIES

Abstract

Artificial reefs and natural banks occur in the western Gulf of Mexico as conspicuous areas among a vast expanse of continental shelf that is dominated by soft sediments such as clay, silt, and sand. These habitats act as predictable aggregation sites for fish and are commonly exploited habitat types by various fisheries. I used a combination of remotely operated vehicle surveys and hydroacoustics to characterize fish communities at artificial reefs and natural banks and provide absolute abundance estimates of four economically important fish species: Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper. I identified differing patterns of fish communities between artificial reefs and natural banks and among three depth strata. Furthermore, although artificial reefs were more speciose and contained higher densities, natural banks contributed more to the overall abundance. My results show high abundance of reef fish, particularly Red Snapper, Vermilion Snapper, and Greater Amberjack. Additionally, age 2+ Greater Amberjack extrapolated estimates were 221% of the most recent stock assessment for this species and identify a need for reevaluation of the habitat use and abundance estimates for this species. This study improves the scientific understanding of the role of these habitats in fish communities and provides fisheries managers with abundance estimates for stock assessments.

Introduction

Artificial reefs and natural banks act as aggregation sites for fish and other marine species (Dennis & Bright 1988, Rooker et al. 1997, Hobday & Campbell 2009) and can enhance natural resources, particularly in areas where they are limited (Streich et al. 2017a). Natural high relief banks typically occur in the western Gulf of Mexico (wGOM) region as conspicuous areas among a vast expanse of continental shelf that is dominated by soft sediments such as clay, silt, and sand (Parker et al. 1983, Rezak et al. 1985). In these areas, both natural and artificial reefs provide complex habitat within adjacent low-relief areas, and are thought to attract fish communities by providing refuge from predators (Lingo & Szedlmayer 2006), acting as food reservoirs (Bohnsack & Sutherland 1985, Langhammer 2012), and supporting other ecological functions such as spawning (McLean et al. 2015, Schwartzkopf & Cowan 2017, Downey et al. 2018). As a result, these sites serve as important fish habitat for resident and transient fish communities and have become an integral component of the Gulfs' fisheries (Bohnsack & Sutherland 1985, Levin & Stunz 2005, Streich et al. 2017a,b).

Artificial reefs and natural banks are commonly exploited fishing sites in the northern GOM (Beck & Odaya 2001, Simard et al. 2016). Recreational and commercial fishers in this region are well-known for exploiting known productive fishing sites (Bradley & Bryan 1975), indicating that overexploitation of certain sites is common, particularly those that are well-known and close to ports. Artificial reefs that act primarily by attraction may increase catchability and thus promote overfishing (Bohnsack 1989); however, artificial reefs that produce significant new biomass have the potential to increase fishery resources (Bohnsack 1989). Recent recommendations suggest that managers consider both of these factors when

designing and using artificial reefs as a management tool (Campbell et al. 2011, Streich et al. 2017b, Stunz et al. 2021).

Offshore oil and gas platforms have been described as potential stepping-stones for the expansion of various marine fish communities both desirable and undesirable (e.g., Lionfish; Broughton 2012). Of the ~7,000 oil and gas platforms that have been installed in the GOM, about 1,600 are active and 573 have been converted into artificial reefs as part of the Bureau of Safety and Environmental Enforcement “Rigs to Reefs” program (www.bsee.gov; accessed 5 June 2022). Platforms that have been designated for the RTR program can either be towed to permitted reefing areas, toppled in place, or partially removed, where the top portion of the structure is severed at a permitted navigational depth (e.g., 25 m). In the northern GOM, both active and reefed platforms support much of the United States Red Snapper harvest (Shipp & Bortone 2009) and provide habitat for reef fish species in areas where they are not normally known to occur (Rooker et al. 1997). Because of their rapid removal and only a proportion of these decommissioned platforms are candidates for reefing, it is important to determine the effect their removal will have on regional fisheries (see Gallaway et al. 2021).

Generally, a lack of fishery-independent studies comparing artificial reefs to their natural counterparts has led to uncertainty surrounding the role of artificial reefs (Streich et al. 2017b). Detailed comparisons of reef-associated fish populations and communities at natural and artificial reefs is the first step towards understanding the potential benefits or drawbacks of artificial reefs (Carr & Hixon 1997). Many techniques, including underwater visual census (UVC) surveys, remotely operated vehicle (ROV), and hydroacoustics can be used to sample fish communities at artificial reefs and natural banks. Traditionally, UVC surveys using SCUBA has been one of the simplest and most cost-effective methods of sampling fish communities (Estes &

Gilbert 1978, Marsh & Sinclair 1989, Thanopoulou et al. 2018). However, in the western GOM, the depth of most sites is generally too deep for SCUBA divers to operate for an extended period, thus other methods need to be explored to capture representative data across spatial domains occupied by species of interest.

Video-based methods provide similar estimates of fish abundance and species diversity as traditional UVC methods using SCUBA (Pelletier et al. 2011), while also overcoming the depth or time limitations of human-based surveys (Villon et al. 2018). Video-based methods also provide a permanent record of surveys so that data collection may be impartial, repeatable, and standardized (Cappo et al. 2006). Due to recent reductions in cost and size and increases in maneuverability, ROVs have become more commonly used by fisheries researchers (Ajemian et al. 2015b). However, ROV surveys only provide snapshots of abundance that are often used to calculate relative abundance. To derive total species-specific abundance estimates, a method designed for recording absolute abundance such as hydroacoustic surveys should be explored in this region. Hydroacoustic surveys provide precise and accurate estimates of absolute abundance and size distribution, is not limited by poor visibility, and has a greater range than visual methods (Stanley & Wilson 1996). Hydroacoustics can also accurately estimate fish school density (Johnson et al. 2019) and can be used to investigate fish density distributions on large spatial scales providing a more holistic view of community health (Zenone et al. 2017).

In this study, I present a comparative study of reef fish community structure (e.g., species richness, diversity, and abundance) between artificial reefs and natural banks, with special emphasis on the distribution of four economically important fish species in the western GOM. Specifically, this study focused on three main questions: 1) Do artificial reefs support a more robust fish community in terms of species richness and diversity than natural banks? 2) Does

depth play a role in community differences (if any)? 3) Do artificial reefs harbor greater total abundances of four economically important fish species: Red Snapper (*Lutjanus campechanus*), Greater Amberjack (*Seriola dumerili*), Gray Triggerfish (*Balistes capriscus*), and Vermilion Snapper (*Rhomboplites aurorubens*)? My research focused on these four species given their importance to fisheries in the region.

Methods

Study Area

The ~59,000 km² Texas continental shelf can be characterized as a gently sloping surface from the shoreface to the shelf break and predominantly consists of fine sands and shelf mud substrata (Flint & Rabalais 1980, Shideler 1981). Myriad natural and manmade structures interrupt this largely featureless landscape. A nepheloid layer, up to 35 m thick, commonly persist at each of these habitats (Shideler 1981, Dennis & Bright 1988). Suspended sediments impair visibility and affect the community ecology at inhabited reefs (Dennis & Bright 1988, Rezak et al. 1990).

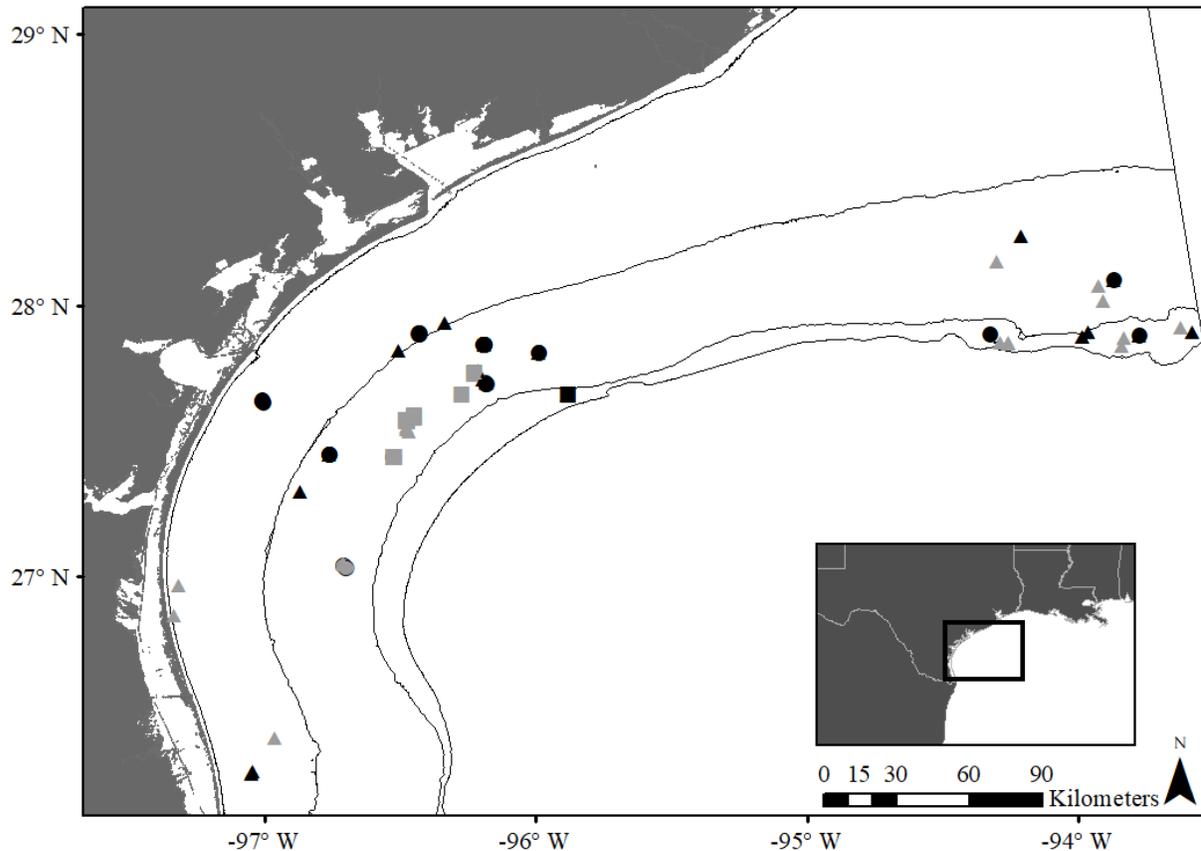


Figure 2-1. ROV and hydroacoustic sampling locations at artificial reefs (black symbols) and natural banks (grey symbols) in the northwestern Gulf of Mexico across three depth strata (10-40 m, 40-100 m, and 100-150 m). *Squares* indicate locations only sampled using ROV, *circles* were only sampled using hydroacoustic surveys, and *triangles* indicate locations that were sampled using both survey methods.

ROV Sampling

Thirty-nine natural bank and 18 artificial reef surveys were conducted in three different depth strata (0-40 m, 40-100 m, 100-150 m), which will henceforth be referred to as shallow, mid, and deep depth zones, along the Texas continental shelf during the summer and fall of 2018 and 2019 (Figure 2-1; Table 2-1). Sampling sites were randomly selected following a stratified random design (habitat and depth zone) based on habitat availability (i.e., areal extent for natural banks and number of structures for artificial reefs) within each depth zone. Habitat-specific sample sizes are reflective of the proportion of those habitats in each depth zone on the Texas

continental shelf (Table 2-1). Artificial reef sites included standing and reefed offshore oil and gas platforms as well as sites composed of multiple structures and varying materials ranging from prefabricated pyramids and concrete box culverts to sunken ships.

Table 2-1. Remotely operated vehicle sampling effort at artificial reefs and natural banks across three depth strata: shallow (10 to 40 m), mid (40 to 100 m), and deep (100 to 150 m).

Depth Zone	Artificial	Natural	Total
Shallow	6	6	12
Mid	8	30	38
Deep	4	3	7
Total	18	39	57

Video-based surveys were conducted using the Mission Specialist Defender ROV (VideoRay, LLC). A rear-facing GoPro Hero7 Black (122.6° horizontal viewing angle, 94.4° vertical viewing angle) was mounted externally to the ROV to increase visual area surveyed and to account for behavioral responses to the ROV (i.e., avoidance or following behaviors). Natural bank ROV sampling consisted of three, independent 40-m transects 1.5 m above the seafloor that minimized double sampling of the same area on the reef crest, reef slope, and base of the reef. Transects were independent due to differing start and stop points (i.e., stop point of first transect differed from the start point of the second transect). Remotely operated vehicle sampling methods at artificial reefs were similar to those outlined in Streich et al. (2017b). The ROV began the survey with a depth interval transect (DIT), starting at the top of the structure and stopping at 10-m intervals for 1 min until reaching bottom depth or a depth where visual surveys were impaired. Following the DIT, three horizontal transects, 30-40 m in length and 1-2 m from the structure, were completed along different horizontal portions of the structure. All ROV surveys were conducted at a constant speed (0.5 m/s) with a constant camera viewing angle and tilt.

Species Composition Analysis

I used a minimum count method (MaxN; Ellis & DeMartini 1995) to enumerate species observed at each site. MaxN is a conservative estimate that enumerates the maximum number of individuals, of a particular species, seen in a single frame throughout the deployment (Ellis & DeMartini 1995). All ROV video analysis occurred in the lab with a minimum of two readers recording species, count, time, and temperature (acquired from the ROV). If the two readers disagreed on a species identification, a third reader was included. Counts were jointly reviewed if readers differed by more than 5%. The count was then averaged between the two readers for each species.

I investigated species richness and diversity patterns using traditional diversity measures such as Shannon diversity index (H') and Pielou's evenness index (J'). I used a two-way analysis of variance (ANOVA; $\alpha = 0.05$) to test for potential differences in species richness, H' , and J' (Ajemian et al. 2015a) between habitat type (artificial reef or natural bank) and among depth zones. I examined variation in fish community structure across and within habitat types based on a Bray-Curtis similarity matrix of fish abundance using protocol outlined in Peters et al. (2015). The square-root of abundance (to limit the impact of overly abundant species) was analyzed with an analysis of similarity (ANOSIM) to determine if community structure differed between habitat types and depth zones. If the analysis resulted in differences in community structure for either test, a similarity percentages analysis (SIMPER) determined which taxa contributed to 70% of the differences observed (Peters et al. 2015). I then used a CLUSTER analysis to visually identify similarities in fish communities among habitat types and depth zones.

Species that accounted for more than 5% of the individuals counted at each site were used to create a species similarity matrix using species counts and Whittaker's index of

association (Whittaker 1952, Streich et al. 2017b). Hierarchical agglomerative clustering with a type 3 SIMPROF testing were used to evaluate species associations (Sommerfield & Clarke 2013). A shade plot was then created to qualitatively visualize selected species contributions across habitat types (Clarke et al. 2014, Streich et al. 2017b). Habitat type and depth zone were set as fixed factors for all significance tests. All community analyses were run in Primer Version 7.0. All statistical analyses were conducted using the *onewaytests* (Dag et al. 2018) and *vegan* (Clarke 1993) packages in RStudio (R Core Team version 3.5.1; RStudio Team, 2018).

Hydroacoustic Sampling

To determine the abundance and distribution of economically important species, I used ROV data in conjunction with hydroacoustic surveys using a Simrad EK80 split beam echosounder (ES70-18CD transducer; circular beam width of 18°, pulse duration = 0.256 μ s, specified ping rate = 300 ms). Seventy-eight hydroacoustic surveys (49 artificial reef and 29 natural bank) were conducted, which encompassed most ROV sites and included additional sites that were not able to be sampled by the ROV (i.e., current, water, or wind conditions prevented anchoring to a site). Several artificial reefs that were sampled harbor multiple structures dispersed within a reefing block. Preliminary visual assessments of biomass distribution around artificial structures identified distinctions between fish schools of adjacent structures was routinely less than 30 m. A 20 m buffer was then added to this distance to classify individual structures. All structures less than 50 m apart were considered to be one structure, while structures greater than 50 m were counted as separate structures.

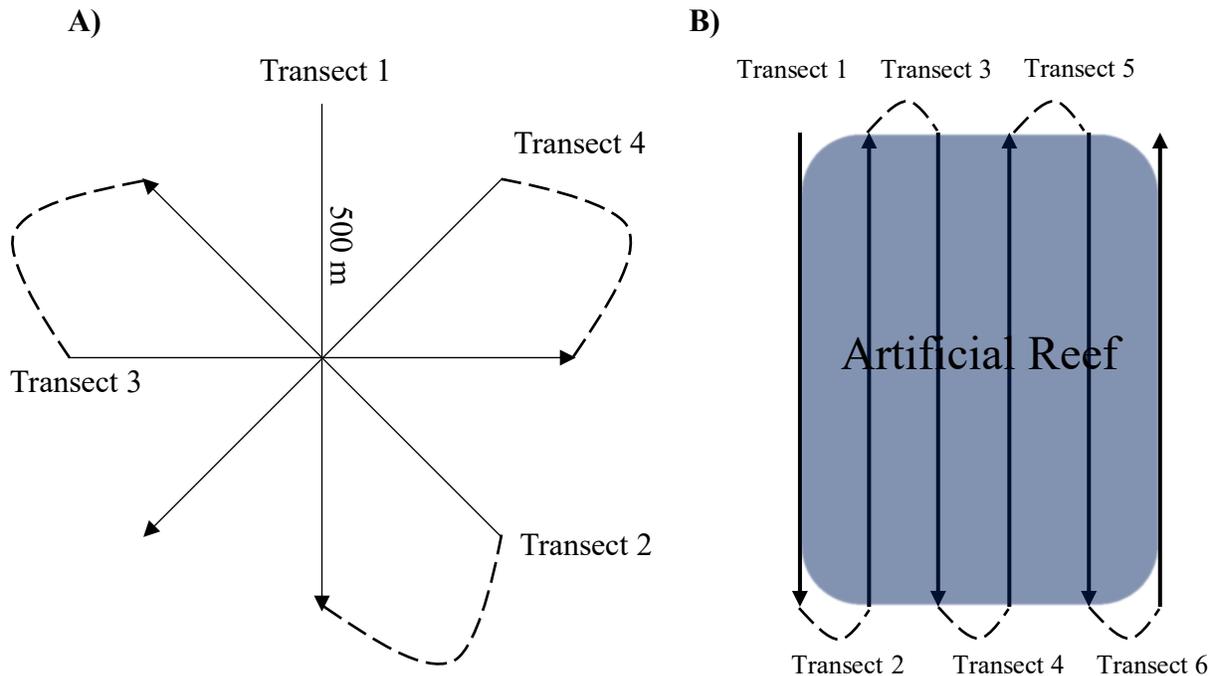


Figure 2-2. Diagram representing the two hydroacoustic tow patterns used for artificial reefs and natural banks. A) Example of a flower-pattern survey where the four, 500-m echosounder transects were centered over the geographic station position for each site. B) Example of a back-and-forth sweeping “mow-the-lawn” survey pattern used at large artificial reef sites with multiple structures.

The EK80 was either mounted to a towed body and towed 50 m behind the boat or fixed to a pole-arm and positioned 1.9 m below the surface. At natural banks and small or singular artificial reefs, survey patterns consisted of four, 500-m transects in a “flower” pattern centered on the geographic station position (Reynolds et al. 2018; Figure 2-2A). At large artificial reefs with more than one structure, transects were in a back-and-forth “mow-the-lawn” pattern covering 30 m beyond the structure in every direction (Figure 2-2B). Prior to each survey event, the EK80 echosounder was calibrated using standard methods and a 38.1 mm tungsten carbide sphere (Foote et al. 1987). All hydroacoustic data were binned into 10-m depth intervals from the sea floor and ordinary kriging was used to account for spatial autocorrelation (detailed below).

To allow for integration of the species composition with hydroacoustic data, further post-processing of the ROV dataset was required. First, species unlikely to be detected acoustically due to size, morphology (e.g., damselfishes, sharks, etc.), cryptic behavior, close association with the benthic fouling, or on the structure were removed prior to further analysis (Simmonds and MacLennan 2005; Wilson et al. 2003, 2006). Second, individuals were then allocated into 10-m depth bins from the sea floor to mirror hydroacoustic depth bins. Because ROV surveys were not conducted at all sampled sites, species composition was pooled within each habitat type and depth zone (shallow, mid, and deep) and applied to the hydroacoustic data. As a result, the average MaxN for each habitat type and depth zone was used to calculate percent species composition within each depth bin. To properly allocate fish species composition to depth bins that mirrored hydroacoustic data, the percent distance from the seafloor (proportional height) was calculated for each site using the following formula and methods outlined in Stunz et al. (2021):

$$(1) \quad \textit{Proportional Height} (\%) = \frac{\textit{Site Depth} - \textit{Fish Depth}}{\textit{Site Depth}} \times 100$$

This proportional height approach places greater emphasis on the association of a fish to vertical structures than depth and ensured that sites with similar structure types (e.g., reefed oil and gas platforms) would be assigned a species composition appropriate to the structure (Stunz et al. 2021). For example, if one Red Snapper was recorded at 40 m at a site with a maximum depth of 53 m, this individual would have a proportional height of 24.5%. If I consider two sites with differing depths of 71.2 m and 45 m, both with an artificial structure elevated 30 m from the bottom, and I apply the actual depth of the Red Snapper to each site, then I observe that the fish is near the bottom at the shallower site, but above the structure at the deep site. On the other hand, my proportional height approach places the individual at 34 m in the shallower site and 54

m in the deeper site, which results in a more representative depth for reef-associated species such as Red Snapper. Once fish occurrences were allocated into depth bins, the MaxN counts were averaged for each depth bin by species and a proportional species composition was calculated to then be applied to the hydroacoustic data (Stunz et al. 2021).

Hydroacoustic Analysis

Raw hydroacoustic data were processed in the lab using Echoview 10.0 (SonarData Pty Ltd., Hobart, Australia). Prior to analyzing, data were edited to exclude unwanted noise and reverberation (Simmonds & MacLennan 2005, Boswell et al. 2007). Excluding noise reverberation resulted in excluding 0.5 m from the sea floor and 1 m around artificial structures. Echo integration approaches were used when individual fish were too closely distributed for echo counting to be successful (Boswell et al. 2007). This study then used methods described in Stunz et al. (2021) to account for spatial autocorrelation. Ordinary kriging was applied to each layer using the following steps. First, a variogram, using initial values calculated from the data, was used to model the spatial continuity of the data collected at each site. Second, a 5 x 5-m survey grid was projected over each depth bin. Third, ordinary kriging interpolated spatially weighted estimates of area density over the survey grid from the predicted variogram. This resulted in depth bin-specific hydroacoustic density estimates for 5 x 5-m cells arranged over the area bounded by the ends of the transects performed at each site (Stunz et al. 2021).

Fish abundance in each cell in each depth bin was calculated by multiplying the estimated density (fish/m²) by 25 m² (the area of the cell). Depth bin-specific abundance was then calculated by summing the abundances from each cell. Species-specific abundances at each site were then calculated by taking the product of abundance and percent species composition for each depth bin. These abundance estimates were summed across all layers to generate a total

species-specific abundance for each site. A two-way ANOVA ($\alpha = 0.05$) then tested for differences in log-abundances between habitat type and depth zone for each of the four economically important species.

Economically Important Species Analysis

All Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper observed during ROV surveys were inferred to be age 2+ based on known age-length relationships for each species (Johnson & Saloman 1984, Goodyear 1995, Manooch III & Potts 1997, Potts et al. 1998, Johnson et al. 2010, Burton et al. 2015). As a result, extrapolated abundance estimates were compared to the most recent ‘SouthEast Data Assessment and Review’ (SEDAR) for each species. To extrapolate abundance estimates for Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper, I found the number of artificial structures and the area of natural banks along the Texas continental shelf from 10-150 m bottom depth. Known artificial structures were identified from various sources including Bureau of Ocean Energy Management (BOEM 2020a,b), National Oceanic and Atmospheric Administration (NOAA 2020), Bureau of Safety and Environmental Enforcement (BSEE 2020), Texas General Land Office (TXGLO 2018), Texas Parks and Wildlife Department (TPWD 2020), RGV Reef (2020), Horner (2013), GMFMC (2013), K. Rose, unpublished data, and Stunz, unpublished data. Artificial reef estimates were categorized by large and small structures to account for the drastic differences in abundance between the two types of structures. Large artificial structures (n = 941) included oil and gas structures and ships, whereas small artificial structures (n = 3,460) included concrete pyramids and culverts. Known natural reef habitats were identified from sources including Shirley (2012), Horner (2013), National Marine Fisheries Service data (NMFS 2019), and Stunz et al. (2021).

Results

ROV Surveys

Video-based ROV surveys resulted in ~38 h of footage. Overall, 10,818 individuals from 75 species representing 29 families were enumerated. Species richness ranged from one species at Hospital_1 (natural) to 26 species at HI-A-596 (artificial - standing platform) and Southern_3 (natural). The most speciose families were Carangidae ($n = 15$), Epinephelidae ($n = 11$), Labridae ($n = 6$), and Pomacanthidae ($n = 6$), which is consistent with other studies conducted in the northern GOM (Ajemian et al. 2015a, Streich et al. 2017a,b, Gallaway et al. 2021). The five most abundant species were Vermilion Snapper ($n = 1,815$), Red Snapper (817), Bermuda Chub (718), Blue Runner (717), and Atlantic Creolefish ($n = 507$). Unidentified fish ($n = 1,435$) accounted for 13% of all individuals counted. Species richness significantly varied with habitat type ($F_{1,51} = 23.67, p < 0.01$) but not with depth zone ($F_{2,51} = 2.57, p = 0.09$) nor with their interaction ($F_{2,51} = 0.14, p = 0.87$). Average species richness was greater at artificial reefs than natural banks for all depth zones (Figure 2-3). Similarly, Shannon diversity index only varied significantly with habitat type ($F_{1,51} = 12.95, p < 0.01$). Pielou's evenness index did not differ with habitat type ($F_{1,51} = 1.49, p = 0.23$) nor with depth zone ($F_{2,51} = 0.07, p = 0.94$).

Non-metric multidimensional scaling (NMDS) plot revealed significant groupings of fish communities by the interaction between habitat type and depth zone (Figure 2-4). The two-way ANOSIM indicated significant differences in community structure among depth zones ($R = 0.318, p < 0.01$), but not between habitat types ($R = 0.137, p = 0.09$). Although a complimentary CLUSTER analysis revealed unremarkable clusters, all groups with greater than 60% similarity were of the same habitat type (Figure 2-5).

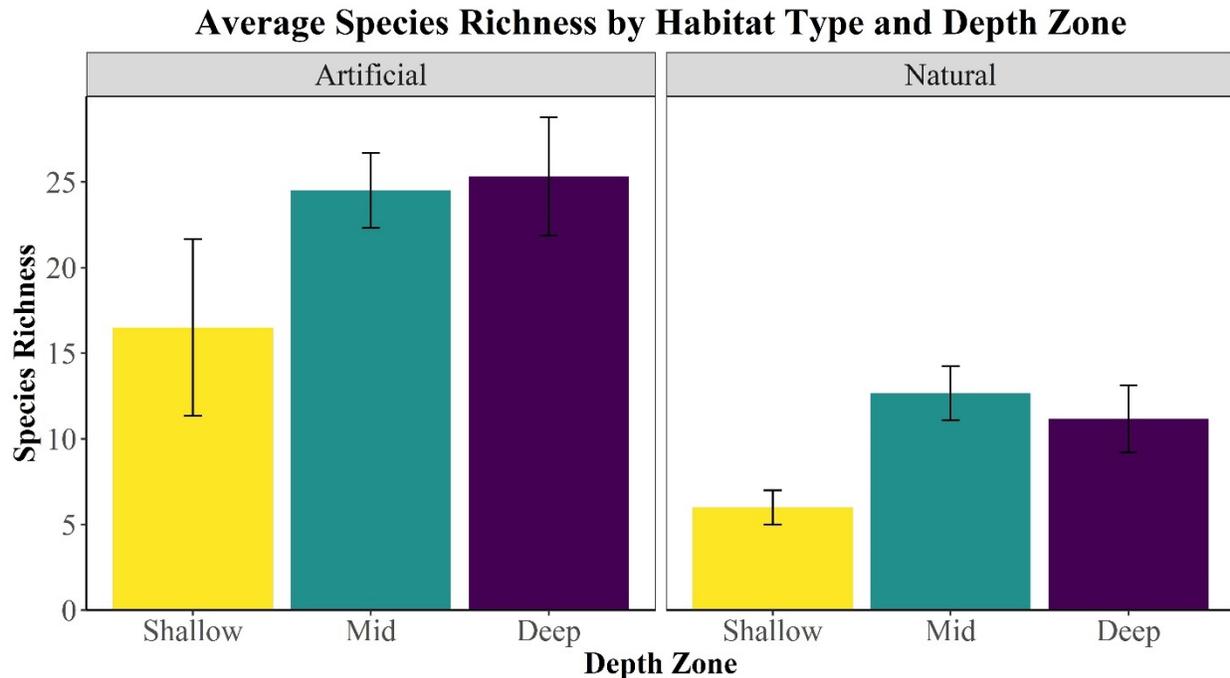


Figure 2-3. Bar graph illustrating the average species richness (mean±SE) within each depth strata (shallow, mid, and deep) between artificial reefs and natural banks.

Average similarity in fish communities between habitat types was around 22% with Vermilion Snapper accounting for more than 10% of the differences seen between habitat type. Likewise, within artificial reefs, Vermilion Snapper contributed to about 14% of the differences observed (average similarity 40.6%). Within natural banks, Red Snapper contributed around 29% to the differences observed in community structure (average similarity 24.8%).

Fish Community NMDS Plot

Resemblance: S17 Bray-Curtis similarity

2D Stress: 0.18

SIMPROF

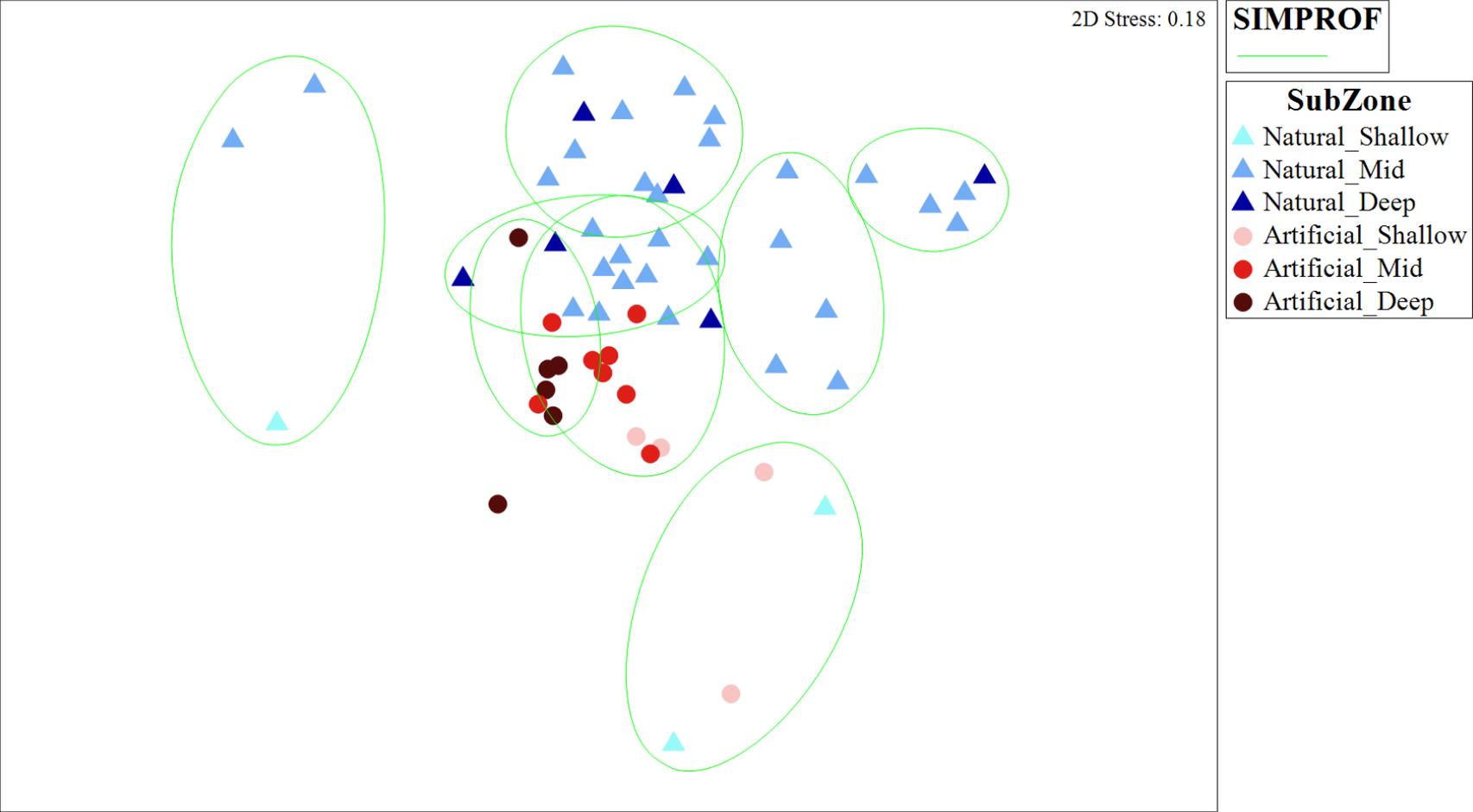


Figure 2-4. Non-metric multidimensional scaling (NMDS) plot of fish communities recorded from remotely operated vehicle surveys. Symbols differentiate habitat types, while colors differentiate depth zones. Percent similarity bubbles (from CLUSTER) represent 50% SIMPROF groups. *SubZone* refers to the variable composed of habitat type and depth zone.

Fish Community Dendrogram

Group average

Resemblance: S17 Bray-Curtis similarity

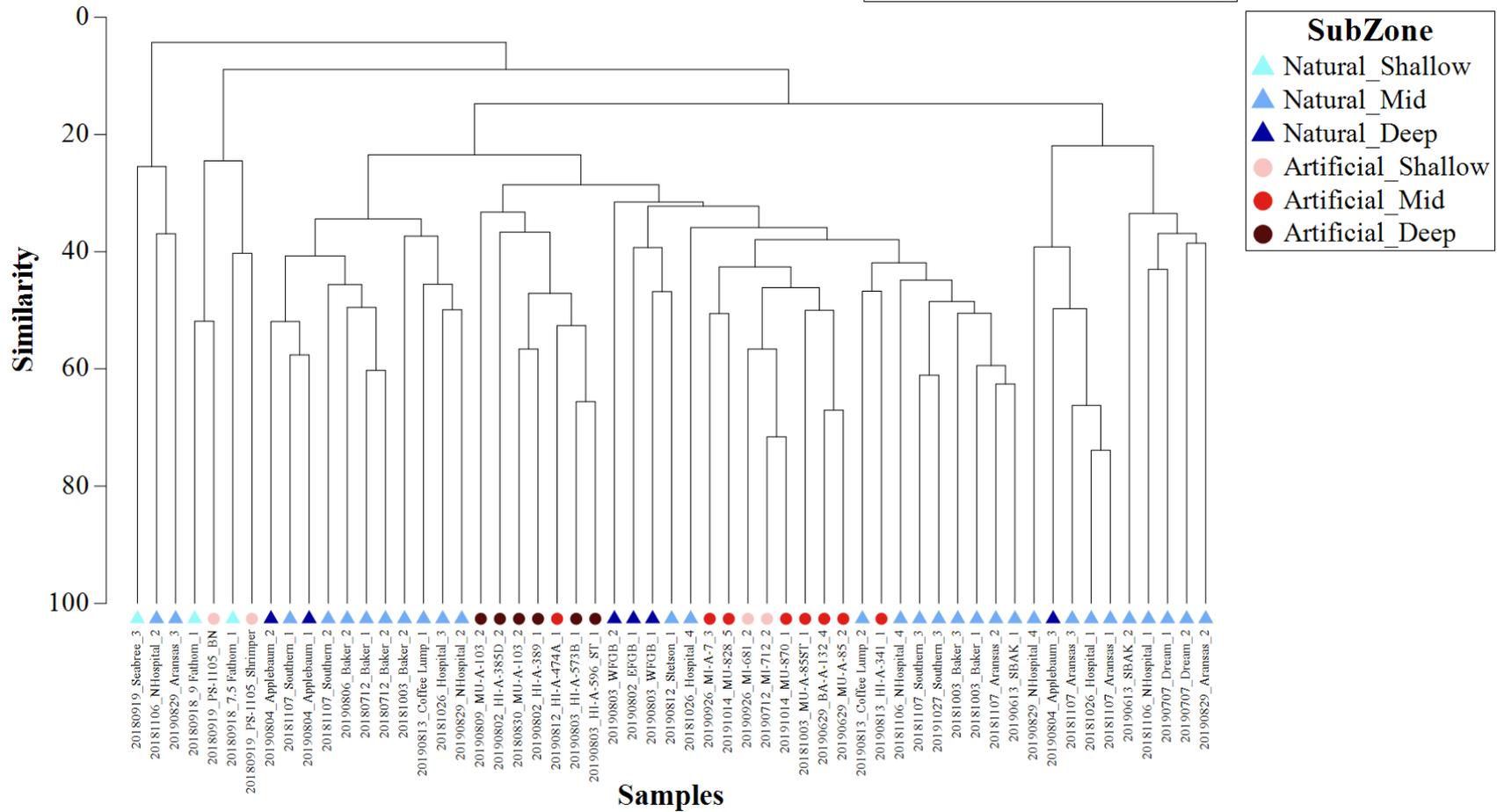


Figure 2-5. CLUSTER dendrogram showing linkages of communities by habitat type and depth zone. Linkages are based on Bray-Curtis similarity matrix of square-root abundances. *SubZone* refers to the variable composed of habitat type and depth zone.

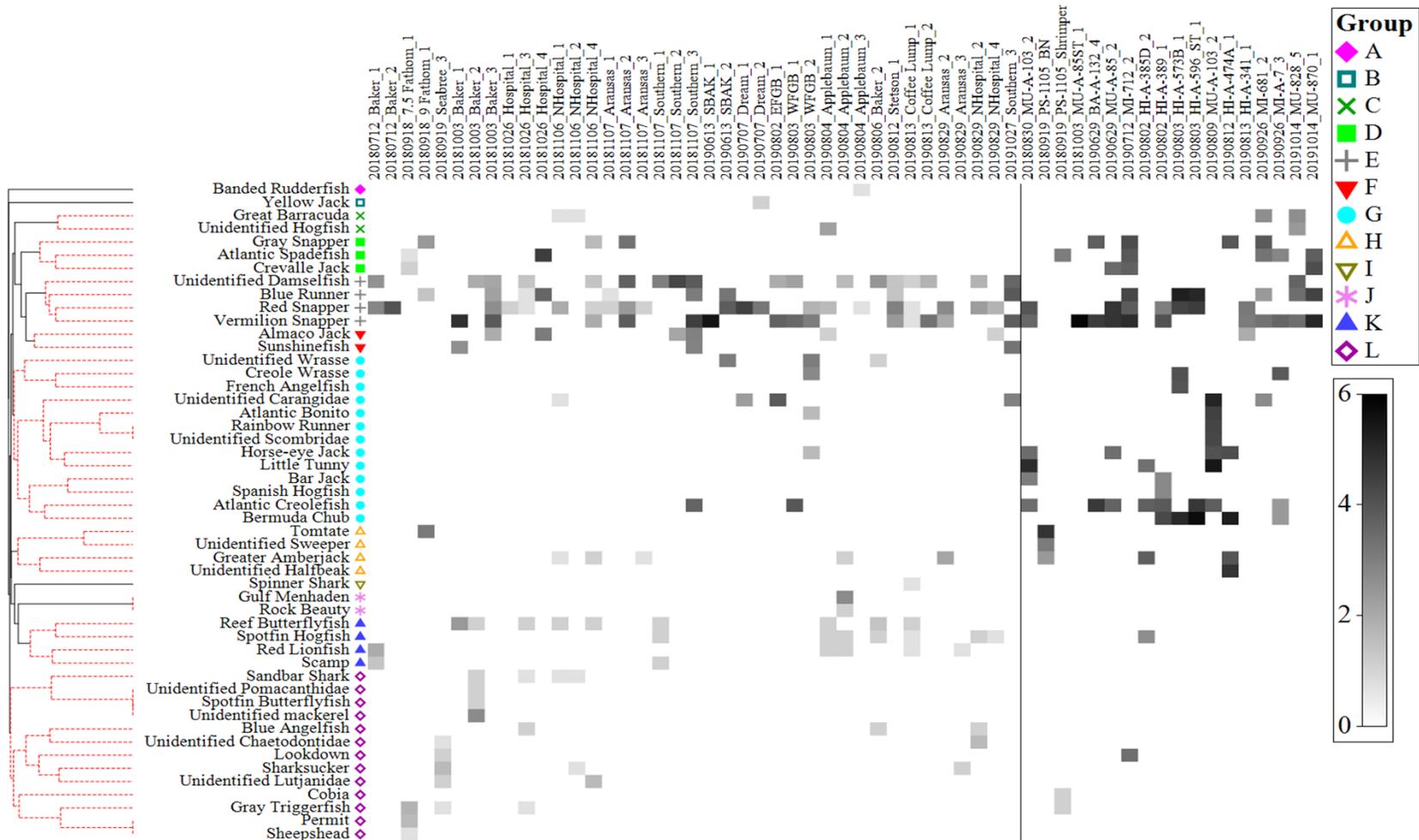


Figure 2-6. Shade plot representing the samples (columns) and species (rows) of the species contributing at least 5% of the counts for each site. Species dendrogram on the left illustrates the hierarchical clustering based on Whitaker's (1952) index of association resemblance matrix computed on natural log MaxN counts. Species groups were categorized using a Type 3 similarity profile testing ($P < 0.001$) as indicated by the connected red lines in the dendrogram and by the symbols displayed next to the species name. All sites to the left of the vertical line are natural banks and all sites to the right of the vertical line are artificial reefs.

The largest dissimilarity in fish communities among depth zones was observed between deep and shallow zones (average similarity 11.1%). Vermilion Snapper, Atlantic Creolefish, and Red Snapper contributed around 15% to these differences with the deep zone having greater abundance for all species. The mid and shallow depth zones were also quite dissimilar (average similarity 12.9%), where Red Snapper and Vermilion Snapper were more abundant in the mid depth strata. When comparing deep and mid depth zones, Red Snapper were more abundant in the mid depth zone and Vermilion Snapper were more abundant in the deep zone (combined contribution of 16%; average similarity 25%). Within depth zone, the greatest dissimilarity was observed within the shallow zone (average similarity 25.3%). Large differences in abundance of Gray Triggerfish and Atlantic Spadefish contributed around 38% of the dissimilarity within the shallow zone.

Results from the SIMPER analysis promoted further investigation into the differences within artificial reefs. I conducted an additional ANOSIM to test for differences in community structure between artificial reef structure types. Due to uneven sample sizes in structure types, artificial reefs were separated into cutoff ($n = 7$), standing ($n = 4$), or toppled platforms ($n = 4$), and other (i.e., deck, concrete, and ship; $n = 3$). Significant differences were observed between artificial reef structure types ($R = 0.209$, $p = 0.04$) with the largest dissimilarity between other structures and standing platforms (average similarity 21.5%). These differences were driven by higher abundances of Bermuda Chub and Vermilion snapper at standing platforms than other structure types.

After removing the species that contributed less than 5% of counts at each site, 50 species remained for further analysis (30 for artificial reefs and 42 for natural banks). Of those 54 species, 17 were only observed at one site. Although artificial reefs contained fewer contributing

species than natural banks, counts were noticeably higher for more species than natural banks. Clustering based on the remaining species and type 3 SIMPROF ($P < 0.001$) identified 12 distinct groups. Three groups contained a single species (groups A, B, and I in Figure 6), while two groups contained 13 species (groups G and L in Figure 2-6). Group E contained four species that were the most abundant across all sites (Figure 2-6). Groups J, K and L contained species that were primarily observed at natural bank sites, while Group G contained species that were primarily observed at artificial reefs (Figure 2-6).

Economically Important Species

Overall, 25,122 Red Snapper, 11,554 Greater Amberjack, 1,656 Grey Triggerfish, and 43,822 Vermilion Snapper were observed across all sites. Abundance estimates were maximized at artificial reefs for all species (Figure 2-7). Greater Amberjack were more abundant in the deep depth zone, while Red Snapper, Gray Triggerfish, and Vermilion Snapper were more abundant in the mid depth zone (Figure 2-7). Vermilion Snapper were not observed at artificial reefs in the shallow zone and Gray Triggerfish were not observed in the deep zone at either habitat type (Figure 6).

Results from the two-way ANOVA revealed Greater Amberjack log-abundances varied significantly with the interaction between habitat type and depth zone (Table 2-2; Figure 2-7). A post-hoc Tukey test confirmed that Greater Amberjack were significantly more abundant at artificial-deep sites than natural-deep sites (adjusted $p = 0.005$). Greater Amberjack also increased in abundance with depth at artificial sites and decreased in abundance with depth at natural sites (Figure 2-7). Additionally, Vermilion Snapper log-abundances varied significantly with depth zone (Table 2-2), with greater abundances at the mid depth strata than the deep strata (adjusted $p = 0.005$) and no Vermilion Snapper observed in the shallow depth zone at natural

sites (Figure 2-8). Neither Gray Triggerfish nor Red Snapper varied significantly with either of the main effects or their interaction (Table 2-2). However, Gray Triggerfish were not observed at any deep sites (Figure 2-8). Within artificial reefs, Red Snapper and Greater Amberjack were most abundant at oil and gas structures in the deep depth zone, Gray Triggerfish were most abundant at vessels in the shallow depth zone, and Vermilion Snapper were most abundant at oil and gas structures in the mid depth zone, and Vermilion Snapper were most abundant at oil and gas structures in the mid depth zone. Vermilion Snapper were only observed at oil and gas structures.

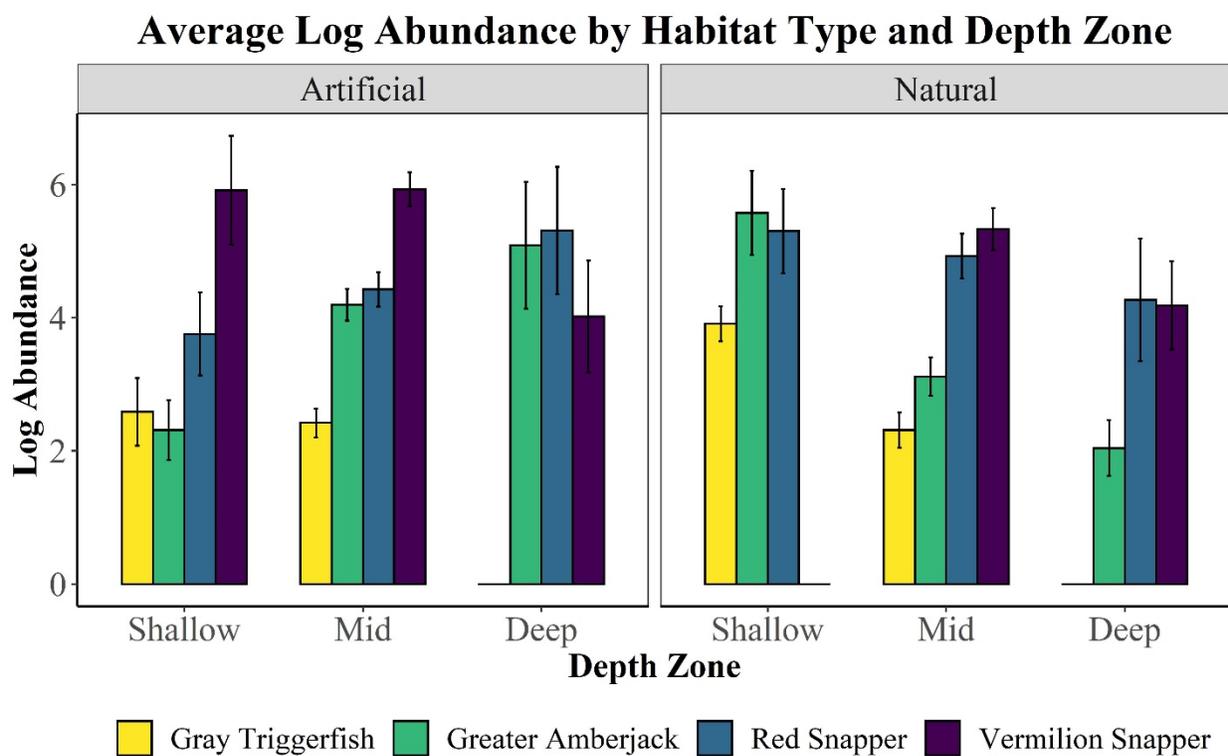


Figure 2-7. Average log abundances (mean±SE) of the four economically important species (color) by habitat type and depth zone.

Absolute abundance estimates for Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper are indicated in Table 2-3. Abundance estimates for age 2+ Red Snapper at artificial reefs and natural banks was about 14% ($n = 5,535,958$) of the Gulf-wide estimate (SEDAR 2018). Similarly, the Gray Triggerfish ($n = 290,697$) and Vermilion Snapper ($n = 4,917,482$) abundance estimates were about 10% and 12%, respectively, of the Gulf-wide

estimates for each species (Table 2-3; SEDAR 2015, SEDAR 2020). Conversely, the Greater Amberjack abundance estimate from this study was 221% (n = 1,483,334) of the Gulf-wide estimate for age 2+ individuals (SEDAR 2020).

Table 2-2. Two-way analysis of variance testing for differences in log-abundance between habitat types and among depth zones for each of the four targeted species. Asterisk indicates a significant result at the 0.05 level of significance and *df* refers to degrees of freedom.

Note: One degree of freedom was reported for the interaction variable for Vermilion Snapper due to no recorded abundances at natural banks in the shallow depth zone.

Species	Variable	<i>df</i>	<i>F</i>	<i>p</i>
Red Snapper	Habitat Type	1	0.737	0.393
	Depth Zone	2	0.87	0.424
	Habitat Type*Depth Zone	2	1.610	0.207
	Residuals	72		
Greater Amberjack	Habitat Type	1	5.700	0.020*
	Depth Zone	2	1.981	0.145
	Habitat Type*Depth Zone	2	13.323	<0.001*
	Residuals	72		
Gray Triggerfish	Habitat Type	1	0.157	0.694
	Depth Zone	1	1.476	0.229
	Habitat Type*Depth Zone	1	2.394	0.127
	Residuals	61		
Vermilion Snapper	Habitat Type	1	1.165	0.285
	Depth Zone	2	5.688	0.005*
	Habitat Type*Depth Zone	1	0.61	0.438
	Residuals	63		

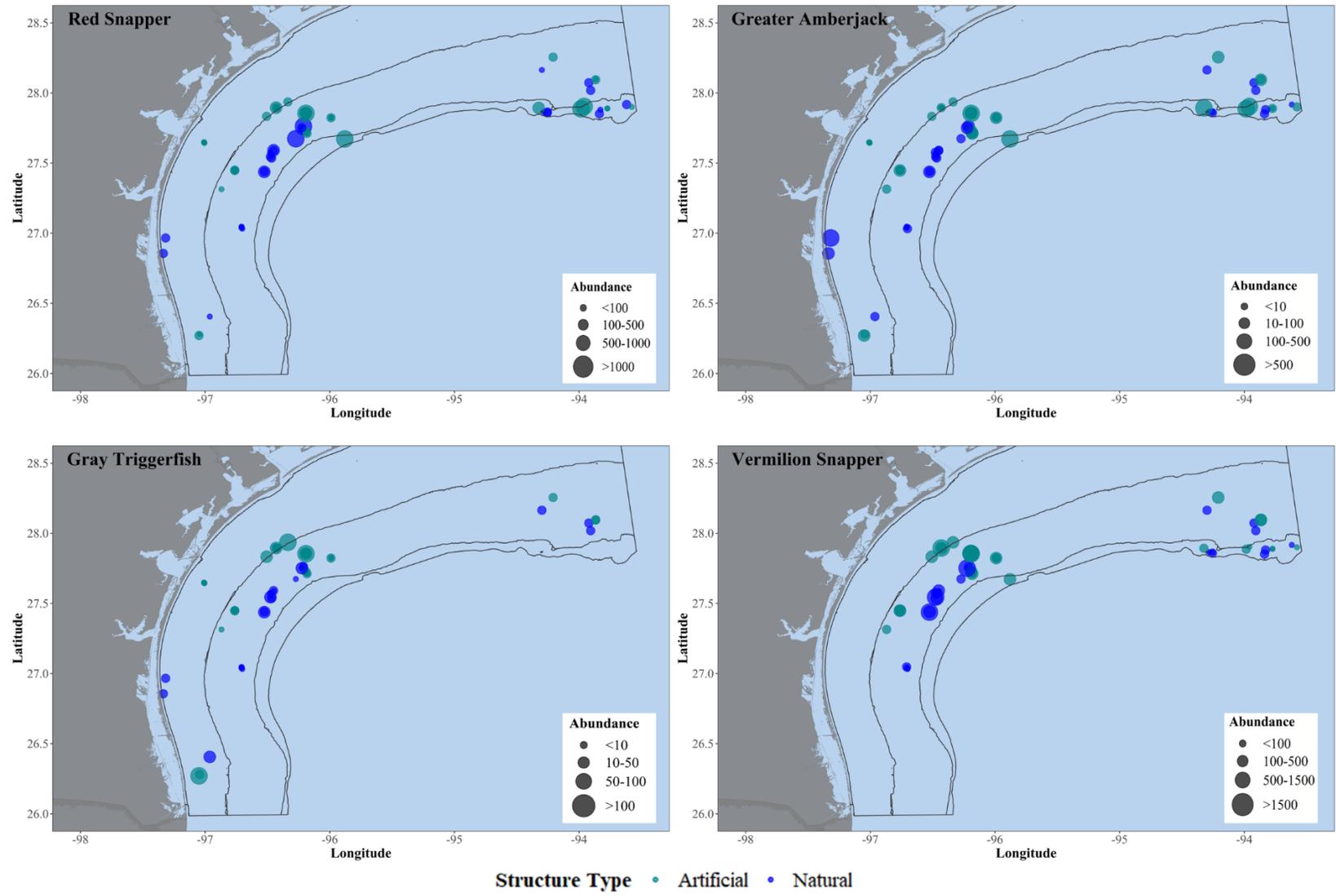


Figure 2-8. Abundance bubble plots for each targeted species in the three different depth strata along the Texas coast. Bubble classifications vary by species. Turquoise bubbles indicate artificial reefs and blue bubbles indicate natural banks.

Table 2-3. Absolute abundance estimates for age 2+ Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper at natural banks and artificial reefs for the Texas coast from 10-150 m. Natural banks were evaluated by area and artificial reefs were evaluated per structure. *SD* refers to standard deviation and *CV* refers to coefficient of variation. SEDAR estimates are from the most recently assessed year for each species (Gallaway et al. 2021).

Species	Habitat Type	Area (km ²) or Structures	Mean Density (100 m ²) or by Structure	Total Abundance	SE	CV (%)	SEDAR Age 2+ Gulf-wide Estimates	Year Estimated
Red Snapper	Natural	1,570	0.292	5,170,609	2,481,892	48		
	Artificial	4,401	-	365,349	112,893	31		
	<i>Large</i>	941	349	328,393	101,703	31		
	<i>Small</i>	3,460	11	36,956	18,871	51		
	Total			5,535,958	1,217,911	22	36,738,000	2016
Greater Amberjack	Natural	1,570	0.081	1,271,700	559,548	44		
	Artificial	4,401	-	211,634	67,723	32		
	<i>Large</i>	941	209	196,894	63,003	32		
	<i>Small</i>	3,460	4	14,969	7,644	51		
	Total			1,483,334	400,500	27	670,340	2018
Gray Triggerfish	Natural	1,570	0.016	309,577	74,298	24		
	Artificial	4,401	-	39,398	8,668	22		
	<i>Large</i>	941	29	27,652	6,222	23		
	<i>Small</i>	3,460	3	11,928	6,091	51		
	Total			290,697	49,418	17	2,822,750	2013
Vermilion Snapper	Natural	1,570	0.267	4,191,900	1,341,408	32		
	Artificial	4,401	-	725,582	183,647	25		
	<i>Large</i>	941	771	725,582	183,647	25		
	<i>Small</i>	3,460	-	-	-	-		
	Total			4,917,482	1,032,671	21	41,046,040	2017

Discussion

Species Composition

This study documented robust fish communities at artificial reefs and natural banks along the Texas continental shelf. Similar to other studies conducted in the northern Gulf of Mexico, artificial reefs were significantly more speciose and diverse than natural banks (Wilson et al. 2003, 2006). While not the focus of this study, greater habitat complexity and vertical relief at artificial reefs than natural banks most likely explain the dissimilarities in diversity and richness (Lingo & Szedlmayer 2006). More specifically, higher densities of predatory species (e.g., Greater Amberjack) at high-relief reefs could result from increased predation success in a more 3-dimensional foraging area compared to a 2-dimensional area at low-relief structures (Garner et al. 2019). On the other hand, predatory activity at natural reefs is less efficient due to fish biomass spread over a larger area (Karnauskas et al. 2017). Overall, habitat generalists typically occupy a variety of habitat types, while more obligate reef species are strongly regulated by reef complexity (Garner et al. 2019).

Despite fewer species at natural banks, natural banks contained 12 more species contributing to greater than 5% of counts than artificial reefs. However, five of those species could only be identified to family and an additional five species were only observed at one site. The dominance of counts from coastal pelagic species at artificial reefs could have also limited the number of species that contributed greater than 5% of counts. Likewise, the species grouping that was primarily observed at artificial reefs, contained species that are typically observed higher in the water column such as Bermuda Chub and several Carangids. These results accentuate the role of verticality driving differences in fish communities between artificial reefs and natural banks.

Visibility may have also played a role in the differences observed between artificial reefs and natural banks. On average, visibility was about 2 m, or a sampled volume of about 12 m³, greater at artificial reefs than natural banks due to a persistent benthic nepheloid layer (Shideler 1981, Dennis & Bright 1988). Sampled volume differences have the potential to impact the counts and number of species visually available to readers because catchability of the gear is changing relative to visibility. Furthermore, some standing platform visibilities decreased from 10 m at the surface to 1 m at the bottom, which could have favored upper and mid water column species (e.g., Greater Amberjack, Blue Runner, Bermuda Chub, etc.). However, for species that spend most of their time at or close to the bottom (e.g., Red Snapper), counts would have been similar between artificial and natural reefs (Stunz et al. 2021).

Along with visibility, video-based surveys might have also been impaired by behavioral responses to the ROV (Stoner et al. 2008). However, the impact of behavioral responses to the ROV may have been mitigated by the addition of the rear-facing GoPro. For example, readers often observed fish such as Great Barracuda (*Sphyraena barracuda*) “following” the ROV, which resulted in individuals missed by the forward-facing camera but were subsequently “caught” on the rear-facing camera. I suggest that additional cameras facing multiple directions be used in future ROV surveys to potentially catch individuals that exhibit various adverse behaviors to the ROV. Even with the addition of the second camera, species actively avoiding the ROV or more cryptic species are still likely to be missed and more charismatic species (i.e., most large-bodied fish) are likely to be inflated (Stoner et al. 2008).

Unlike other studies conducted in the northern GOM, my study sampled across a wide range of depths and latitudes along the Texas continental shelf. Consequently, fish communities did not differ between habitat types, despite fish densities being consistently higher at artificial

reefs than natural banks. However, this study observed fish community differences among depth zones, which is consistent with other studies conducted along the Texas continental shelf (Ajemian et al. 2015a,b). Bottom depth has long been recognized to play an important role in shaping fish communities (Gallaway & Lewbel 1982, Bell 1983, McClatchie et al. 1997, Garcia-Sais 2010, Zintzen et al. 2012, Ajemian et al. 2015). In this study, Red Snapper, Vermilion Snapper, Gray Triggerfish, and Atlantic Creolefish were often the main drivers of fish community differences among depth zones. Depth has been observed affecting abundances or playing a role in the life histories for each of the aforementioned species (Posado-Lopez 1996, Harper & McClellan 1997, Rooker et al. 2006, Allman 2007, Gallaway et al. 2009). Significant differences in fish communities were also observed within artificial reefs, which has been observed in other studies conducted in the northern GOM (Wilson et al. 2003, 2006, Lingo & Szedlmayer 2006, Ajemian et al. 2015). Similar to Ajemian et al. (2015a), structure type differences were mainly driven by high abundances of Bermuda Chub at standing platforms. Previous studies have suggested that this species is likely more dominant at standing platforms due to greater availability of photosynthetic biomass at the top of the water column (Randall 1963, Silvano & Güth 2006, Downie et al. 2013, Michael et al. 2013, Ajemian et al. 2015a,b).

The lack of clear clustering due to habitat type or depth indicates that fish community structure is nuanced in several other factors not examined in this study. Future studies should consider incorporating other mechanisms such as species-specific resource use, predator-prey interactions, ontogenetic shifts in habitat requirements, and reef relief and complexity, which have all been identified as factors shaping fish community structure in the northern GOM (Chandler et al. 1985, Lingo & Szedlmayer 2006, Adams & Ebersole 2009, Dance et al. 2011,

Patterson et al. 2014, Garner et al. 2019). Future studies should also surveying fish communities over successive seasons and years to reveal other factors affecting their structure in the GOM.

Economically Important Species

There are several biases that may have affected abundance estimates of economically important species. First, recent studies have suggested that density estimates from ROV surveys are potentially biased due to highly dynamic, species-specific behavioral responses to the ROV (Campbell et al. 2021). Although ROV calibration attempts have yet been unsuccessful (Campbell et al. 2021), a recent study found minimal evidence for potential sources of error that would bias Red Snapper density or abundance estimates using a ROV (Garner et al. 2022). Certainly, there is potential for species demonstrating strong attraction to sampling equipment to positively bias counts and detections, and thus gear calibrations are recommended for future studies.

Second, abundance estimates from hydroacoustic surveys were likely underestimated due to the exclusion procedure for unwanted noise and reverberation. Data 0.5 m above the sea floor as well as data within and 1 m around artificial structures were omitted due acoustic reverberation. This procedure would have drastically reduced the abundance estimates at artificial reefs, particularly at oil and gas platforms, where fish were commonly observed within the structure during video analysis. Furthermore, hydroacoustic analyses resulted in several (n = 95) schools which did not have single targets associated with them and could not be scaled. Unscaled schools could not be converted from biomass to abundance and were omitted. Thus, the data presented in Table 2-3 are likely underestimated abundances for each of the four economically important species, particularly at artificial reefs.

Despite biases associated with using hydroacoustics as a sampling technique, numerous studies conducted in the GOM have also used hydroacoustics for reef fish abundance estimates (Stanley 1994, Stanley & Wilson 1995, 1996, 1997, 2000, Gallaway 2018, Reynolds et al. 2018, Egerton et al. 2021). Red Snapper abundance ranges (158 – 1,240) at standing platforms from this study are consistent with ranges observed in other studies using hydroacoustics (Stanley and Wilson 1997, 2000, Gallaway et al. 2021). Likewise, Gallaway et al. (2021) observed similar average abundances at standing platforms for Greater Amberjack, Vermilion Snapper, and Gray Triggerfish to those reported in this study.

A study conducted in the Gulf of Mexico, commonly known as the Great Red Snapper Count (GRSC), was part of a Gulf-wide study dedicated to estimating the abundance of Red Snapper. While very comparable, slight differences between this study and the GRSC were observed in abundance estimates for artificial reefs and natural banks, these differences can largely be attributed to post stratification, weighting differences when compared to a gulf-wide estimate, and analytical methodologies (Stunz et al. 2021). For example, I obtained mean abundances by habitat type and depth strata and extrapolated based on the number of structures (artificial) or area (natural) within those criteria. Conversely, the GRSC extrapolated abundances using a more complex form of analysis incorporating differential weighting of abundance estimates based on the number of samples collected for each habitat type (natural or artificial), structure type (large or small artificial), depth zone (shallow, mid, and deep), and region-specific (north, central, and south) variances (Stunz et al. 2021). Coincidentally, sampling effort in the Texas region at natural banks was concentrated in the mid depth zone, which coincides with higher reported abundances for Red Snapper (Gallaway et al. 2009, Karnauskas et al. 2017, Stunz et al. 2021). In essence, natural banks sites with higher abundance estimates were

weighted more heavily. Because the majority of artificial and natural habitat reside within the preferred depth strata on the Texas continental shelf, I do not believe that this form of analysis is incorrect, merely a different way to use data from a random stratified sampling design.

Unlike the comparison with the GRSC, extrapolated Red Snapper abundance estimates at large artificial reefs from this study and those reported in Gallaway et al. (2021) at standing platforms were remarkably similar. Abundance estimates reported by Gallaway et al. (2021) for Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper at standing oil and gas platforms in Texas were roughly 12-53% of Texas-wide abundance estimates presented in Table 2-3 for large artificial reefs. Interestingly, standing platforms represent around 22% of artificial reefs that were classified as “large” in this study. Due to similarities in sampling region and technique, comparisons made between this study and the GRSC and Gallaway et al. (2021) are appropriate and corroborate my findings.

Abundance Distributions

In the GOM, it has been previously understood that the majority of suitable natural reef habitat for age 2-10 yr old Red Snapper occur at depths between 18 and 91 m (Parker et al. 1983, Gallaway et al. 2009); although in some regions this and other studies have shown Red Snapper occur at much shallower as well as deeper depths (Jaxion-Harm & Szedlmayer 2015, Powers et al. 2018, Gallaway et al. 2021, Stunz et al. 2021). Comparatively, abundance distributions presented in this study indicate greater abundances for Red Snapper at natural banks in the mid depth zone (40-100 m). Similar distributions were also observed for Vermilion Snapper at both habitat types. Most Vermilion Snapper were located in the mid depth zone with little to no sites with individuals in the shallow depth zone. Findings from Allman (2007) have indicated that Vermilion Snapper recruit to deeper depths as they age, with the average individual less than 265

mm in total length at depths less than 40 m. Because smaller or more cryptic species are typically more difficult to observe on video-based surveys (Stoner et al. 2008), Vermilion Snapper simply could have been missed by the ROV. However, Gallaway et al. (2021) also discovered an overwhelming majority of Vermilion Snapper in the northern GOM in the 31-90 m depth zone. As a result, Vermilion Snapper along the Texas continental shelf may not be abundant at depths less than 40 m.

Abundance distributions observed for Gray Triggerfish and Greater Amberjack at artificial reefs are also supported by Gallaway et al. (2021). Median abundances reported by Gallaway et al. (2021) for Gray Triggerfish were highest in 31-90 m depth zone and lowest in the 10-17 m and the 91-300 m depth zones. Gallaway et al. (2021) also recorded the highest median Greater Amberjack abundances in the 91-300 m depth zone. Although the depth zone classifications are different, similar abundance distributions can be observed in in this study, where abundance estimates for Gray Triggerfish were greater in the mid to shallow zones. Greater Amberjack abundances were also consistently greater in the deep depth zone, especially at artificial reefs. Although higher fish densities were observed at artificial reefs, these habitats may simply be concentrating the fish biomass and increasing catchability of predatory species using the vertical structure for increased foraging efficiency (Cowan et al. 2011, Karnauskas et al. 2017, Garner et al. 2019).

Management

Management priorities for each of the four economically important species studied are related to their abundance and overall value to the fisheries. Red Snapper and Vermilion Snapper represented two of the five most abundant species observed from video-based surveys. These species are also the primary targeted species by commercial fishers and are considered to be two

of the most lucrative offshore finfish fisheries in the northern GOM (Keithly Jr & Roberts 2017). In 2019, commercial landings of Red Snapper and Vermilion Snapper in the GOM equaled over \$35 million (fisheries.noaa.gov; accessed 9 June 2021). Commercial landings in Texas for Red Snapper and Vermilion Snapper accounted for 31% of total commercial landings in the GOM. Remarkably, recreational landings, which have an estimated added value of \$3.3 billion, often exceed commercial landings for both of these species (Keithly & Roberts 2017). Red Snapper specifically yield nearly twice as many landings from recreational fishers than commercial fishers in the GOM. However, despite the economic importance of Red Snapper and Vermilion Snapper, their respective stock assessments may be inaccurate in the GOM (Johnson et al. 2010, Cowan 2011, Cowan et al. 2011, Simmons et al. 2019, Scyphers et al. 2021, Stunz et al. 2021). For example, Vermilion Snapper population estimate models in the past have lacked data describing fish from the northwestern GOM (Johnson et al. 2010). Furthermore, recent controversy surrounding population estimates of Red Snapper in the GOM has led to discord between management agencies and fishers (Cowan 2011, Cowan et al. 2011, Simmons et al. 2019, Scyphers et al. 2021, Stunz et al. 2021). To summarize, available data on the status of the Red Snapper stock has led to proposed underestimates of the population; meanwhile, high catch-per-unit-effort suggests to fishers that the population is healthier than the stock assessments indicate (Simmons et al. 2019, Scyphers et al. 2021). Because Red Snapper stocks are rebuilding in the Gulf of Mexico (SEDAR 2018), population monitoring is critical for evaluating the status of the stock (Cowan 2011). With the next Red Snapper stock assessment scheduled to conclude in early 2023, this study, along with data from the GRSC, provide absolute abundance estimates at artificial reefs and natural banks along the Texas continental shelf to improve population estimates of the region and help alleviate discord between fishers and managers.

Similar to Red Snapper, both Gray Triggerfish (SEDAR 2015) and Greater Amberjack (SEDAR 2020) stocks are classified as rebuilding. Historically, evidence of recovery for overfished stocks in the United States has been essentially non-existent (Murawski 2010). In many cases, the absence of consistent monitoring affects appropriate adjustments to management practices and limits the success of the rebuilding plan (Rosenberg et al. 2006, Murawski 2010). In fact, the most recent stock assessment for Gray Triggerfish had to be halted after reviews identified inaccuracies in essential data inputs for growth and ageing (SEDAR 62). This study improves the scientific understanding of three species whose stocks are rebuilding in the GOM and presents population estimates at artificial reefs and natural banks to improve stock evaluation of all four species for future assessments.

Greater Amberjack

My study estimated an age 2+ Greater Amberjack abundance that was 221% of the current Gulf-wide abundance estimate. The severe disparity in abundance estimates could be a result of the misidentification of individuals during video surveys. For example, this study assumed that none of the individuals observed were Lesser Amberjack (*Seriola fasciata*). Lesser Amberjack are known to reside in the Gulf of Mexico, but misidentification with Greater Amberjack is common due to few differentiating features between species (Carpenter 2002, Deidun et al. 2011, SEDAR 2016). One of the most reliable methods for discerning the two amberjack species is via gill-raker counts, which was not within the scope of this study. Additionally, Lesser Amberjack are a data-limited species (SEDAR 2016), and I am unaware of any studies dedicated to estimating the ratio of Lesser to Greater Amberjack in the GOM.

The Greater Amberjack estimate from this study could also be an overestimate due to the over-representation of this species in the species composition that was applied to hydroacoustic

estimates. Greater Amberjack are a large-bodied, charismatic species that are easily observed on video-based surveys. Because they are rarely missed, the proportional MaxN of Greater Amberjack observed in this study could be larger than actual values. Furthermore, the presence of this large predatory species could have resulted in limited abundance or altogether exclusion of certain prey species and thus, amplifying the proportional abundance of Greater Amberjack. However, the average abundance of Greater Amberjack at large artificial reefs (Table 2-3) was similar to values observed in Gallaway et al. (2021) at standing platforms. Additionally, due to the hydroacoustic limitations previously discussed, I believe results from this study generally underestimated abundances.

Finally, Greater Amberjack estimates may be different due to inaccuracies in the stock assessment for this species in the GOM. The most recent stock assessment recognized issues concerning a significant fraction (30-40%) of the total abundance of Greater Amberjack to occur on oil and gas platforms in the central and western GOM (SEDAR 2020). Similarly, Gallaway et al. (2021) estimated that 45.1-48.3% of the GOM Greater Amberjack stock occurred on standing platforms in the northern GOM. Unfortunately, the most recent assessment abandoned changes to the current estimate due to a lack of time and data needed to restructure the model (SEDAR 2020). The Greater Amberjack stock assessment relies heavily on fishery data for stock size estimates, and landings off the west coast of Florida have historically dominated commercial and recreational catches in the GOM (Murie & Parkyn 2013). As a result, the stock assessment for this species may only be appropriate for the eastern GOM (Gallaway et al. 2021). To clarify issues surrounding the Greater Amberjack stock in the GOM, a study, known as the Great Amberjack Count (GAJC), is currently underway to enumerate Greater Amberjack and related species in the GOM. Until the conclusion of the GAJC, I recommend that the western GOM be

evaluated separately from the eastern GOM due to discrepancies in fishing effort between the two regions.

Conclusion

This research improves the scientific understanding of the role of artificial reefs in fish community dynamics as they relate to natural banks and informs managers of their impact on the populations of four economically important species. This study also provides absolute abundance estimates at artificial reefs and natural banks of four key species that support the majority of recreational and commercial fishing in the Gulf of Mexico. Although abundance estimates from hydroacoustics are not perfect, they are the ideal sampling methodology given the constraints for this type of challenging work in these environments with limited visibility. Furthermore, they are crucial in aiding conservation efforts and should be considered by fisheries managers when conducting stock assessments. Finally, this study illustrates the need to reevaluate the Greater Amberjack stock assessment to account for discrepancies in fishery effort between the eastern and western GOM.

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CHAPTER III:
ARTIFICIAL REEF LOCATION AND ORIENTATION MATTER FOR FISH
COMMUNITIES

Abstract

Artificial reefs and natural banks act as predictable aggregation sites for predatory fish and other marine species, which makes them especially vulnerable to exploitation. These reefs are also important resource enhancement tools, yet little is known about the impact of reefing location, orientation, and overall design on fish community structure and abundance, and how these factors could contribute to the most efficient reef design and construction. The purpose of this study was to evaluate the efficacy of differing densities and orientations of three nearshore prefabricated pyramid reefs on the Texas continental shelf. I conducted surveys using stereo-remote underwater video surveys to address the effect of prefabricated pyramid reefing location and orientation on fish community structure. Although reefing location nor orientation had a significant effect on species richness, diversity, and evenness, I identified differing communities between both variables (reefing location and orientation) with Red Snapper, Greater Amberjack and Gray Triggerfish among the main contributing species to these differences. Results from this study also showed the importance of lower relief reefs as important habitat for younger cohorts. Absolute abundance estimates were derived from combining the video surveys and hydroacoustic surveys and were used to calculate two types of cost-benefit ratios based on overall fish density and the densities of economically important species such as Red Snapper, Greater Amberjack, Gray Triggerfish, Vermilion Snapper, and Almaco Jack. Port O'Connor nearshore reef was considered the least cost-effective reefing location in terms of overall fish density, but not for economically important species density. Port O'Connor was configured with

science in mind and produced the most consistent results compared to the other reefing locations. This study provides examples of cost-effective artificial reefs and highlights the importance of developing a reef with science in mind to evaluate reef benefits more competently.

Introduction

In the northern Gulf of Mexico (GOM), artificial reefs have become an important resource enhancement tool (Bohnsack & Sutherland 1985). These structures act as fish aggregation sites on a continental shelf region that contains little to no vertical structure for fishes to seek refuge (Bohnsack & Sutherland 1985, Dennis & Bright 1988). Consequently, artificial reefs also act as fishing hotspots and overexploitation of certain easily accessible sites is common (Bradley & Bryan 1975, Beck & Odaya 2001, Simard et al. 2016, Stunz et al. 2021). Artificial reefs that produce significant new biomass have the potential to increase fishery resources (Bohnsack 1989, Streich et al. 2017b); however, artificial reefs that act primarily by attraction may increase catchability and thus promote overfishing (Bohnsack 1989). Whether for attraction or production, artificial reefs in the northwestern GOM have become an integral component in the regions' fisheries (Streich et al. 2017b). Economically important fish species (EIS) such as Red Snapper (*Lutjanus campechanus*), Vermilion Snapper (*Rhomboplites aurorubens*), Greater Amberjack (*Seriola dumerili*), Almaco Jack (*Seriola rivoliana*) and Gray Triggerfish (*Balistes capriscus*) often reside on artificial reefs, have integrated them into their life histories (Gallaway et al. 2009, Gallaway et al. 2021), and have been known to serve as nursery habitat for these species as well (Streich et al. 2017a,b, Karnauskas et al. 2017, Szedlmayer & Bortone 2020, Blount et al. 2021, Gallaway et al. 2021).

In response to the recognized importance of artificial reefs, the Artificial Reef Act of 1989 provided guidance for the development of cost-effective artificial reefs. One of the

recommendations of this Act states to arrange materials in as complex a fashion to not jeopardize the structural integrity, but to maximize effectiveness, durability, and stability (Stephan et al. 1990). Due to differing limitations in fish productivity or recruitment by region, artificial reefs should be designed to maximize abundance based on the needs of the region (Grossman et al. 1997, Strelchek et al. 2005, Blount et al. 2021). Furthermore, the materials used can also play a role in the overall effectiveness of the reef. Concrete materials, such as prefabricated pyramids, that have biomimetic properties, increase the benefits of reefs to the local environment (Vivier et al. 2021). An increase in artificial reefs in a region like the western GOM that lacks structured habitat for fishes should, in turn, increase survivability via refuge from predators and access to unexploited prey resources (Strelchek et al. 2005). Additional benefits of artificial reefs include acting as essential fish habitat for fisheries management (Bohnsack & Sutherland 1985, Shipp & Bortone 2009, Patterson et al. 2009, Ajemian et al. 2015a,b) and relieving the restraint on population abundance with additional habitats (Bortone 2008) that can also facilitate range expansion (Rooker et al. 1997, Love & York 2005, Broughton 2012).

While the cost of the reef is relatively straightforward to obtain, benefits can be categorized as either human or ecological. Prior to 2000, relatively few studies documented the human benefits of habitat enhancements such as artificial reefs (Seaman Jr. & Jensen 2000). After 2000, numerous studies focused on evaluating the benefits of artificial reefs (Black 2001, Pitcher et al. 2002, Stanley & Scarborough 2004, Adams et al. 2006, Whitmarsh et al. 2008, Chen et al. 2013, Polak & Shashar 2013, Schuett et al. 2016, to name a few). However, many of the studies fail to address how the benefit(s) contributes to the local economy. Human benefits of an artificial reef are directly related to its ecological benefits. More specifically, a healthy artificial reef is likely to contribute more to the local economy than an unhealthy reef. Evaluating

these ecological benefits of artificial reefs are vital for determining the relative success of the reef. Furthermore, when the ecological benefit is balanced with the cost associated with artificial reef implementation, return on investments in the project can be maximized (Adams et al. 2006, Graham et al. 2016). Quantifying the economic importance of an artificial reef may help justify future public artificial reef development projects and assist in developing adaptive strategies associated with reef deployment (Adams et al. 2006). With increasing legislative regulations on the types of materials permitted for artificial reefs, it is important to examine which materials provide the most benefit (Becker et al. 2018). In the United States, emphasis for artificial reef construction has been placed on ‘materials of opportunity’ (Stone et al. 1991, Pickering et al. 1999). However, few studies have examined the fish community response to different reefing materials (Brock & Norris 1989, Reed et al. 2006, Thanner et al. 2006, Ajemian et al. 2015b).

In general, reefing programs are limited by both materials and funds. Thus, having fundamental understanding of and developing wise reefing strategies are essential for programs to be effective. Therefore, the purpose of this study was to evaluate the efficacy of differing densities and orientations of three nearshore prefabricated pyramid reefs on the Texas continental shelf. My objectives were to: 1) assess the influence of reefing location and pyramid orientation on fish species richness, diversity, and evenness, 2) evaluate community differences among reefing location and pyramid orientation, and 3) develop two different cost-benefit ratios to evaluate the efficacy of each reef with respect to total fish biomass and economically important species biomass.

Methods

Study Area

Approximately 59,000 km², the Texas continental shelf and composed of primarily fine sands and shelf mud substrata (Flint & Rabalais 1980, Shideler 1981, Stunz et al. 2021). I studied three prefabricated pyramid reefing locations along the Texas coast: Port O'Connor Nearshore Reef (POC), Mustang Island 775 (MU-775), and Rio Grande Valley (RGV). The sampled sites for POC, MU-775, and RGV were reefed in 2017, 2013, and 2017, respectively. All three nearshore reefs were less than 20 km from the nearest inlet and in approximately 20 m bottom depth. Each location was composed of prefabricated pyramids with varying densities and groupings of pyramids (Figure 3-1). Sites with similar pyramid densities and groupings were categorized into a variable called orientation. POC nearshore reef and MU-775 artificial reef (Figure 3-1) are part of the Texas Parks and Wildlife (TPWD) artificial reefing program.

Port O'Connor nearshore reef was composed of approximately 700 prefabricated reef pyramids arranged in nine sites spanning 1.54 km² (Figure 3-1). Originally, there were three replicates of 40, 80, and 113 pyramids. however, several sidescan sonar surveys conducted in 2019 by TPWD on the Shell Oil reef in POC observed that 79 of the 200 pyramids were missing and another 34 pyramids were relocated outside of their original groupings due to Hurricane Harvey (Kline & Moody 2019, Moody 2019). Because Kline and Moody (2019) reported that roughly half of the pyramids were misplaced beyond their original grouping, I defined the number of pyramids at each site in POC as half of the number that were originally reefed (Table 3-1). For the purposes of this study, I refer to the differing densities as small (n = 20), medium (n = 40) and large (n = 57; Table 3-1). For MU-775 and RGV, I was able to gather a rough estimate of the number of individual pyramids using hydroacoustic surveys (methods below). Similar to

POC, the number of pyramids defined at MU-775 were also fewer than the number that were originally reefed (Table 3-1). All sites at MU-775 were grouped into a category labelled “grid” due to the lack of replication available and similar orientation among sites. RGV sites are all composed of four prefabricated pyramids categorizes into “nursery” and “non-nursery” sites. Nursery orientations harbored an additional 2-tons of loose concrete cinder blocks that provided low relief rubble habitat along with the four pyramids.

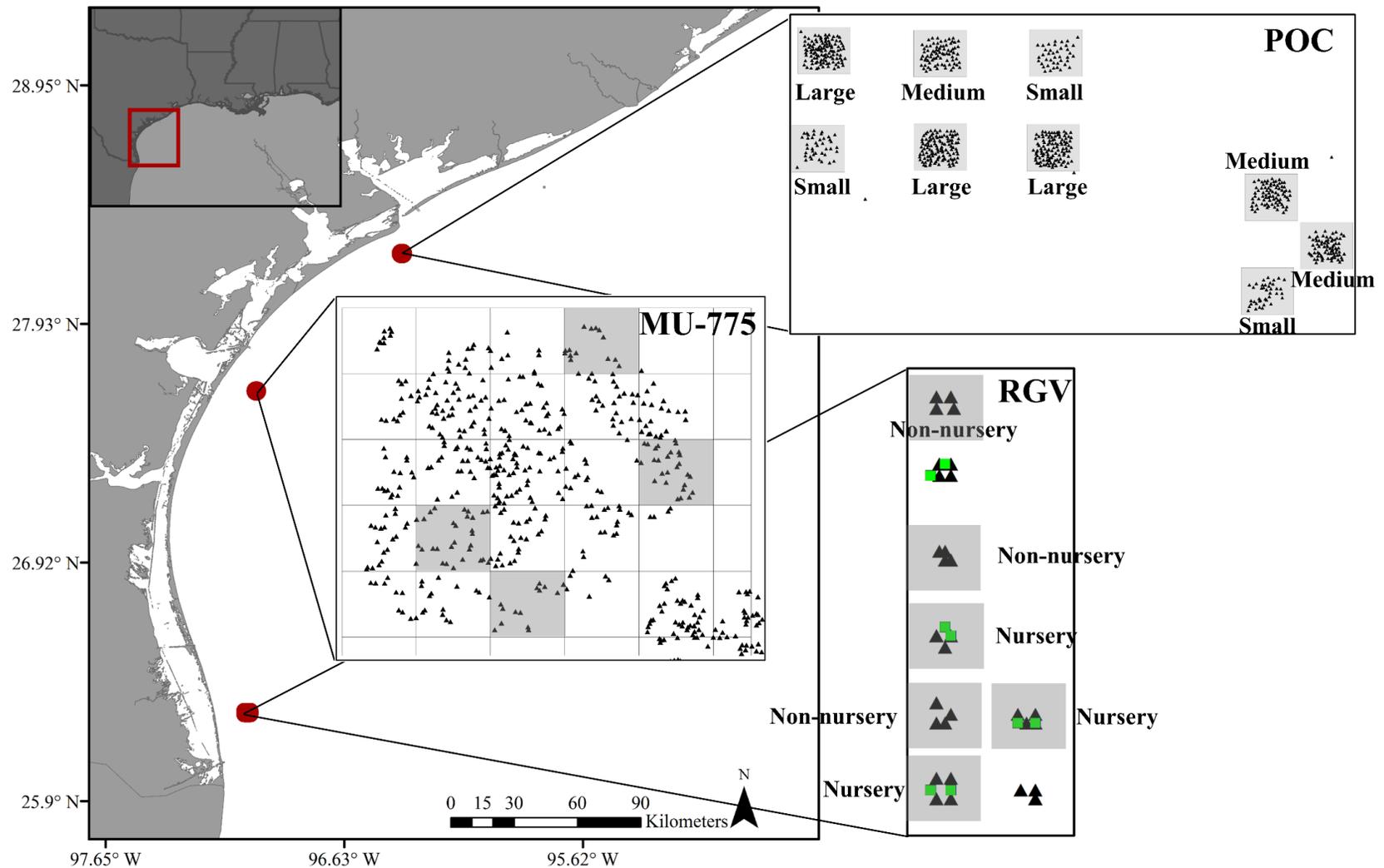


Figure 3-1. Three different sampling locations: Port O'Connor (POC), Mustang Island 775 (MU-775), and Rio Grande Valley (RGV). All prefabricated pyramid points are indicated by solid black triangles and sampling regions are indicated by the gray shaded squares. Also shown are the varying pyramid orientations for POC (small, medium, and large), grids for MU-775, and the nursery (black triangles with green squares) and non-nursery (black triangles) for RGV.

Stereo-RUVS Sampling

Fifty-one stereo-remote underwater video systems (RUVS) surveys were conducted across three sampling locations (Table 3-1). Twelve surveys were conducted at RGV and MU-775 and 27 surveys were conducted at POC (Table 3-1). Uneven sample sizes were due to differing number of pyramid clusters at each site. Furthermore, only two replicates were used at RGV due to tightly grouped pyramids compared to the other two locations.

Table 3-1. Describes the number of pyramids originally placed, the number of pyramids defined (used in calculations), the number of sites, and the number of replicate and total surveys conducted within each reefing location and pyramid orientation.

Location	Pyramid Orientation	Number of Pyramids Placed	Number of Pyramids Defined	Number of Sites	Stereo-RUVS Replicates	Stereo-RUVS Surveys
POC	Small	40	20	3	3	9
	Medium	80	40	3	3	9
	Large	113	57	3	3	9
MU-775	Grid - 1	14	10	1	3	3
	Grid - 2	31	23	1	3	3
	Grid - 3	27	20	1	3	3
	Grid - 4	17	13	1	3	3
RGV	Nursery	4	4	3	2	6
	Non-nursery	4	4	3	2	6

Stereo-RUVS comprised of two GoPro® Hero 4+ cameras mounted 0.7 m apart on a base bar inwardly angled at 4° (Figure 3-2; Harvey & Shortis 1996, Watson et al. 2010, Letessier et al. 2015). The stereo cameras and the optimum measurement field-of-view were calibrated with a standard calibration frame in a similar environment to each of the three sampling locations using VidSync software (Neuswanger et al. 2016). Stereo-RUVS were deployed for 60 min at each site and the first five min were removed from the analysis to prevent visibility and charismatic species bias (Watson et al. 2005). Due to battery-life limitations, only 45 min of the video was analyzed after the allotted five min resting period. Pelagic and benthic stereo-RUVS surveys were conducted simultaneously and arranged on the same line. The pelagic system was

suspended 10-m below the surface and the benthic system typically rested on the seafloor. Two 10-lb weights anchored the two systems, and a surface buoy marked their location.

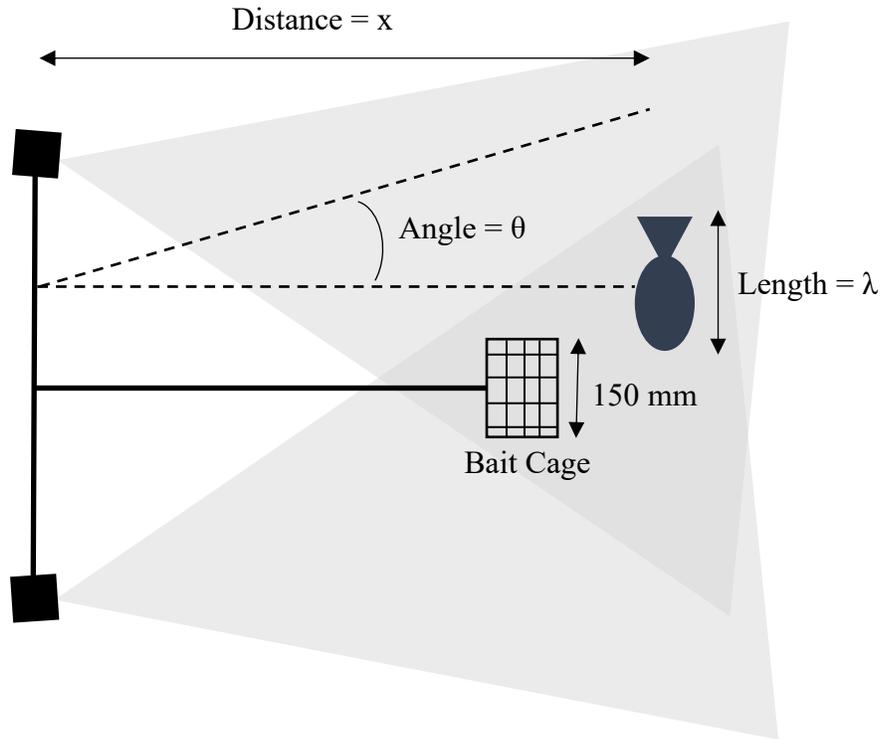


Figure 3-2. Stereo-RUVS configuration, field-of-view (shaded regions), and bait cage that was used to calibrate abundances.

Video Analysis

Video analyses were conducted in the laboratory with two independent readers recording species, count, time, and, if applicable, measurements. All species were time stamped and identified to the lowest possible taxon. Fishes were counted using MaxN (Priede et al. 1994, Ellis & DeMartini 1995), or the maximum number of individuals for each species seen in the frame at any given time. This technique produces a conservative estimate that minimizes the probability of double counting (Streich et al. 2017b). Due to anticipated low visibilities, MaxN was used in favor of MeanCount (Conn 2011) to increase the likelihood of capturing brief encounters with individuals.

If the two readers disagreed on a species identification, a third reader was included. Counts were jointly reviewed if reader counts differed by more than 5%. The counts were then averaged between the two readers for each species and divided by the estimated visibility to remove potential visibility-bias on abundance, which was assumed to be linear. Fork length (FL) was measured to the closest mm for individuals that were visible in both cameras. Fish measurements were calibrated with a calibration frame (Neuswanger et al. 2016) and scaled to a 150-mm wide bait cage suspended in front of the cameras by a 1-m pole (Figure 3-2). The 1-m pole was also used as a reference to estimate visibility. All fish were measured and enumerated using VidSync (Neuswanger et al. 2016).

Species Composition Analysis

I investigated species richness along with diversity using traditional diversity measures such as Shannon's diversity index (H') and Pielou's evenness index (J'). I used a two-way analysis of variance (ANOVA; $\alpha = 0.05$) to test for potential differences in species richness, H' , and J' (Ajemian et al. 2015a) among site locations (POC, MU-775, and RGV) and pyramid orientations (grid, small, medium, large, nursery, and non-nursery). I examined variation in fish community structure among locations and orientations based on a Bray-Curtis similarity matrix of fish abundance using protocol outlined in Peters et al. (2015). The log of abundance (to limit the impact of overly abundant species) was analyzed with an analysis of similarity (ANOSIM) to determine if community structure differed among reefing locations and pyramid orientations. If the analysis resulted any differences in community structure for either test, a similarity percentages analysis (SIMPER) determined which taxa contributed to 70% of the differences observed (Peters et al. 2015). I then used a CLUSTER analysis to visually identify similarities in fish communities among locations and orientations. Unidentified fish were included in the

analysis as an indicator for visibility differences among locations and orientations. All community analyses were run in Primer Version 7.0. All statistical analyses were conducted using the *onewaytests* (Dag et al. 2018) and *vegan* (Clarke 1993) packages in RStudio (R Core Team version 3.5.1; RStudio Team, 2018).

Hydroacoustic Sampling

To determine the abundance and distribution of economically important species, I used video data combined with hydroacoustic surveys using a Simrad EK80 split beam echosounder (70-18CD transducer; circular beam width of 18°, pulse duration = 0.256 μ s, specified ping rate = 300 ms). Nineteen hydroacoustic surveys were completed encompassing all individual sites within each sampling location (Figure 3-1). The EK80 was fixed to a pole-arm and positioned 1.9 m below the surface during transects. Transects over RGV pyramids were conducted in a “flower” pattern centered on the geographic station position of the cluster (Reynolds et al. 2018; Figure 3-3A). At POC and MU-775, where pyramids were not clustered around a single point, transect patterns were in a back-and-forth “mow-the-lawn” pattern (Figure 3-3B). Spatial autocorrelation in hydroacoustic sampling methods was accounted for via ordinary kriging (detailed below). Prior to each survey event, the EK80 echosounder was calibrated using standard methods with a 38.1 mm tungsten carbide sphere (Foote et al. 1987). To allow for integration of the species composition with hydroacoustic data, further post-processing of the video dataset was required. Species unlikely to be detected acoustically due to size, morphology (e.g., damselfishes, sharks, etc.), cryptic behavior, or close association within the benthic fouling, or on the structure were removed prior to further analysis (Simmonds and MacLennan 2005; Wilson et al. 2003, 2006).

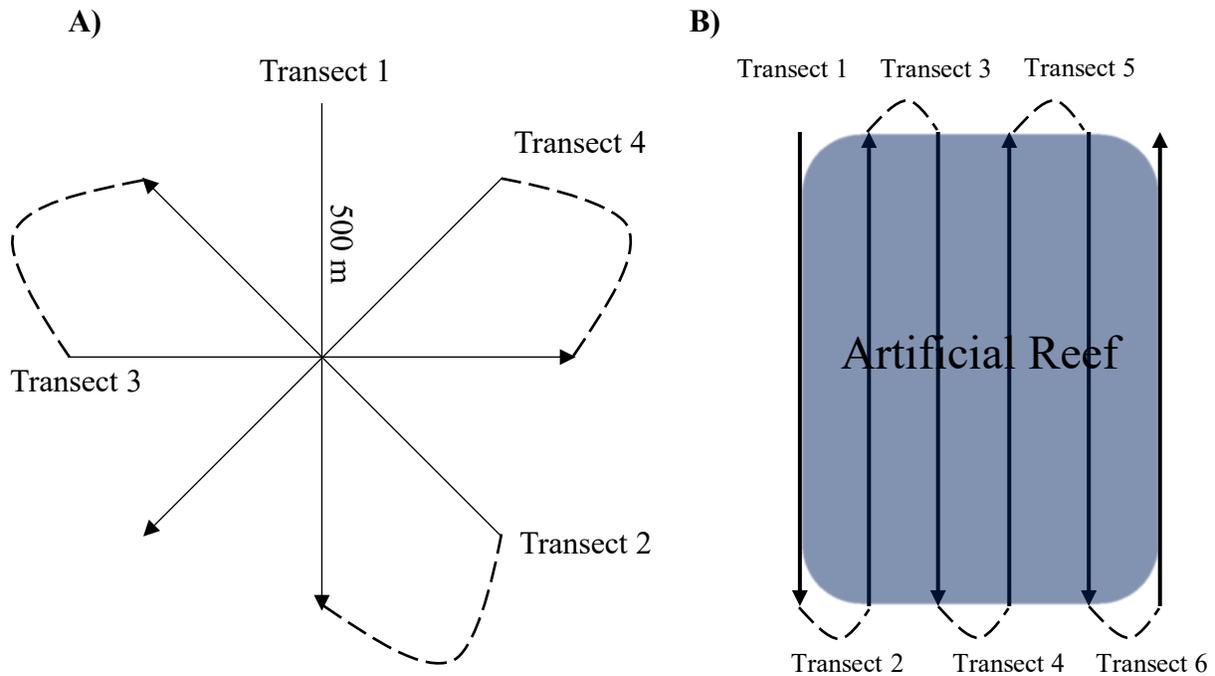


Figure 3-3. Diagram representing the two hydroacoustic tow patterns. A) Example of a flower-pattern survey where the four, 250-m echosounder transects were centered over the geographic station position for each site at RGV. B) Example of a back-and-forth sweeping “mow-the-lawn” survey pattern used at MU-775 and Port O’Connor sites.

Hydroacoustic Analysis

Raw hydroacoustic data were processed in the lab using Echoview 11.0 (SonarData Pty Ltd., Hobart, Australia). Prior to analyzing, data were edited to exclude unwanted noise and reverberation (Simmonds & MacLennan 2005, Boswell et al. 2007). Excluding noise reverberation included excluding 0.5 m from the seafloor and 1 m around artificial structures. Echo integration approaches were used when individual fish were too closely distributed for echo counting to be successful (Boswell et al. 2007). To account for spatial autocorrelation, ordinary kriging was applied to each layer using the following steps. First, a variogram, using initial values calculated from the data, was used to model the spatial continuity of the data collected at each site. Second, a 5 x 5-m survey grid was projected over each depth bin. Third, ordinary kriging interpolated spatially weighted estimates of area density over the survey grid

from the predicted variogram. This resulted in depth layer-specific hydroacoustic density estimates for 5 x 5 x 5-m cells arranged over the area bounded by the ends of the transects performed at each site.

Fish abundance in each cell in each depth layer was calculated by multiplying the density by 25 m² (the area of the cell). Depth layer-specific abundance was then calculated by summing the abundances from each cell. Species-specific abundances at each site were obtained by taking the product of abundance and percent species composition for the associated depth layer. Species composition from benthic stereo-RUVS were applied to the bottom-most layer, or the layer representing the bottom 5 m from the seafloor. Species composition from the pelagic surveys were applied to the summed abundance from the remaining layers. Species-specific abundances were then summed across depth bins to produce a species-specific abundance at each site. Because hydroacoustic methods and, as a result, survey areas, varied significantly among reefing locations, densities (in fish per m²) were used in lieu of abundances for location and orientation comparisons.

Cost-Benefit Ratios

Two different cost-benefit ratios were calculated to represent the ecological and human (fisheries) benefits of the reef site. The first ratio was an overall cost-benefit ratio, which was calculated by taking the overall fish density (fish per m²) and dividing by the number of pyramids for each site:

$$(1) \quad \text{Overall Cost: Benefit Ratio}_s = \text{Pyramid Cost} \div \left(\frac{TD}{npyr} \right)_s$$

Where TD is total fish density (fish/m²) and $npyr$ is the number of pyramids for each sampled site s .

The second cost-benefit ratio incorporated three economically important species along the Texas coast: Red Snapper, Greater Amberjack, and Gray Triggerfish. Vermilion Snapper and Almaco Jack were not observed at any of the nearshore pyramid locations and were not included in the analysis. Each species was enumerated and weighted based on their respective commercial value (Table 3-2). I applied weightings due to the high importance of Red Snapper compared to the other species being investigated (Eq. 2). Furthermore, because recreational value is uncertain and difficult to measure for all the species in question, I assumed that the commercial weighting was representative of the general economic weighting for each species (Eq. 2). As future economic valuation studies become available, they could be expanded based on these findings.

$$(2) \quad EIS \text{ Cost: Benefit Ratio}_s = Pyramid \text{ Cost} \div \sum_1^i \left(\left(FL_i \cdot (FLtoW_i) \cdot \left(\frac{D_i}{npyr_s} \right) \cdot \left(\frac{value_i}{Landings_i} \cdot \frac{value_i}{\sum_1^i value_i} \right) \right) \right)_s$$

Where *FLtoW* corresponds to fork length (mm) to whole weight (lb) conversion, *D* is fish density (fish/m²), *npyr* is the number of pyramids, *i* represents each economically important species, and *s* indicates each site sampled. In essence, a cost-benefit ratio was assigned to each site based on the total weight of each economically important species. This weight was calculated based on FL, or an averaged FL if no measurement was available, using FL-to-whole weight conversions from the most recent stock assessment for each species. The weight was then multiplied by the weighted value for each species and then summed across sites to produce a value (in US\$) per pyramid for each site (Table 3-2). I used the estimate presented by Shively (2014) of \$1,600 to reef one pyramid and divided by the density per pyramid. I tested for differences in cost-benefit ratios among reefing locations and pyramid orientations using a two-way ANOVA ($\alpha = 0.05$).

Table 3-2. Commercial value in US\$ of each economically important species and weightings based on the total value. Values are based on 2020 Texas commercial landings (fisheries.noaa.gov).

Species	2020 TX Commercial Value (\$)	Weighting	2020 Price per Pound (\$/lb)	Weighted Price per Pound (\$/lb)
Red Snapper	12,176,300	9.96E-01	4.42	4.403E+00
Greater Amberjack	45,695	3.74E-03	2.06	7.69E-03
Gray Triggerfish	527	4.31E-05	1.59	6.86E-05
Total	12,222,522			

Results

Over 38 hours of video for both benthic and pelagic surveys (76 hours total) were collected and processed. One benthic survey had to be removed from the analyses due to improper weighting causing the system to be greater than 5 m above the seafloor. Of the 101 total surveys analyzed, 31 yielded a result of “no fish”. In the remaining surveys, 1,334 individual fish were counted (596 for benthic surveys; 739 for pelagic surveys). About 15% of fish (n = 203) were counted as unidentified primarily due to limited visibility on benthic surveys. Most of the unidentified fish were counted at MU-775 (n = 183). Overall, I observed 25 different species representing 18 families. Eighty-six individuals were measured across all three sampling locations. The most measured fish were Gray Triggerfish (n = 20; \bar{x} = 247 mm FL), Crevalle Jack (*Caranx hippos*; n = 19; \bar{x} = 566 mm FL), and Red Snapper (n = 18; \bar{x} = 243 mm FL). Only Atlantic Spadefish (*Chaetodipterus faber*) were measured at all three sampling locations.

Table 3-3. Results from the two-way ANOVA testing the effect of reefing location and pyramid orientation on species richness, Shannon’s diversity index, and Pielou’s evenness index, where *df* refers to degrees of freedom.

Index	Parameter	<i>df</i>	<i>F</i>	<i>p</i>
Species Richness	Location	2	1.046	0.379
	Pyramid Orientation	3	0.7	0.569
	Residuals	13		
Shannon's Diversity Index	Location	2	0.683	0.523
	Pyramid Orientation	3	0.676	0.582
	Residuals	13		
Pielou's Evenness Index	Location	2	0.483	0.628
	Pyramid Orientation	3	0.538	0.664
	Residuals	13		

Species richness and the two diversity indices did not vary significantly with reefing location nor pyramid orientation (Table 3-3; Figure 3-4). Conversely, results from the ANOSIM revealed significant separation in fish communities among site locations ($R = 0.195$, $p = 0.030$) and pyramid orientations ($R = 0.241$, $p = 0.022$). The largest difference in fish communities occurred between MU-775 and RGV ($R = 0.698$, $p = 0.005$). These groupings were reflected in the corresponding non-metric multidimensional scaling (NMDS) plot (Figure 3-5). Additionally, CLUSTER analysis revealed similar groupings, where all MU-775 surveys were separated from RGV surveys (Figure 3-6). The CLUSTER analysis also indicated little overlap between the POC and RGV surveys (Figure 3-6). The species contributing 70% of the separation observed between MU-775 and RGV were unidentified fish, Blue Runner, Atlantic Spadefish, Crevalle Jack, Red Snapper, Greater Amberjack, and unidentified scad (Figure 3-5). Correspondingly, Blue Runner, Gray Triggerfish, Greater Amberjack, Red Snapper, unidentified scad, unidentified fish, Redear Herring, and Crevalle Jack contributed 70% of the observed differences between POC and RGV. Along with differences between MU-775 and RGV locations, which were

indicated by significant separation between “grid” and “nursery” and “grid” and “non-nursery” orientations, pyramid orientation factor tests also revealed significant separation between “grid” and “medium” orientations (Figure 3-5). Unidentified fish, Atlantic Spadefish, Red Snapper, and Blue Runner contributed 70% of the observed differences between “grid” and “medium” orientations.

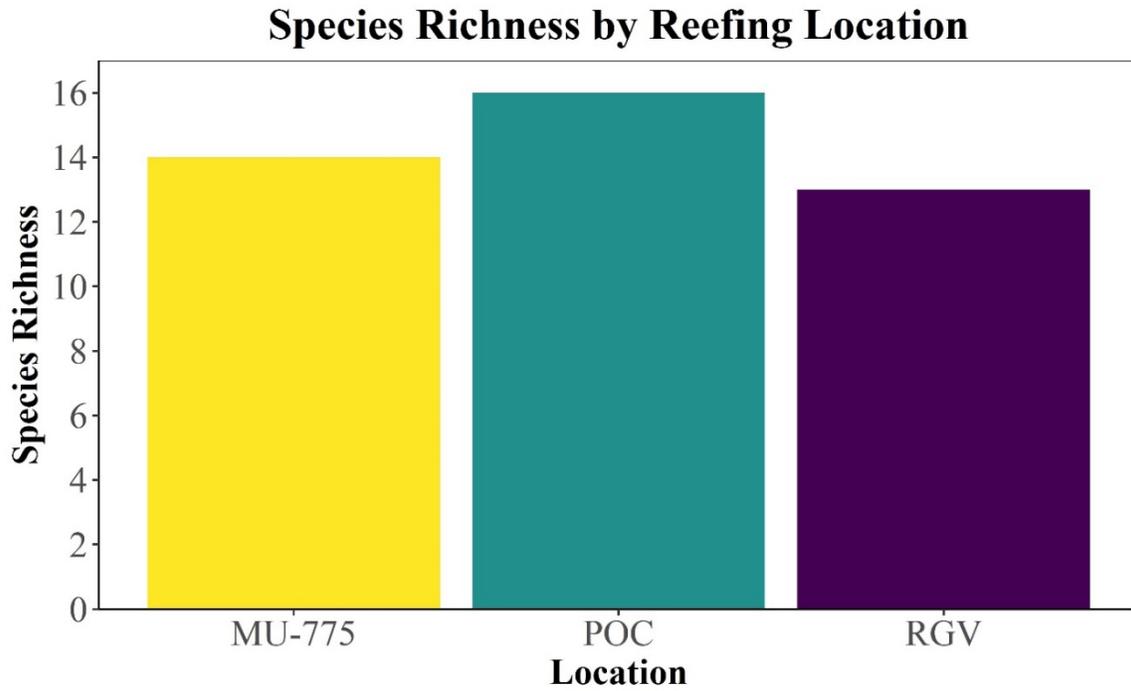


Figure 3-4. Bar plot showing the overall species richness for each reefing location.

Fish Community NMDS Plot

Resemblance: S17 Bray-Curtis similarity

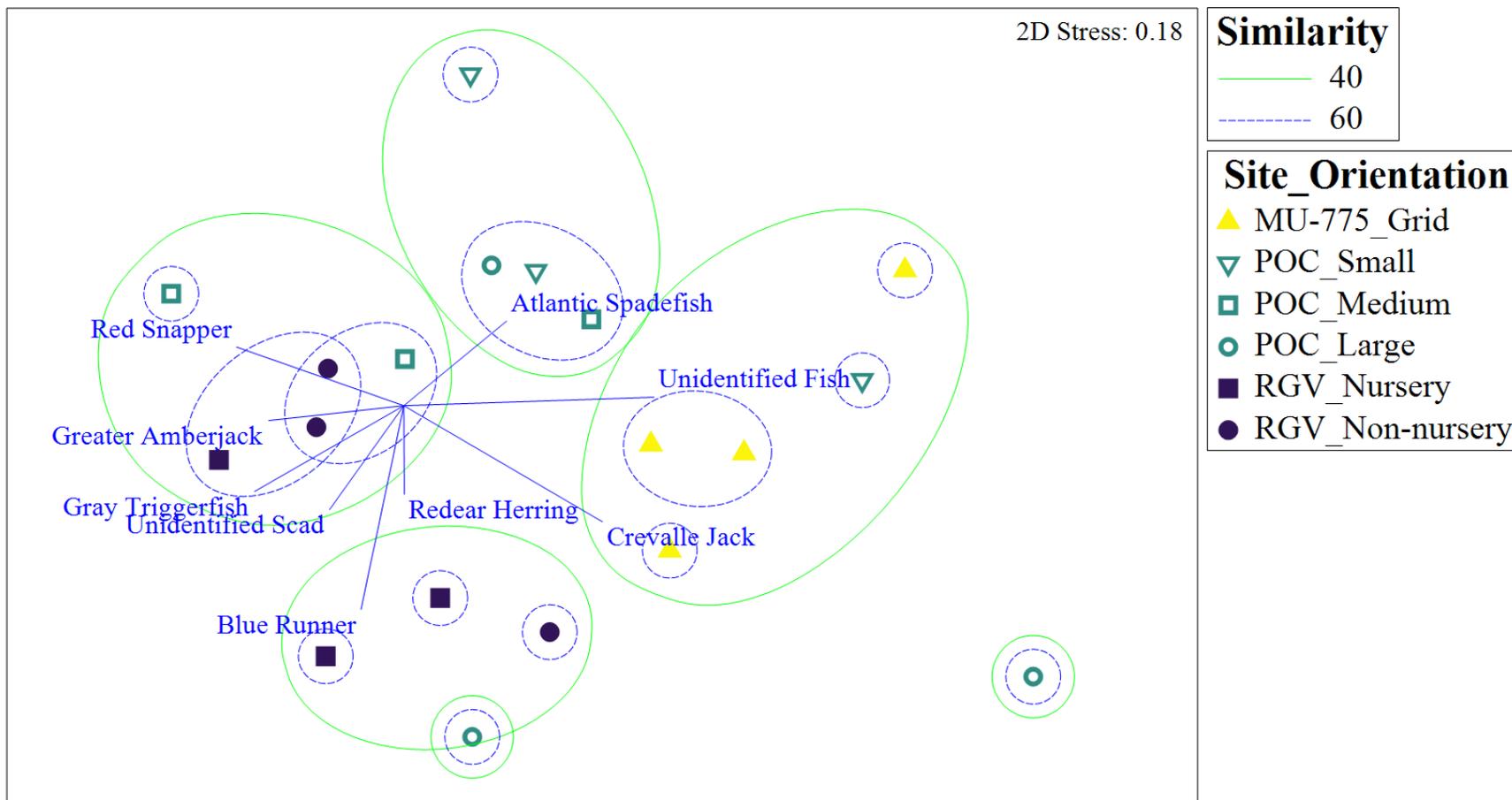


Figure 3-5. Non-metric multidimensional scaling (NMDS) plot of fish communities from stereo-RUVS surveys. Differing colors indicate reefing locations and differing symbols indicate pyramid orientations. Species vectors are based on Pearson correlation coefficients. Percent similarity bubbles (from CLUSTER) represent 40% (green) and 60% (blue) similarity. *Site_Orientation* refers to the variable composed of reefing location and pyramid orientation.

Fish Community Dendrogram

Group average

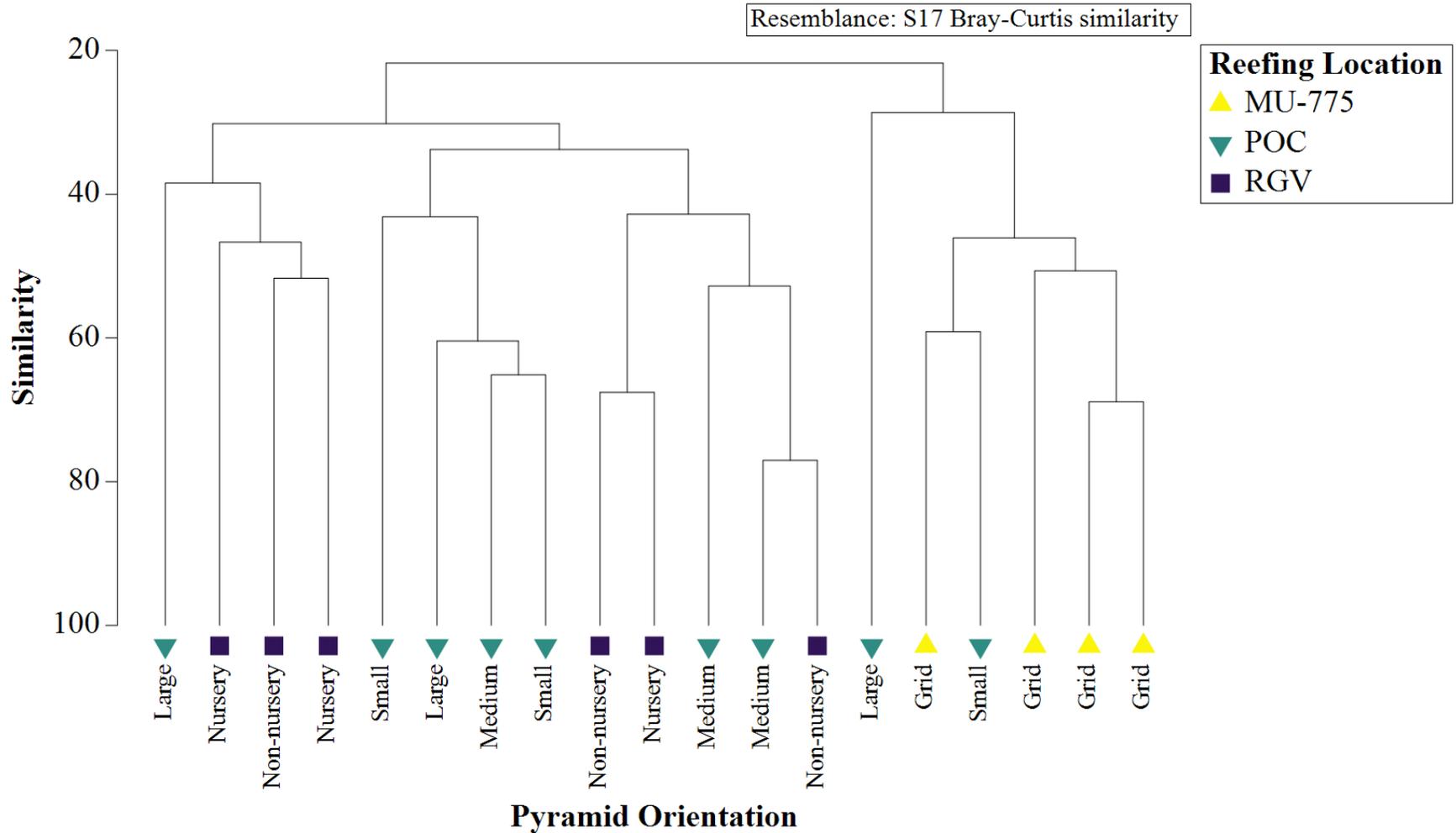


Figure 3-6. CLUSTER dendrogram exhibiting linkages of fish communities by site location and pyramid orientation. Linkages are based on Bray-Curtis similarity matrix of natural log abundances.

Stereo-RUVS and Hydroacoustics

I observed a total of 4,045 fish from hydroacoustic surveys across all three reefing locations (Table 3-4). Total fish density (fish per m²) was highest at POC-medium and lowest at MU-775-Grid-4 (Table 3-4). A two-way ANOVA testing for differences in total fish density with site location and pyramid orientation resulted in no significant difference in site location ($F_{2,13} = 1.654, p = 0.229$) nor with pyramid orientation ($F_{3,13} = 0.622, p = 0.613$). Fish density exhibited a non-linear relationship with pyramid density at each survey location (Figure 3-7). Furthermore, fish densities appeared to be relatively similar across all sampled sites apart from MU-775 Grid-2 (Figure 3-5) and were not significantly correlated ($\rho = 0.212, p = 0.381$) using Spearman's rank correlation coefficient.

Table 3-4. Total fish abundance, survey area (m²), and fish density (fish/m²) by reefing location and pyramid orientation.

Location	Pyramid Orientation	Total Abundance	Survey Area (m²)	Density (fish/m²)
POC	Small	303	16,195	0.019
	Medium	633	16,195	0.039
	Large	626	16,195	0.039
MU-775	Grid - 1	69	4,346	0.016
	Grid - 2	184	4,961	0.037
	Grid - 3	52	4,846	0.011
	Grid - 4	50	5,059	0.010
RGV	Nursery	1113	28,972	0.038
	Non-nursery	1015	28,972	0.035

Fish Density as a Function of Pyramid Density

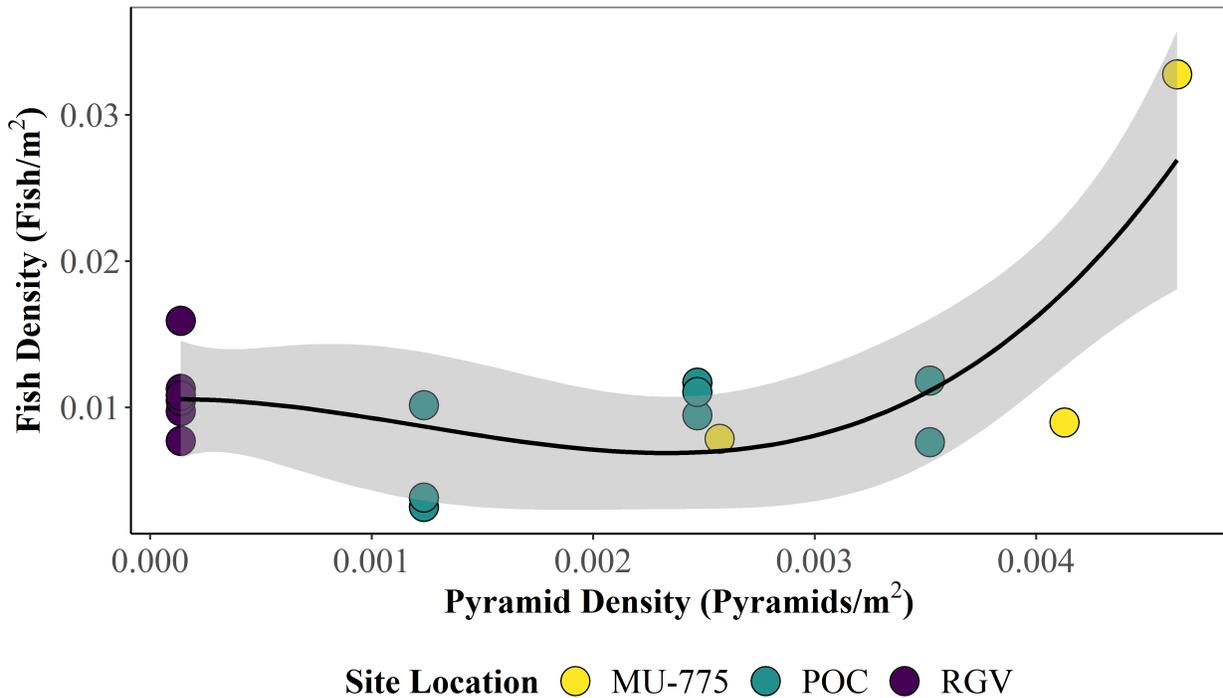


Figure 3-7. Overall fish density (fish/m²) as a function of pyramid density (pyramids/m²). Black line indicates a smoothed regression line using spline interpolation. Shaded region represents a 95% confidence interval. Color differentiates reefing locations.

The most abundant species was Red Snapper ($n = 1,299$), followed by Gray Triggerfish ($n = 906$), Blue Runner ($n = 460$), and Greater Amberjack ($n = 427$). The most densely populated pyramid orientation and species were Atlantic Spadefish at MU-775-Grid-1 (density = 0.016 fish/m²) and the least dense were Unicorn Filefish at RGV-Nursery (density = 2.12×10^{-5} fish/m²). Red Snapper density varied from 0.0007 to 0.0109 fish/m² and slightly increased with increasing pyramid density ($\rho = 0.379$, $p = 0.148$; Figure 3-8).

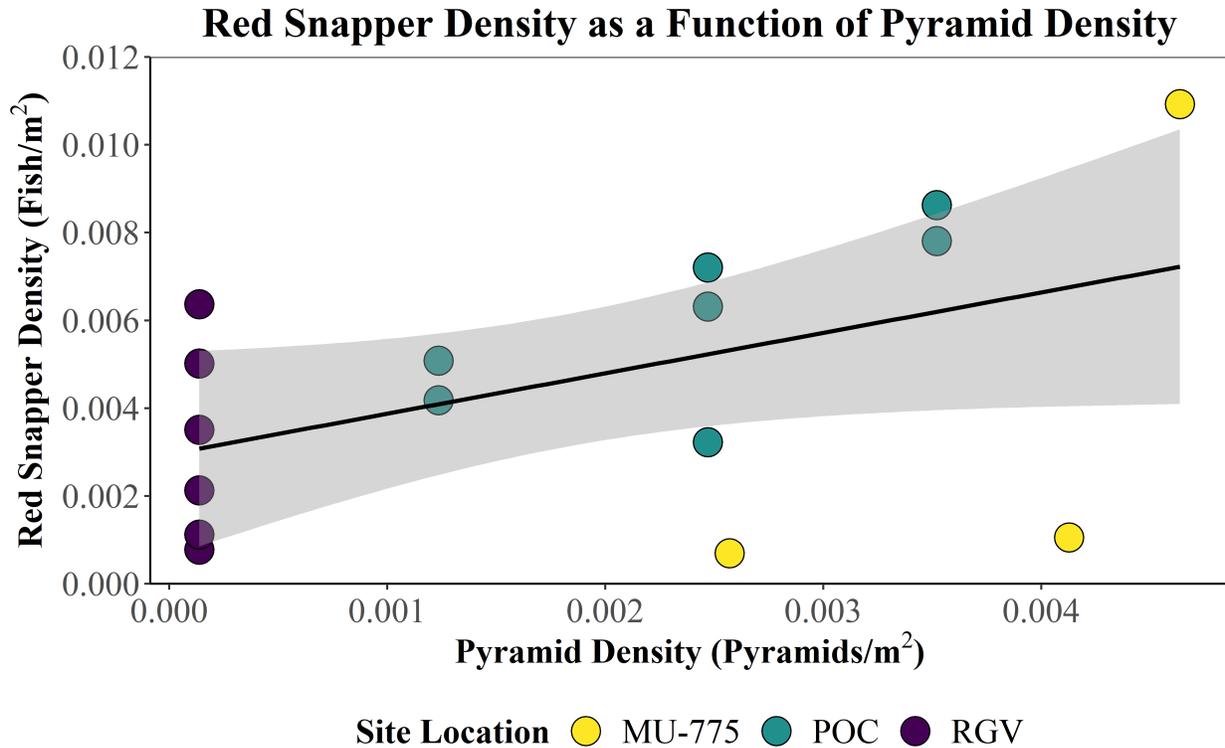


Figure 3-8. Red Snapper density (fish/m²) as a function of pyramid density (pyramids/m²). Black line indicates a linear regression line. Shaded region represents a 95% confidence interval. Color differentiates reefing location.

The overall cost-benefit ratio varied significantly with site location ($F_{2,13} = 8.463, p < 0.01$), but not with pyramid orientation ($F_{3,13} = 0.572, p = 0.644$). Rio Grande Valley reef had the lowest overall cost-benefit ratio and POC was drastically the highest (Figure 3-9). Neither site location ($F_{2,11} = 3.037, p = 0.080$) nor pyramid orientation ($F_{3,11} = 0.346, p = 0.793$) had a significant effect on EIS cost-benefit ratio (Figure 3-10). The three locations with the best, or lowest, economically important species cost-benefit ratios were all at RGV (Figure 3-10). Additionally, Grid-1 at MU-775 and one of the “Large” orientations at POC did not contain any economically important species being discussed and were thus excluded from the visual representation of the data.

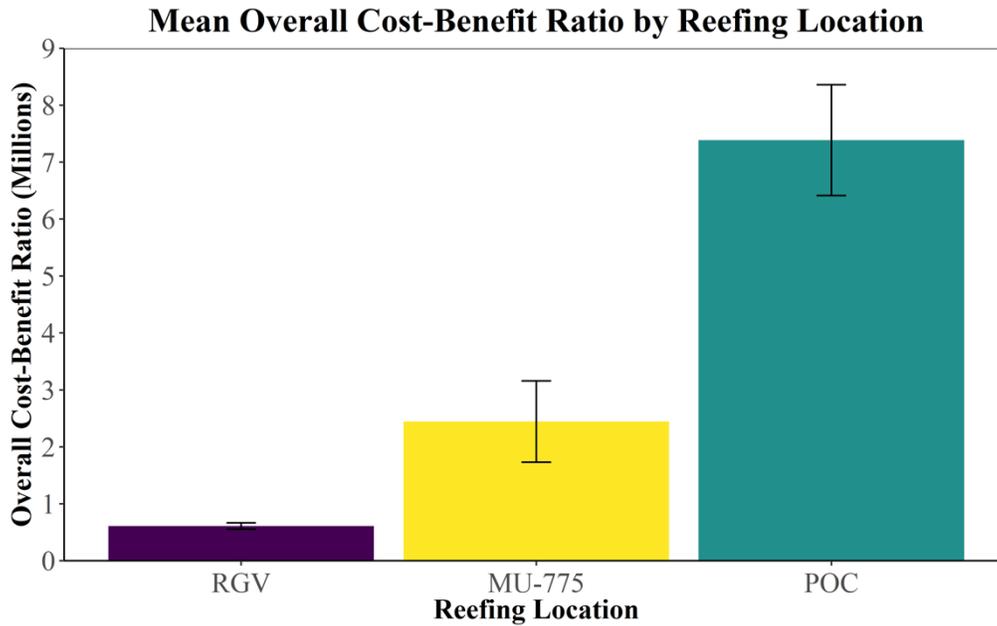


Figure 3-9. Mean overall cost-benefit ratio by reefing location scaled in the millions. Lower values indicate a more cost-effective reef, while higher values indicate a less cost-effective reef.

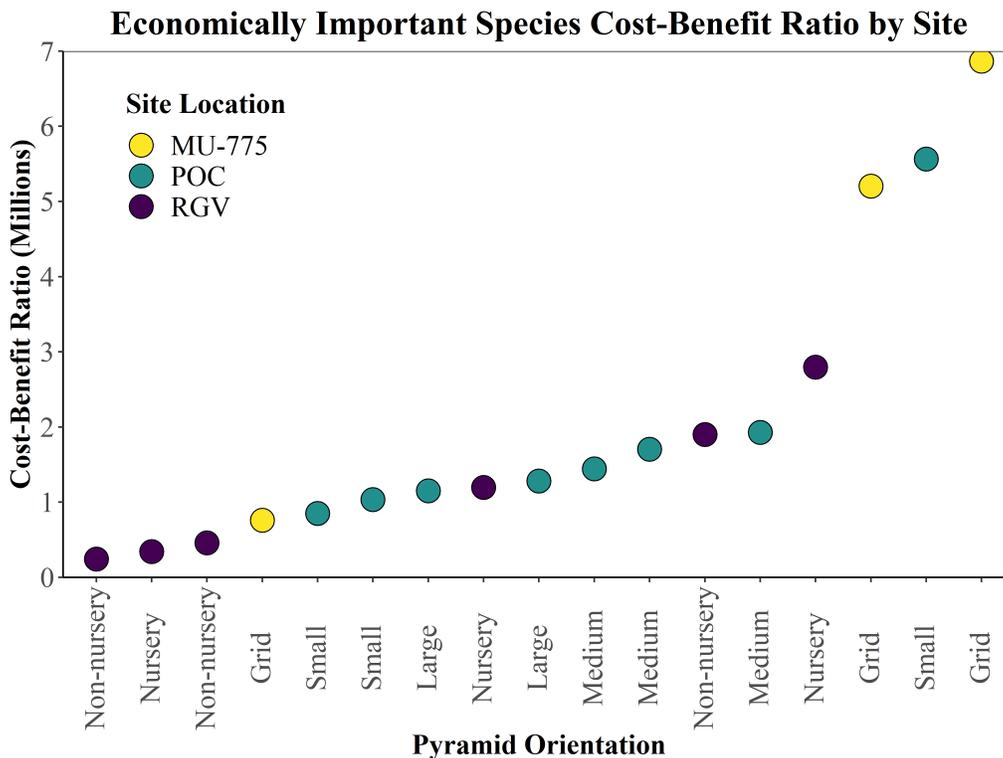


Figure 3-10. Economically important species cost-benefit ratios for each site arranged from smallest to largest, where the unique colors correspond to the different site locations and the labels on the x-axis represent the different pyramid orientations. Lower values indicate a more cost-effective reef, while higher values indicate a less cost-effective reef.

Discussion

This study documented reef fish communities of primarily young individuals inhabiting prefabricated pyramid nearshore artificial reefs of varying locations and orientations along the Texas coast. The presence of younger cohorts suggests the potential for these reefs to act as nurseries for several EIS, but more data are needed to confirm these results. This study identified fish community differences among reefing locations and orientations. By combining stereo-video and hydroacoustic surveys, this study developed a unique approach to calculating cost-benefit ratios that could be adopted for future studies. Furthermore, I found that POC was the least cost-effective reef but, due to its design with science in mind, presented the clearest results. Although the best reefing design remains unclear, this study presents different ways to examine reef efficacy and provides recommendations for future reefing projects.

Species Richness and Abundance

Although prefabricated pyramids appear to house a relatively robust fish community, the number of species observed ($n = 25$) and Red Snapper density were lower than the number of species observed in other studies using similar methods along the Texas coast (Rooker et al. 1997, Ajemian et al. 2015a,b, Streich et al. 2017a). Differing survey methods or, more likely, the difference in reef types and bottom depth of artificial reefs surveyed could explain the discrepancy in species richness and Red Snapper density. Many studies mentioned previously examined larger artificial reefs such as ships and both reefed and standing oil and gas platforms. Unlike natural banks, which are spread over large areas, artificial reefs have a relatively small 2-dimensional footprint and thus, species richness and fish density is dependent on the vertical relief of the structure (Ajemian et al. 2015a). Larger artificial reefs, such as ships or platforms, offer greater vertical relief than smaller artificial reefs like prefabricated pyramids (Ajemian et

al. 2015a, Streich et al. 2017b). More specifically, prefabricated pyramids offer only about 2.4 m of vertical relief compared to 10-30 m of vertical relief from ships and reefed platforms.

Furthermore, a recent study conducted at POC found that the pyramids had been buried deeper into the sediment by more than 1 m due to Hurricane Harvey (Kline and Moody 2019), thus further reducing the vertical relief.

Despite the smaller verticality, prefabricated pyramids in nearshore regions may be important for younger cohorts. This study commonly observed smaller carangids, lutjanids, and other species occupying these reefs potentially using the pyramids protection. Prefabricated pyramids are embedded with limestone and are perforated with holes in which small fishes can swim in to seek refuge from predators. Although classifying nursery habitat was outside the scope of this study, the nearshore artificial reefs in my study likely serve as important habitat for juvenile Red Snapper and Greater Amberjack based on visual observations and measurement data. A recent study similarly found that MU-775 functions as nursery habitat for juvenile Red Snapper (Streich et al. 2017a), suggesting the possibility that POC and RGV could also play a similar role for juvenile Red Snapper. Furthermore, I did not observe any Red Snapper that were harvestable based on identifiable markings (i.e., dorsal spot) and current length-age relationships for this species (Fischer et al. 2004, SEDAR 2016). Conversely, larger artificial reefs in deeper waters are typically characterized by higher abundances of harvestable sizes of Red Snapper (Stanley & Wilson 2000, 2003, Gallaway et al. 2009, Ajemian et al. 2015a, Streich et al. 2017a,b, Gallaway et al. 2021, Stunz et al. 2021). Similarly, all Greater Amberjack measurements in this study were less than 210 mm FL, which is well below the minimum size limit for this species in Texas state waters (tpwd.texas.gov). Although data are generally lacking on localized movements of juvenile Greater Amberjack and on ontogenetic shifts in habitat,

young-of-year carangids have been observed using floating sargassum patches as nurseries (Wells & Rooker 2004, Stachowiak 2020). I also observed young-of-year carangids hovering around pelagic cameras for an extended period, which could indicate that their use of floating objects is not limited to sargassum patches and that they potentially exhibit positive attraction to suspended sampling gear. Despite the widely accepted notion that jacks are highly mobile fishes (Burch 1979, McClellan & Cummings 1997, Ingram & Patterson 2001), recent evidence suggests that Greater Amberjack may experience a high degree of site fidelity (Baeyaert et al. 2018). While my data indicate the potential for these prefabricated pyramid reefs to act as nurseries for carangids and lutjanids, more data are needed on the density of these juveniles as well as the degree of residency compared to surrounding areas.

Alternative to the nursery hypothesis, the prevalence of young/small fish in the length-frequency composition could also be due to persistent fishing of these nearshore and easily accessible harvestable size-classes. The reefing locations sampled in this study are all less than 20 km from the nearest inlet and are well-known and widely publicized fishing "hotspots" (tpwd.texas.gov, Priest 2018). Because these nearshore reefs are located in state waters, recreational harvest of Red Snapper and other species over a specified size limit occurs year-round with liberal bag limits. Additionally, I sampled during months that are typically most popular for recreational anglers on the Texas coast (Ditton et al. 1978). Because all three reefing locations were similar distances from shore and from the nearest inlet and all areas have similar fishing capacity, I assumed that fishing effort would be similar. Future studies should include year-round sampling to gauge changes in density of EIS with seasonal differences in fishing pressure.

Bottom depth is likely the primary reason for the presence of younger life stages in my study compared to other studies. Pyramids have been observed harboring larger individuals of Red Snapper compared to other low-relief nearshore artificial reefs (Harrison 2015, Jaxion-Harm & Szedlmayer 2015). However, much of the adult Red Snapper population typically resides in depths greater than 35 m (Gallaway et al. 2009, Harrison 2015, Jaxion-Harm & Szedlmayer 2015). Red Snapper and other reef fishes exhibit ontogenetic movements to suitable habitats across a depth gradient (Eggleston 1995, Gallaway et al. 2009, Hujibers et al. 2015, Dance & Rooker 2019). Particularly, Red Snapper exhibit shifts with age from inshore (20 m depth) to offshore (100 m depth) across the Texas continental shelf (Dance & Rooker 2019). However, the Great Red Snapper Count, a Gulf-wide study dedicated to estimating the abundance of the Red Snapper population, showed large populations do occur nearshore, which is likely a function of additional sampling and identification of suitable habitat. Resolving the mechanisms behind these abundance patterns is limited by serious data gaps of on-shelf nearshore habitat features.

Fish Communities

This study observed significant differences in fish communities among reefing locations. Differences in fish communities between MU-775 and RGV could be due to differing years since reef construction. For example, the sampled locations for MU-775 were reefed in 2013, whereas the sampled locations for RGV were reefed in 2017. Fish communities tend to grow and adapt over time following the establishment of an artificial reef until reaching an equilibrium (Walsh 1985, Bohnsack 1991, Bohnsack et al. 1994). However, peak colonization of artificial reefs can occur in as little as two months from reef establishment (Walsh 1985, Bohnsack 1991, Bohnsack et al. 1994, Cummings 1994, Streich et al. 2017b). Additionally, a study conducted at MU-775 in 2015 observed peak abundances of Red Snapper and Gray Triggerfish at 11- and 23-months post

reef construction, respectively (Streich et al. 2017b). Because this study was conducted at least 72 months after construction for all sampled locations, I suggest observed differences among locations are not likely to be due to differing timelines since construction. Alternatively, these results suggest that community differences may be influenced by other factors such as visibility and, by association, water quality, latitudinal gradients in species composition, or proximity to other natural or artificial reefs.

Due to a persistent benthic nepheloid layer observed in much of the western GOM (Shideler 1981, Ajemian et al. 2015b, Streich et al. 2017b), visibility was never greater than 2 m at benthic surveys. Benthic visibility was also found to significantly vary among reefing locations, with POC experiencing the lowest visibilities. Furthermore, the inclusion of unidentified fish as a contributing species indicates that visibility may have contributed disproportionately to the observed community differences among reefing locations and pyramid orientations. Although I accounted for the effect of visibility on species abundance through the scaling of counts, the effect of visibility on community structure remains a focal area where more research is needed to resolve limitations. For example, inconspicuous or cryptic species would have been harder to capture at POC and could have influenced community results. In general, only a few cryptic or smaller reef fish species were observed on benthic surveys including juvenile Tomtate (*Haemulon aurolineatum*), Bandtail Puffer (*Sphoeroides spengleri*), Frillfin Goby (*Bathygobius soporator*), and Harlequin Bass (*Serranus tigrinus*), indicating that this study may have been biased towards charismatic or larger individuals due to overall poor benthic visibility.

Along with visibility, other water quality parameters such as nitrogen, phosphorous, dissolved oxygen, or chlorophyll could have also influenced fish communities. Nitrogen and

phosphorous have been previously observed influencing fisher catch-rate success in North Carolina sounds (Kaoru & Smith 1990, Kaoru 1991). Furthermore, while more severe hypoxic conditions are experienced around the Mississippi delta (Hetland & DiMarco 2008), hypoxic events also occur on the Texas continental shelf (Harper et al. 1981, Harper et al. 1991, Rabalais & Turner 2001, Hetland & DiMarco 2008), which can lead to significant changes in fish community structure (Craig & Bosman 2013). Measuring the impact of these factors was beyond the scope of this study; however, incorporating water quality parameters in future studies could lead to more predictive approaches to evaluating fish community differences.

The largest difference in communities did not occur between the two locations with the largest latitudinal difference (POC and RGV). Although visibility and water quality differences are also likely explanations for the observed differences in fish communities, latitudinal shifts in fish community composition are well-documented and could explain some of the observed differences (Ebeling & Hixon 1991, Rohde 1992, Hobson 1994, Briggs 1995, Floeter et al. 2004). Typically, fish species diversity decreases from tropical to temperate latitudes (Ebeling & Hixon 1991, Hobson 1994, Briggs 1995, Floeter et al. 2004). While species richness, diversity, or evenness did not significantly vary among reefing locations (latitudes), my study was conducted over a relatively small spatial scale compared to other studies where this pattern was noted (Ebeling & Hixon 1991, Rohde 1992, Briggs 1995, Floeter et al. 2004).

Although latitude and visibility are important to the observed community differences, RGV results may have been confounded by the presence and close proximity of a large number of other structures at this location. The RGV reef complex is comprised of many structures and material apart from the ones surveyed in this study. These structures are often within a few hundred meters of one another and the entire complex is roughly 6.5 km², which more than likely

influenced the fish communities that were observed. Overall, results from this study indicate that reefing location was the main driver behind these differences despite the many other factors that may have contributed to the differences observed in fish communities.

Future Reefing Recommendations

This study developed a unique approach to calculating cost-benefit ratios at artificial reefs. Rather than quantifying a single economically important species (Streich et al. 2017a), all relevant economically important species were incorporated into the analysis. This study also recognized the significance of certain species (i.e., Red Snapper) and applied weightings to represent species-specific contributions in the Texas commercial fishing industry. Future studies should consider weighted approaches when incorporating EIS densities in cost-benefit analysis.

Despite having the most pyramids, POC's efficacy was the lowest among all three locations for overall cost-benefit ratio. As a result, if the objective of an artificial reef is to accommodate an abundant fish community, smaller clusters of pyramids or a pyramid-field is preferred to large, high-density reefs. However, POC did not display low efficacy regarding EIS. In fact, Red Snapper density increased with increasing pyramid density at POC sites. The large dispersal between reefed sites within POC coincides well with Red Snapper home ranges and potentially increases prey diversity, reduces fishing pressure, and reduces conspecific competition (Brandt & Jackson 2013, Froehlich et al. 2019). The orientation of POC likely suits dominant predators, like most EIS in the GOM (Ouzts & Szedlmayer 2003, Froehlich & Kline 2015), and should be considered if the objective of the reef is focused on predatory fish species. Following the example of POC reef, I also recommend constructing artificial reefs with science in mind to evaluate the efficacy of different reef configurations more effectively.

Economically important species, particularly Red Snapper, rely on the presence of larger structures or natural banks to transition to beyond the first few years of age (Gallaway et al. 2009). Because all Red Snapper observed in this study appeared to be within their first few years of age, I recommend that prefabricated pyramid reefs, and perhaps even smaller lower relief structures where Red Snapper are known to occur, are located within reasonable distances of other, larger artificial reefs or natural banks. Proximity to other reefs may guide ontogenetic shifts in habitat to maintain healthy populations of harvestable individuals available to recreational and commercial fishers.

In summary, this study exemplifies the importance of lower relief reefs as important habitat for younger cohorts and illustrates the effect of location and pyramid orientation in fish communities. While many factors may have contributed to some of the observed differences in fish communities, this robust design found reefing location to be the main driver. Future studies should examine community structure at similar artificial reefs along a depth gradient (i.e., nearshore to offshore) to examine interspecies habitat partitioning. Year-round sampling should also be considered to determine how seasonal changes in water temperature, salinity, and, more importantly, recreational fishing intensity impact fish communities. I also recommend that side-scan sonar be incorporated into future studies to determine exact density, location, and surface area of artificial structures. Port O'Connor nearshore reef was considered the least cost-effective reefing location in terms of overall fish density and economically important species density; however, POC is a good example of a reef designed to produce replicable results. This study provides examples of cost-effective artificial reefs and highlights the importance of developing reefs with science in mind to evaluate reef benefits more competently.

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CHAPTER IV:

REEFING QUANTITY MATTERS FOR FISH COMMUNITIES: A CASE STUDY FOR CLEAR OBJECTIVES FOR ARTIFICIAL REEF CONSTRUCTION

Abstract

Artificial reefs enhance natural resources, particularly in areas where structured habitats are limited, and serve a variety of economic benefits from sport diving to recreational and commercial fishing. Evaluating the ecological benefits of an artificial reef is vital for determining the relative success of the reef. Furthermore, when the ecological benefit is balanced with the cost associated with artificial reef implementation, return on investments in the project can be maximized and can inform efficiency metrics for future reefing projects. The purpose of this study was to examine fish communities at different reefing materials and quantities at Rio Grande Valley reef in the northwestern Gulf of Mexico to determine their efficacy. I used a combination of stereo-remote underwater video systems and hydroacoustic surveys to assess fish community differences and calculate two different types of cost-benefit ratios based on overall fish density and the densities of economically important species such as Red Snapper, Greater Amberjack, Gray Triggerfish, Vermilion Snapper, and Almaco Jack. Species richness was found to increase with increasing tonnage of railroad ties and is likely due to differences in structural complexity. Evaluation of the two different types of cost-benefit ratios identified a massive 4000-ton railroad tie pile as the least cost-effective in terms of overall fish density and economically important species density. As a result, I do not recommend considerably large piles of reefing material if the goal is to provide habitat for economically important fish species, rather spread this material in smaller widely dispersed groupings. This study identifies the need for specifying clear objectives when designing artificial reefs to maximize efficiency. Future reefing

projects should also consider placing different materials further distances apart while still occupying similar geography, depth, and proximity to shore to properly examine their efficacy.

Introduction

Ecological and/or human benefits are the ultimate objective of habitat enhancement (Grove & Wilson 1994). Typically, a successful reef is one that is useful to people (Milon et al. 2000) with a high return on investment. Artificial reefs serve a variety of economic benefits from sport diving (Ditton & Baker 1999, Ditton et al. 2002) to recreational and commercial fishing (Seaman & Sprague 1991, Stone et al. 1991, Baine 2001, Adams et al. 2006). Human benefits of an artificial reef are directly related to its ecological benefits. More specifically, a healthy artificial reef is likely to contribute more to the local economy than an unhealthy reef. In the western Gulf of Mexico (GOM), artificial reefs have become a critical resource enhancement technique (Bohnsack & Sutherland 1985). They can also enhance the natural resources, particularly in areas where they are limited (Streich et al. 2017a). A seafloor that would otherwise be mostly barren now has vertical structures that serve as essential fish habitat (Bohnsack & Sutherland 1985, Shipp & Bortone 2009, Ajemian et al. 2015a,b), are important for fisheries management (Patterson et al. 2009), and relieve the restraint on population abundance with additional habitats (Bortone 2008) that can also facilitate range expansion (Rooker et al. 1997, Love & York 2005, Broughton 2012).

In response to the recognized importance of artificial reefs, the Artificial Reef Act of 1989 provided guidance for the development of cost-effective artificial reefs. One of the recommendations of this Act states to arrange materials in as complex a fashion to not jeopardize the structural integrity, but to maximize effectiveness, durability, and stability (Stephan et al. 1990). Due to differing limitations in fish productivity or recruitment by region, artificial reefs

should be designed to maximize abundance based on the needs of the region, yet it remains unclear what exactly an ideal reef design involves (Grossman et al. 1997, Strelchek et al. 2005, Blount et al. 2021). Furthermore, the materials used can also play a role in the overall effectiveness of the reef. Concrete materials, such as prefabricated pyramids, that have biomimetic properties, increase the benefits of reefs to the local environment (Vivier et al. 2021). An increase in artificial reefs in a region like the western GOM that lacks structured habitat for fishes should, in turn, increase survivability via refuge from predators and access to unexploited prey resources (Strelchek et al. 2005). However, region-specific environmental data and the specific benefits to the local community are often needed to evaluate reef effectiveness to maximize efficiency.

While the cost of the reef is relatively straightforward to obtain, benefits can be categorized as either human or ecological and have only recently been studied. Prior to 2000, relatively few studies documented the human benefits of habitat enhancements such as artificial reefs (Seaman Jr. & Jensen 2000). After 2000, numerous studies focused on evaluating the benefits of artificial reefs (Black 2001, Pitcher et al. 2002, Stanley & Scarborough 2004, Adams et al. 2006, Whitmarsh et al. 2008, Chen et al. 2013, Polak & Shashar 2013, Schuett et al. 2016, to name a few); however, many of the studies fail to address how the benefit(s) contributes to the local economy. Along with human benefits, evaluating the ecological benefits of an artificial reef is vital for determining the relative success of the reef. Quantifying the economic importance of an artificial reef may help justify future public artificial reef development projects and assist in developing adaptive strategies associated with reef deployment (Adams et al. 2006). Furthermore, when the ecological benefit is balanced with the cost associated with artificial reef implementation, return on investments in the project can be maximized and can inform future

reefing projects (Adams et al. 2006, Graham et al. 2016). With increasing legislative regulations on the types of materials permitted for artificial reefs, it is important to examine which materials provide the most benefit (Becker et al. 2018). In the United States, emphasis for artificial reef construction has been placed on ‘materials of opportunity’ (Stone et al. 1991, Pickering et al. 1999). However, few studies have examined the fish community response to different reefing materials (Brock & Norris 1989, Reed et al. 2006, Thanner et al. 2006, Ajemian et al. 2015b). At Rio Grande Valley (RGV) reef off the southern Texas coast, an assortment of artificial reefing materials have been placed to enhance the fish community in the area. These materials consist of prefabricated pyramids, culverts, railroad ties, oyster reefs, various concrete materials, watermills, and sunken vessels (Figure 4-1). Thus, this reef complex presents the ideal study site to examine multiple structures with the same latitude and environmental variables (e.g., visibility, temperature, etc.) that may not be similar when surveying among various locations (see chapter 3). Furthermore, this site has replicated materials and densities to examine the effects among different reef types while attempting to minimize pseudoreplication.

The purpose of this study was to examine fish communities at a few reefing materials and quantities at RGV reef to determine their efficacy. Specifically, this study aimed to: 1) test the effect of reefing material and quantity, henceforth structure type, on species richness, standard diversity indices, and fish community structure; 2) develop a cost-benefit ratio based on overall fish density; and 3) develop a cost-benefit ratio based on the densities of five economically important species in the GOM: Red Snapper (*Lutjanus campechanus*), Greater Amberjack (*Seriola dumerili*), Vermilion Snapper (*Rhomboplites aurorubens*), Gray Triggerfish (*Balistes capriscus*), and Almaco Jack (*Seriola rivoliana*).

Methods

Study Area

RGV reef is located 13.9 nautical miles from the port of Port Isabel. Reefing began in 2017 and has been updated with additional structures nearly every year since initial deployment. As previously mentioned, the RGV reef comprises of numerous types of artificial reefing materials and quantities (Figure 4-1). Thus, the location and extensive reefing in the area creates an ideal study site for sampling multiple reef types along the same isobath, isotherm, and visibility. All materials are located on the seafloor in about 20 m depth and are positioned within 0.5 km of each other (Figure 4-1). Sampling sites consisted of three nursery and three non-nursery pyramid locations and two 25-, 50-, and 100-ton railroad tie piles as well as a massive 4000-ton railroad tie pile (Figure 4-1). Both nursery and non-nursery pyramid sites consisted of four pyramids; however, “nursery” orientations harbored an additional 2-tons of loose concrete cinder blocks that provided low relief rubble habitat along with pyramids. (Figure 4-1).

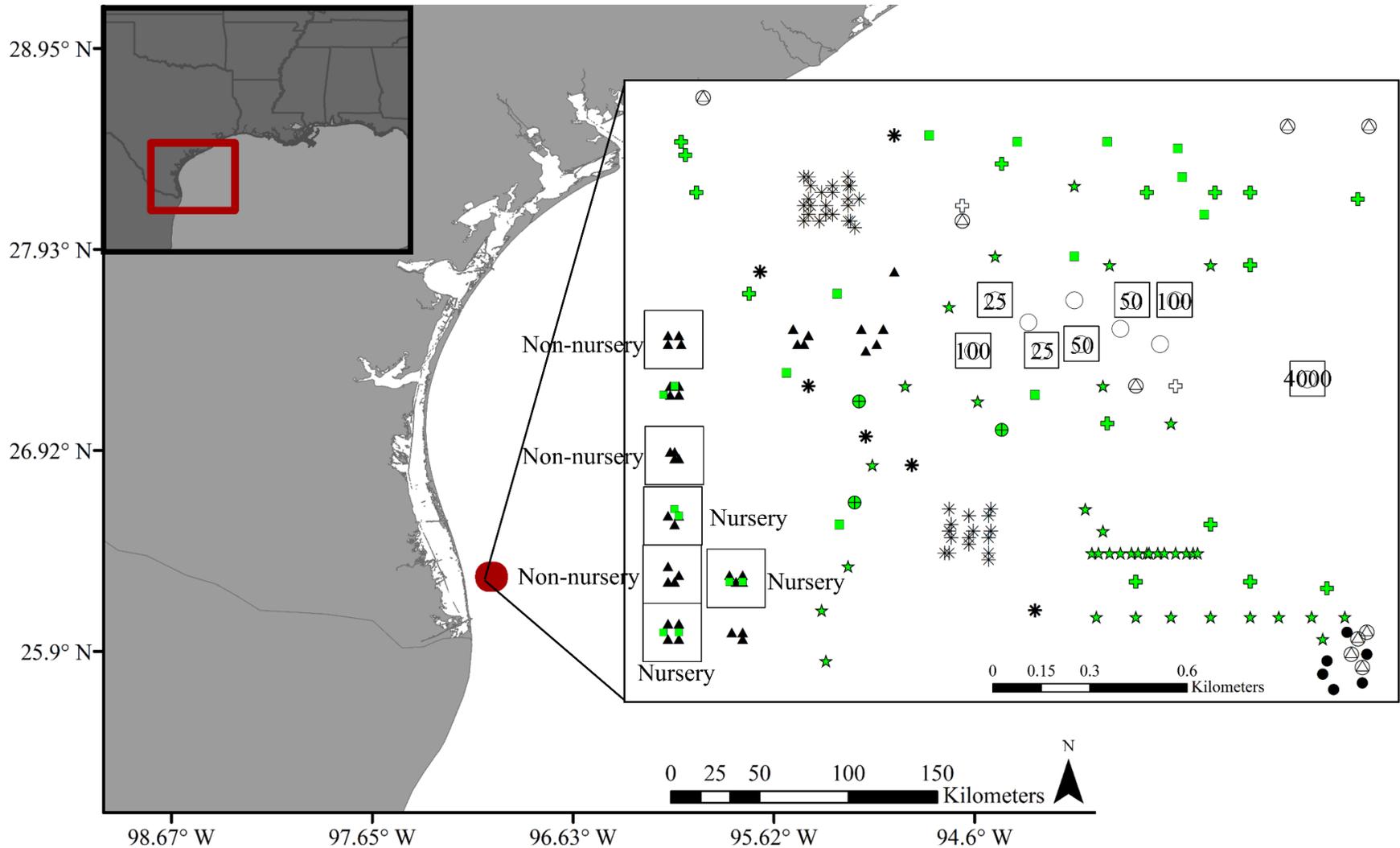


Figure 4-1. Map of Rio Grande Valley reef and various materials that have been reefed as indicated by differing symbols. Green symbols indicate different types of loose concrete material. Black squares are the hydroacoustic sampling areas for prefabricated pyramids (black triangles) and railroad tie piles (numbered squares). Numbers refer to tonnage of material for all sampled railroad tie piles.

Stereo-RUVS Sampling

Twenty-six stereo-remote underwater video systems (RUVS) surveys were conducted at four different tonnages of railroad tie piles and two types of prefabricated pyramid habitats (Table 4-1). Stereo-RUVS comprised of two GoPro® Hero 4+ cameras mounted 0.7 m apart on a base bar inwardly angled at 4° (Harvey & Shortis 1996, Watson et al. 2010, Letessier et al. 2015). The stereo cameras and the optimum measurement field-of-view were calibrated using VidSync software (Neuswanger et al. 2016) with a standard calibration frame in a similar environment to the sampled locations.

Table 4-1. Enumerates the number of stereo-RUVS surveys conducted at each type of site within Rio Grande Valley reef.

Material	Structure Type	Number of Sites	Stereo-RUVS Replicates	Stereo-RUVS Surveys
Pyramids	Nursery	3	2	6
	Non-nursery	3	2	6
Railroad Ties	25-ton	2	2	4
	50-ton	2	2	4
	100-ton	2	2	4
	4000-ton	1	2	2
Total		13		26

Stereo-RUVS were deployed for 60 min at each site and the first five min were removed from the analysis to prevent visibility and charismatic species bias (Watson et al. 2005). Due to battery-life limitations, only 45 min of the video was analyzed after the allotted five-min resting period. Pelagic and benthic stereo-RUVS were conducted simultaneously and arranged on the same vertical line anchored by 20 lbs of weight and marked with a surface buoy. The pelagic system was suspended 10-m below the surface and the benthic system typically rested on the seafloor.

Video Analysis

Video analyses were conducted in the laboratory with two independent readers recording species, count, time, and, if applicable, measurements. All species were time stamped and identified to the lowest possible taxon. Abundance estimates were made using MaxN (Priede et al. 1994, Ellis & DeMartini 1995), or the maximum number of individuals for each species seen in the frame at any given time. This technique produces a conservative estimate that minimizes the probability of double counting (Streich et al. 2017b). Due to anticipated low visibilities, MaxN was used in favor of MeanCount (Conn 2011) to increase the likelihood of capturing brief encounters with individuals.

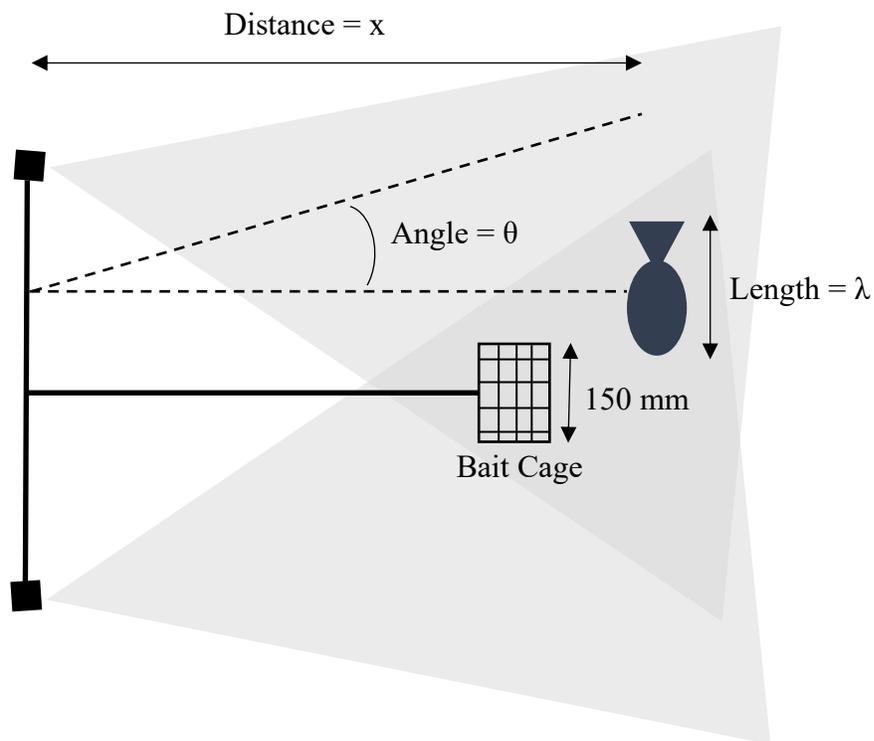


Figure 4-2. Stereo-RUVS configuration, field-of-view (shaded regions), and bait cage that was used to calibrate abundances.

If the two readers disagreed on a species identification, a third reader was included.

Counts were jointly reviewed if reader counts differed by more than 5%. The counts were then

averaged between the two readers for each species and then divided by the estimated visibility to remove potential visibility-bias on abundance. Fork length (FL) was measured to the closest mm for individuals that were visible in both cameras. Fish measurements were calibrated with a calibration frame (Neuswanger et al. 2016) and scaled to a 150-mm wide cage suspended in front of the cameras by a 1-m pole (Figure 4-2). The 1-m pole was also used as a reference to estimate visibility. All fish were measured and enumerated using VidSync (Neuswanger et al. 2016).

Species Composition Analysis

I investigated species richness along with diversity using traditional diversity measures such as Shannon's diversity index (H') and Pielou's evenness index (J'). I used a one-way analysis of variance (ANOVA; $\alpha = 0.05$) to test for potential differences in species richness, H' , and J' among structure types (Table 4-1; Ajemian et al. 2015a). I examined variation in fish community structure across and within site locations based on a Bray-Curtis similarity matrix of fish abundance using protocol outlined in Peters et al. (2015). The log of abundance (to limit the impact of overly abundant species) was analyzed with an analysis of similarity (ANOSIM) to determine if community structure differed between materials and among structure types. If the analysis resulted in differences in community structure for either test, a similarity percentages analysis (SIMPER) determined which taxa contributed to 70% of the differences observed (Peters et al. 2015). I then used a CLUSTER analysis to visually identify similarities in fish communities among structure types.

Finally, species that accounted for more than 5% of the individuals counted at each site were used to create a species similarity matrix using species counts and Whittaker's index of association (Whittaker 1952, Streich et al. 2017b). Hierarchical agglomerative clustering with a type 3 SIMPROF testing were used to evaluate species associations (Somerfield & Clarke 2013).

A shade plot was then created to qualitatively visualize selected species contributions across habitat types (Clarke et al. 2014, Streich et al. 2017b). All community analyses were run in Primer Version 7.0. All statistical analyses were conducted using the *onewaytests* (Dag et al. 2018) and *vegan* (Clarke 1993) packages in RStudio (R Core Team version 3.5.1; RStudio Team, 2018).

Hydroacoustic Sampling

To determine the abundance and distribution of economically important species, I used video data combined with hydroacoustic surveys using a Simrad EK80 split beam echosounder (70-18CD transducer; circular beam width of 18°, pulse duration = 0.256 μ s, specified ping rate = 300 ms). Thirteen hydroacoustic surveys were completed encompassing all individual sites within each structure type (Table 4-1). The EK80 was fixed to a polearm and positioned 1.9 m below the surface. Prior to each survey event, the EK80 echosounder was calibrated using standard methods with a 38.1 mm tungsten carbide sphere (Foote et al. 1987). Transects were conducted in a “flower” pattern centered on the geographic station position of each site (Reynolds et al. 2018; Figure 4-3). Spatial autocorrelation in hydroacoustic sampling methods was accounted for via ordinary kriging (detailed below). To allow for integration of the species composition with hydroacoustic data, further post-processing of the video dataset was required. Species unlikely to be detected acoustically due to size, morphology (e.g., damselfishes, sharks, etc.), cryptic behavior, or close association within the benthic fouling or structure were removed prior to further analysis (Simmonds and MacLennan 2005; Wilson et al. 2003, 2006).

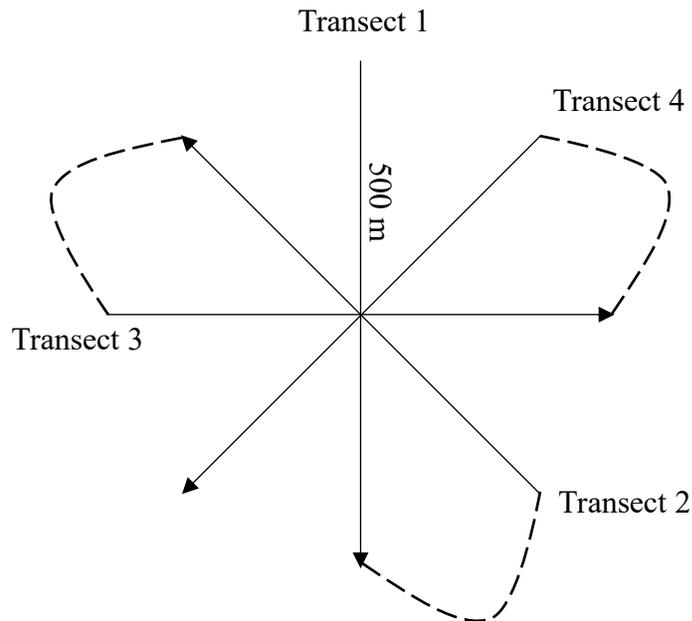


Figure 4-3. Example of a flower-pattern survey where the four, 250-m echosounder transects were centered over the geographic station position for each site.

Hydroacoustic Analysis

Raw hydroacoustic data were processed in the lab using Echoview 11.0 (SonarData Pty Ltd., Hobart, Australia). Prior to analyzing, data were edited to exclude unwanted noise and reverberation (Simmonds & MacLennan 2005, Boswell et al. 2007). Excluding noise reverberation included excluding 0.5 m from the sea floor and 1 m around artificial structures. Echo integration approaches were used when individual fish were too closely distributed for echo counting to be successful (Boswell et al. 2007). To account for spatial autocorrelation, ordinary kriging was applied to each layer using the following steps. First, a variogram, using initial values calculated from the data, was used to model the spatial continuity of the data collected at each site. Second, a 5 x 5-m survey grid was projected over each depth bin. Third, ordinary kriging interpolated spatially weighted estimates of area density over the survey grid from the predicted variogram. This resulted in depth bin-specific hydroacoustic density estimates

for 5 x 5 x 5-m cells arranged over the area bounded by the sampling squares placed over each site (Figure 4-1).

Fish abundance in each cell in each depth bin was calculated by multiplying the density by 25 m² (the area of the cell). Depth bin-specific abundance was then calculated by summing the abundances from each cell. Species-specific abundances at each site were calculated by taking the product of the abundance and percent species composition for each depth bin. Proportional species composition from the benthic stereo-RUVS surveys was applied to the depth bin encompassing the bottom 5 m, while proportional species composition from pelagic surveys was applied to 5 m from the bottom to the sea surface. Finally, these abundances were summed across all layers to generate a total species-specific abundance for each site.

Cost-Benefit Ratios

Two different cost-benefit ratios were calculated to represent the ecological and human (fisheries) benefits of the reef site. The first ratio was an overall cost-benefit ratio, which was calculated by taking the overall fish density (fish per m²) and dividing by the number of pyramids or tonnage of material for each site:

$$(1) \quad \text{Overall Cost: Benefit Ratio}_s = \text{Pyramid Cost} \div \left(\frac{TD}{nmat} \right)_s$$

Where *TD* is total fish density (fish/m²) and *nmat* is the number of pyramids for each sampled site *s*.

The second cost-benefit ratio incorporated three economically important species along the Texas coast: Red Snapper, Greater Amberjack, and Gray Triggerfish. Vermilion Snapper and Almaco Jack were not observed at any of the nearshore pyramid locations and were not included in the analysis. Each species was enumerated and weighted based on their respective commercial value (Table 4-2). I applied weightings due to the high importance of Red Snapper compared to

the other species being investigated (Eq. 2). Furthermore, because recreational value is uncertain and difficult to measure for all the species in question, I assumed that the commercial weighting was representative of the general economic weighting for each species (Eq. 2). As future economic valuation studies become available, they could be expanded based on these findings.

$$(2) \quad EIS \text{ Cost: Benefit Ratio}_s = Pyramid \text{ Cost} \div \sum_1^i \left(\left(FL_i * (FLtoW_i) * \left(\frac{D_i}{nmat_s} \right) * \left(\frac{Value_i}{Landings_i} * \frac{Value_i}{\sum_1^i Value_i} \right) \right) \right)_s$$

Where *FLtoW* corresponds to fork length (mm) to whole weight (lb) conversion, *D* is fish density (fish/m²), *nmat* is the number of pyramids or the tonnage of material, *i* represents each economically important species, and *s* indicates each site sampled. In essence, a cost-benefit ratio was assigned to each site based on the total weight of each economically important species.

Because measurements for Almaco Jack and Vermilion Snapper were not obtainable, I used an average weight based on commercial landings in the most recent stock assessment for both species (SEDAR 2016, 2020, respectively). The weight was then multiplied by the weighted value for each species and then summed across sites to produce a value (in US\$) per pyramid for each site (Table 4-2). I used the estimate presented by Shively (2014) of \$1,600 to reef one pyramid and divided by the density per pyramid. I tested for differences in cost-benefit ratios among reefing locations and pyramid orientations using a two-way ANOVA ($\alpha = 0.05$).

Table 4-2. Texas commercial value in US\$ of each of the five economically important species and weightings based on the total value. Values are based on 2020 Texas commercial landings (fisheries.noaa.gov).

Common Name	2020 TX Commercial Value (\$)	Weighting	Worth (\$/lb)	Weighted Worth (\$/lb)
Vermilion Snapper	276,239	2.21E-02	2.990	6.60E-02
Red Snapper	12,176,300	9.74E-01	4.420	4.30E+00
Greater Amberjack	45,695	3.65E-03	2.058	7.52E-03
Almaco Jack	5,981	4.78E-04	1.921	9.19E-04
Gray Triggerfish	527	4.21E-05	1.592	6.71E-05
Total	12,504,742			

Results

Over 38 hours of video from 52 stereo-RUVS surveys were processed. Six surveys yielded a result of no fish. Of the remaining 48 surveys, I counted 938 fish on benthic surveys and 620 fish on pelagic surveys, totaling 1,558 fish. Overall, I recorded 24 different species representing 14 families. Red Snapper was the most counted species ($n = 286$), followed closely by Gray Triggerfish ($n = 240$). Only 16 individuals were counted as unidentified. Gray Triggerfish, Red Snapper, and Blue Runner were seen at all structure types. The most speciose families were Carangidae ($n = 6$), Lutjanidae ($n = 4$), and Labridae ($n = 3$). Species richness varied significantly with structure type and appeared to increase with increasing tonnage of railroad ties (Table 4-3; Figure 4-4). Primary differences in species richness occurred between the nursery and 25-ton railroad tie pile and the other structure types (Figure 4-4).

Table 4-3. Results from the one-way ANOVA testing the effect of structure type on species richness, Shannon’s diversity index, and Pielou’s evenness index, where *df* refers to degrees of freedom.

Index	Parameter	<i>df</i>	<i>F</i>	<i>p</i>
Species Richness	Structure Type	5	5.794	0.02
	Residuals	7		
Shannon's Diversity Index	Structure Type	5	3.016	0.091
	Residuals	7		
Pielou's Evenness Index	Structure Type	5	1.109	0.434
	Residuals	7		

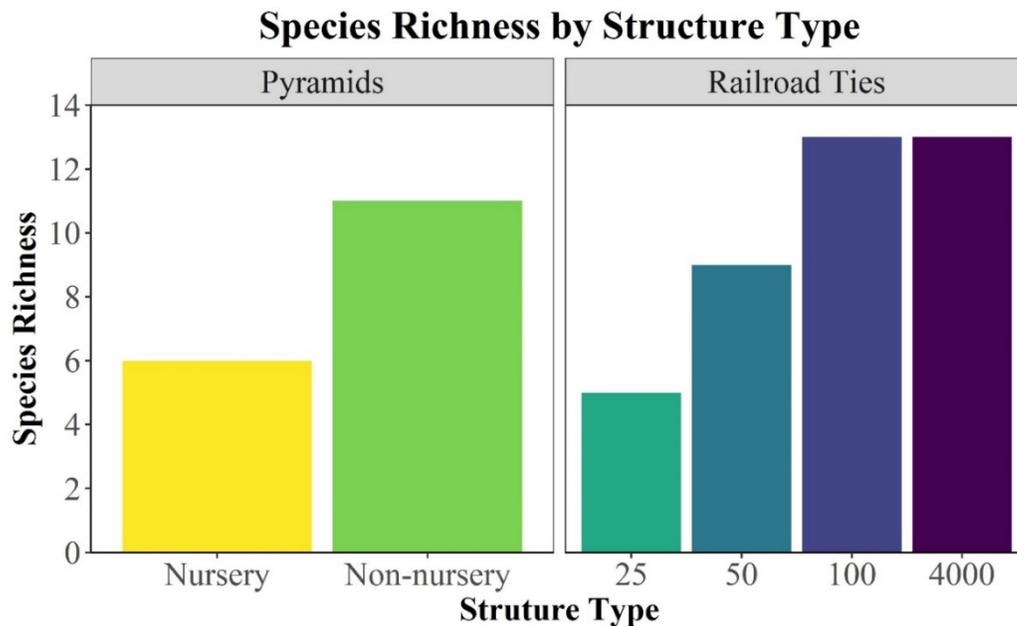


Figure 4-4. Overall species richness by structure type and categorized by reefing material. Numbers on the x-axis for railroad ties refer to tonnage of material.

Seventy-four individuals were measured across all surveys. The four most-commonly measured fish were Gray Triggerfish ($n = 25$; $\bar{x} = 258$ mm FL), Red Snapper ($n = 16$; $\bar{x} = 251$ mm FL), Blue Runner ($n = 12$; $\bar{x} = 170$ mm) and Atlantic Spadefish ($n = 10$; $\bar{x} = 212$ mm FL). No single species was measured at all structure types. Communities were remarkably similar across all structure types ($R = -0.101$, $p = 0.688$). In fact, no two structure types differed by more

than 33% (Figure 4-5). Some distinction occurred among the species that were observed at all structure types (Figure 4-5); however, these separations were not a result of structure type. The corresponding CLUSTER analysis also revealed indistinct clustering with structure type (Figure 4-6). Likewise, neither analysis revealed significant clustering by reefing material ($R = 0.095$, $p = 0.189$; Figure 4-5, 4-6).

After removing the species that contributed less than 5% of counts at a site, 22 species remained for further analysis. Clustering based on the remaining species and type 3 SIMPROF ($P < 0.001$) identified seven cooccurring species groups. Five of these groups contained a single species that was observed at only one site (groups A-E in Figure 4-7). Group F (Figure 4-7) contained Red Snapper as well as other EIS and were generally found at all structure types. Species in group G (Figure 4-7), except for Tomtate (*Haemulon aurolineatum*), were only observed at railroad tie pile sites.

Fish Community NMDS

Resemblance: S17 Bray-Curtis similarity

2D Stress: 0.15

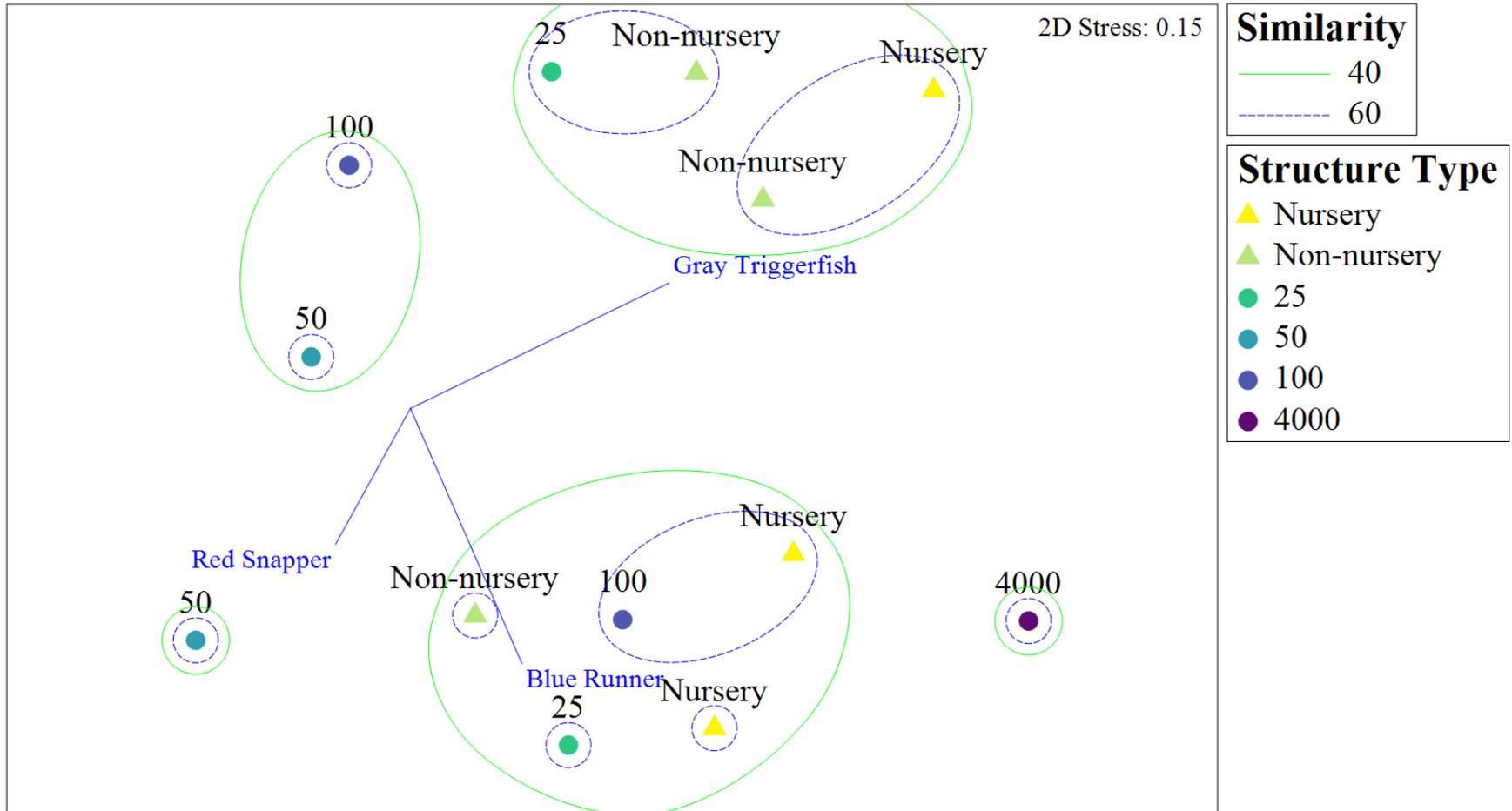


Figure 4-5. Fish community non-metric multidimensional scaling plot by structure type. Structure type is distinguished by color and structure material is distinguished by shape. Numbers refer to the tonnage of railroad tie material. Species vectors are based on Pearson correlation coefficients. Percent similarity bubbles (from CLUSTER) represent 40% (green) and 60% (blue) similarity.

Fish Community Dendrogram

Group average

Resemblance: S17 Bray-Curtis similarity

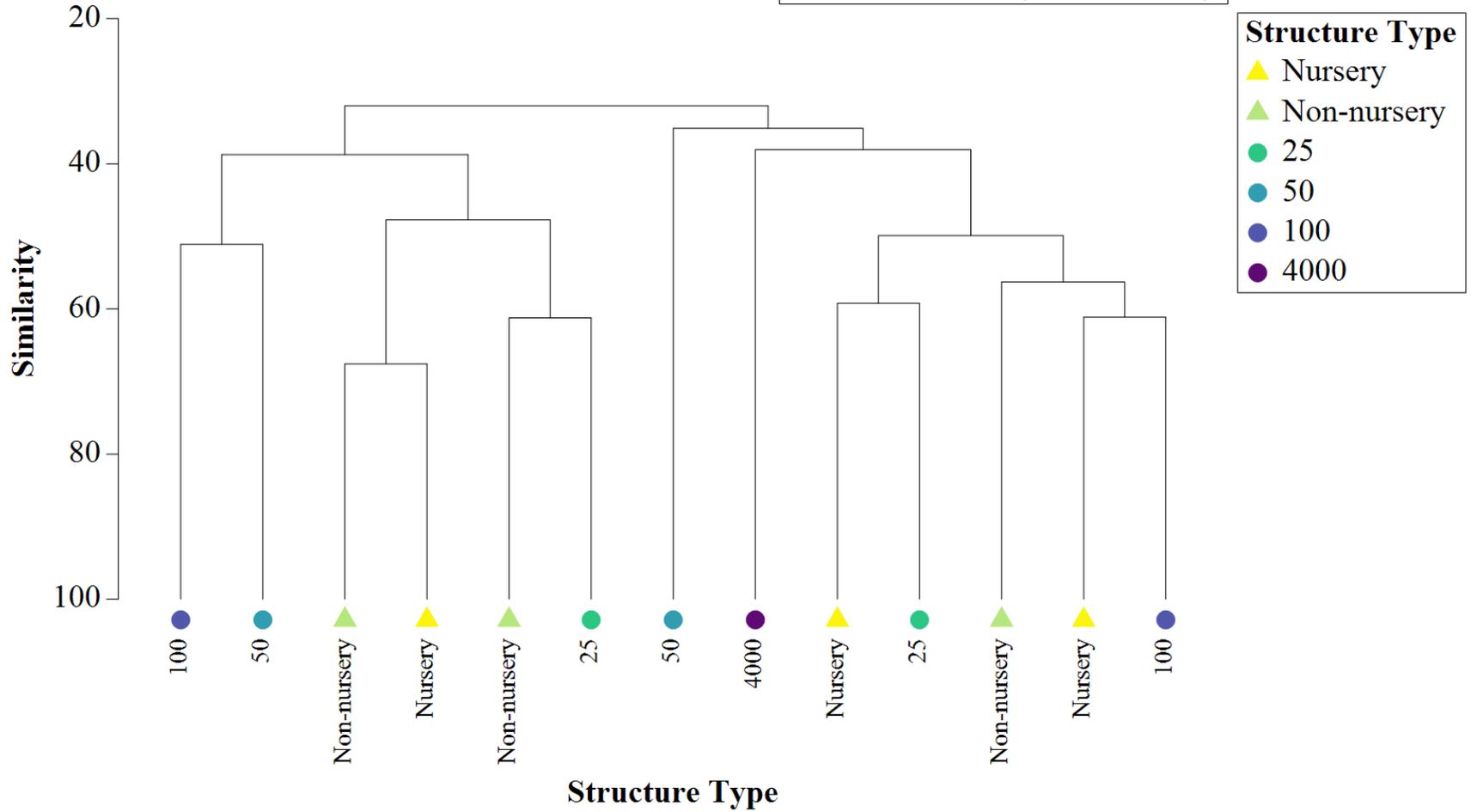


Figure 4-6. CLUSTER dendrogram representing groupings of communities. Structure type is distinguished by color and structure material is distinguished by shape. Numbers refer to the tonnage of railroad tie material. Linkages are based on Bray-Curtis similarity matrix of natural log abundances.

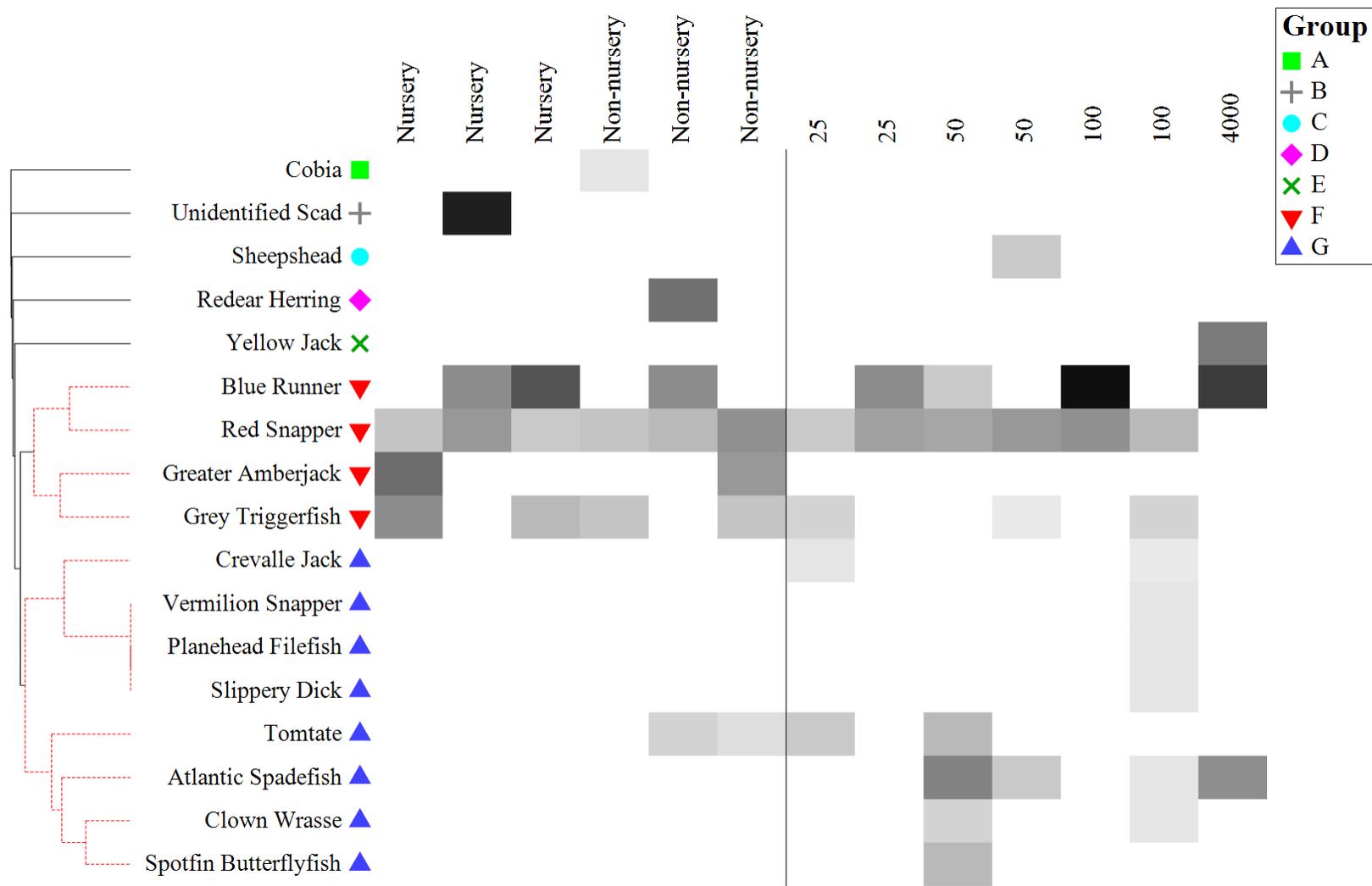


Figure 4-7. Shade plot representing the samples (columns) and species (rows) of the species contributing at least 5% of the counts for each site. Species dendrogram on the left illustrates the hierarchical clustering based on Whitaker's (1952) index of association resemblance matrix computed on natural log MaxN counts. Species groups were categorized using a Type 3 similarity profile testing ($P < 0.001$) as indicated by the connected red lines in the dendrogram and by the symbols displayed next to the species name. Samples to the left of the line are pyramid sites and samples to the right of the vertical line are railroad tie sites.

Hydroacoustic Analysis

I observed 8,059 fish across all structure types from the hydroacoustic analysis. Total abundance varied from 336 at the 25-ton structures to 4,396 at the 100-ton structures. Blue Runner were the most abundant species ($n = 3,969$), followed by Red Snapper ($n = 1,533$) and Gray Triggerfish ($n = 777$). Total fish abundance did not vary significantly with structure type ($F_{5,7} = 1.583, p = 0.280$). However, I saw higher average total fish abundance at the 100-ton railroad tie piles compared to the other structure types, but a large standard error (SE = 1645.9) indicates drastic differences in total fish abundance between the two 100-ton railroad tie piles sites (Figure 4-8).

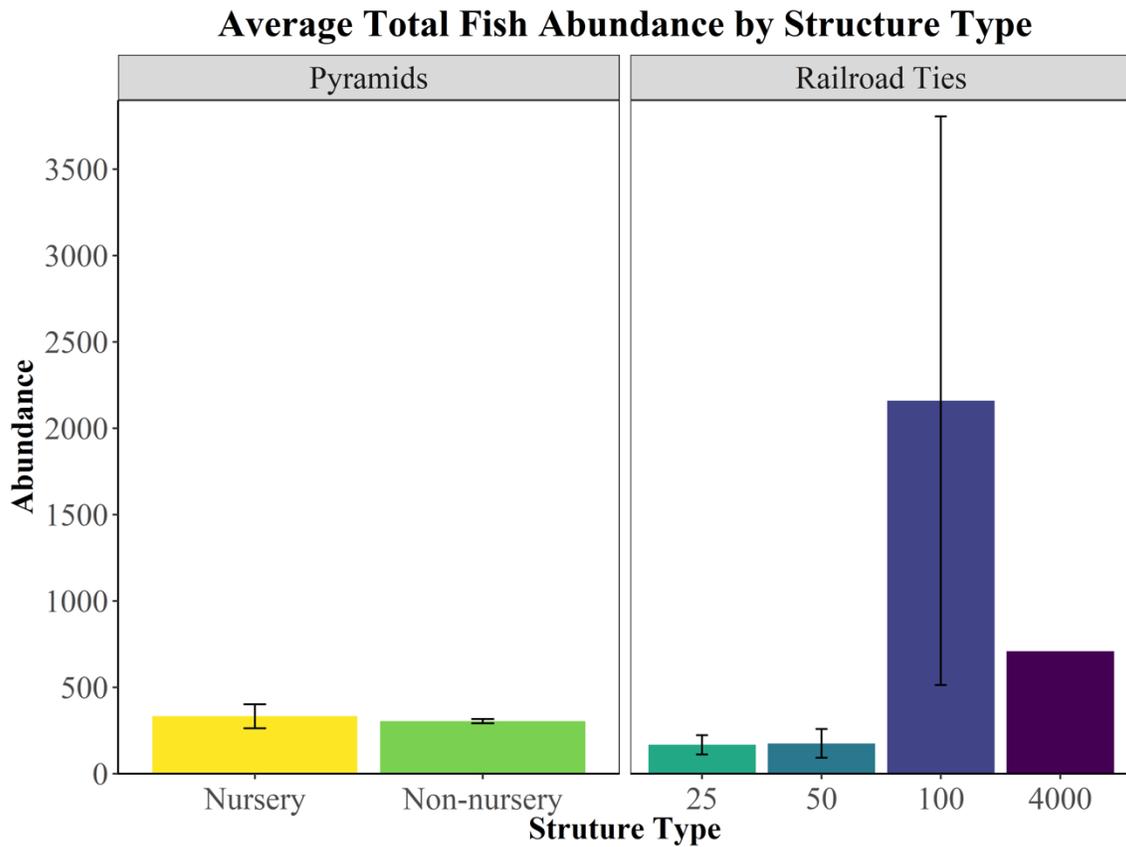


Figure 4-8. Average total abundance by structure type (mean±SE) for both pyramid (left) and railroad tie (right) reefing materials. Although the 100-ton railroad tie pile had the highest average abundance, this result was not significant.

The structure type with the lowest, or “best”, average overall cost-benefit ratio was the nursery pyramid structure type. The 25-ton railroad tie pile had the lowest EIS cost-benefit ratio. Both the overall ($F_{5,7} = 402.363, p < 0.001$) and economically important species ($F_{5,7} = 136.19, p < 0.001$) cost-benefit ratios were highly significant with structure type. For both cost-benefit ratios, the 4000-ton railroad tie site was drastically higher than the other structure types (Figure 4-9). As a result, I reanalyzed the data with the removal of the 4000-ton pile and found that neither the overall cost-benefit ratio ($F_{4,7} = 2.812, p = 0.110$), nor the economically important species cost-benefit ratio ($F_{4,7} = 1.166, p = 0.402$) remained significant (Figure 4-9). However, there remained significant variation in overall cost-benefit ratio between reefing materials ($F_{1,10} = 5.444, p = 0.042$).

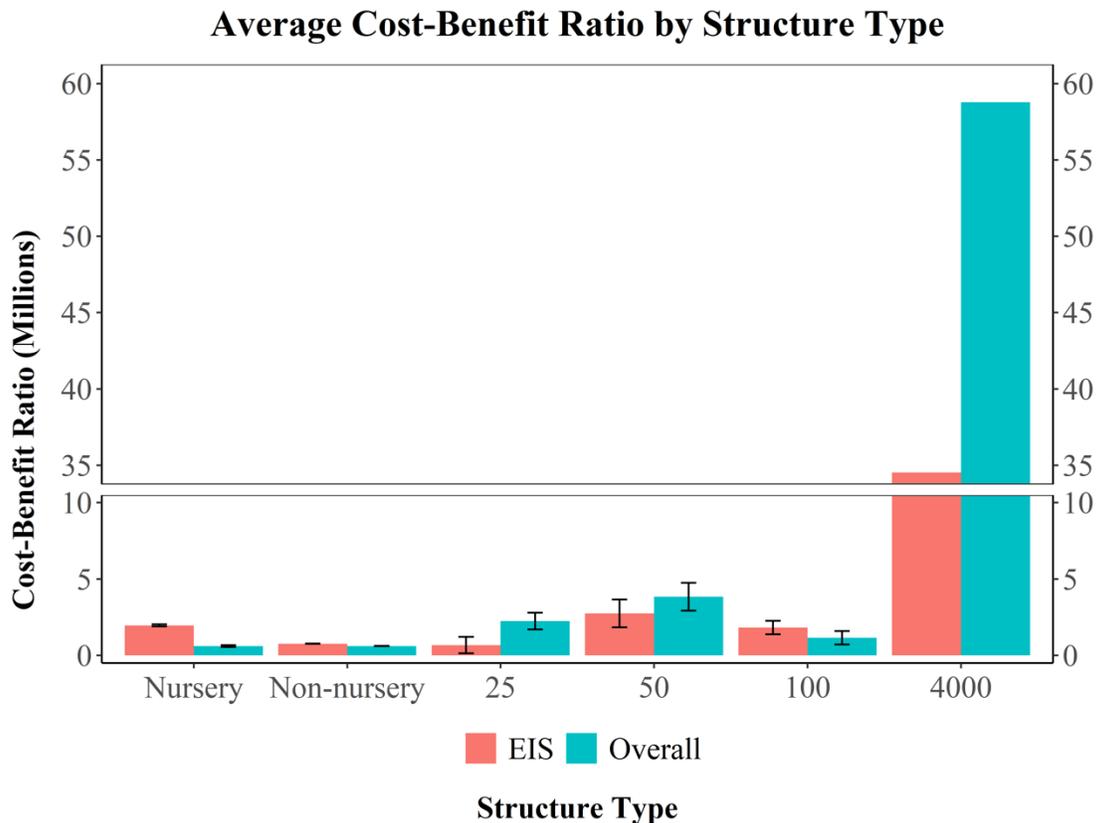


Figure 4-9. Bar plot of the average overall and economically important species cost-benefit ratios by structure type (mean±SE). *EIS* refers to the economically important species cost-benefit ratio and *Overall* refers to the overall cost-benefit ratio. Numbers on the x-axis refer to the tonnage of railroad tie material.

Discussion

This study observed reef fish communities inhabiting a nearshore artificial reef on various reefing materials and quantities on the south Texas coast. I observed differences in species richness among structure types, which are likely due to differences in structure complexity (e.g., rugosity). Fish community results indicated that there were differences in communities that were not due to the parameters explored in this study. This study also documented the use of these habitats by economically important species, especially Red Snapper. Although the “best” reefing material and density remains unclear, both cost-benefit ratios determined that the 4000-ton railroad tie pile was significantly less cost-effective than the other structure types. This study presents different ways to examine artificial reef efficacy and provides recommendations for future reefing projects.

Fish Communities

Concurrent pelagic and benthic surveys enumerated species with a variety of life histories and behaviors. Video-based approaches also captured economically important species and their potential prey species (Grimes 1979, Manooch & Haimovici 1983, Vose & Nelson 1994, Szedlmayer & Lee 2004). The presence of EIS at all sampled sites suggests that the materials I sampled provide suitable habitat for these species. Furthermore, this study showed most of the Red Snapper inhabiting RGV reef are below the minimum size limit for harvest in Texas state waters, indicating that RGV reef may be important habitat for the development of younger cohorts. Nearshore artificial reefs along the Texas coast have been previously described functioning as nursery habitat for juvenile Red Snapper (Beck et al. 2001, Streich et al. 2017b). The prefabricated pyramids used here make ideal nursery habitat in that they are embedded with limestone and are perforated with holes. Furthermore, the layering of railroad ties provides a

complex habitat for smaller fish to seek refuge. Although classifying nursery habitat was outside the scope of this study, RGV reef likely serves as important habitat for juvenile Red Snapper and Greater Amberjack. Four of the five Greater Amberjack measurements were under 206 mm FL, with only one larger individual observed (790 mm FL). Despite the widely accepted notion that jacks are highly mobile fishes (Burch 1979, McClellan & Cummings 1997, Ingram & Patterson 2001), recent evidence suggests that Greater Amberjack, and likely Almaco Jack, may experience a high degree of site fidelity (Baeyaert et al. 2018). Although data are generally lacking on localized movements of juvenile Greater Amberjack and on ontogenetic shifts in habitat, young-of-year carangids have been observed using floating sargassum patches as nurseries (Wells & Rooker 2004, Stachowiak 2020). In my study, young-of-year Crevalle Jack (*Caranx hippos*; \bar{x} = 38 mm FL) were regularly observed hovering around the pelagic cameras, which could suggest that their use of floating objects is not limited to sargassum patches and that they potentially exhibit positive attraction to suspended sampling gear. While my data suggest the use of RGV reef as a juvenile nursery for carangids, lutjanids, and potentially other species, more data are needed on the density of these juveniles at RGV reef compared to surrounding areas as well as the degree of residency.

Results from this study support both recruitment and productivity limited hypotheses. Species recruiting to the artificial reef in this study may be limited by the amount of available space, while overall abundance is limited by amount of recruitable stock in the region. Total abundance at RGV reef was not found to be significant with structure type and did not increase with increasing tonnage of material; however, species richness appeared to increase with increasing tonnage of material. These conflicting results suggest that fishes at RGV reef are likely both recruitment and productivity limited (Strelchek et al. 2005). As a result, fish

abundance in this region does not appear to be hindered by a lack of suitable habitat, but species richness is habitat limited (Strelchek et al. 2005). Increasing species richness with increasing structure size is not uncommon with artificial reefs (Walsh 1985, Ajemian et al. 2015a,b). Likewise, anecdotal observations of hydroacoustic data found an increase in structure height with increasing tonnage of material. Because railroad ties were haphazardly deployed over a single geographic point, materials stacked to build a highly rugose reef. Rugosity is a topographic measure of reef complexity and has been recognized as one of the most important predictors of fish species richness (Gratwicke & Speight 2005, Kuffner et al. 2007). Along with species richness, rugosity often correlates positively with abundance and biomass (Luckhurst & Luckhurst 1978, Caley & St. John 1996, Friedlander & Parish 1998, Chapman & Kramer 1999, Gratwicke & Speight 2005, Kuffner et al. 2007). Future studies should include measurements of reef complexity including rugosity, areal footprint, and surface area to identify the specific tonnage and type of material that has properties often associated with highly productive reefs (Chapman & Kramer 1999, Gratwicke & Speight 2005). Future studies should also incorporate side-scan sonar to determine the exact location of artificial structures. In this study, random site selection for stereo-RUVS deployments may have disproportionately favored sites with larger areal footprints. More specifically, randomly selected sampling locations had a higher likelihood of being directly on the structure at sites with a larger areal footprint than those with a smaller footprint. Sampling directly on the structure may lead to more representative fish communities at each structure type and could unearth more definitive community differences.

Indistinct clustering of fish communities may be due to the high connectivity potential in the RGV reef complex. RGV reef contains a multitude of reefing materials that are closely distributed (Figure 4-1). Because RGV reef complex has been inundated with artificial

structures, the close proximity of materials may have confounded my results and the results of future studies at this location. More specifically, many of the structures at RGV reef are spaced within a few hundred meters of each other. High connectivity and the diverse array of structure types likely explain why I observed higher abundances of habitat or diet generalists such as carangids, lutjanids, and labrids (Gallaway et al. 2009, Fitzpatrick et al. 2012, Kramer et al. 2015, van Lier et al. 2018). These groups of species typically respond positively with connectivity and seascape complexity (Staveley et al. 2017) and may be indicator species for high connectivity within a region or site.

Cost-Benefit and Future Recommendations

Collectively, RGV reef provides a diverse array of artificial reef habitat for reef fish communities. The structure type with the highest efficacy in terms of overall fish density were the nursery pyramids sites. These sites were built with the intention of providing refuge for smaller individuals or species (G. Glick pers. communication). For EIS, the 25-ton railroad tie piles had the highest efficacy; yet both the overall and economically important species cost-benefit ratios were not significantly different from the other structure types upon the removal of the 4000-ton railroad tie pile from analyses. Initial expectations were that such a large structure would show very positive responses; however, there was not a clear “best” structure type. In fact, the 4000-ton railroad tie pile was undeniably the least effective in terms of cost-benefit. The cost of placing 4000 tons of railroad ties greatly outweighed the ecological and economic benefit when compared to the other structure types. These results are corroborated by a previous study (Chapter 3) that identified large, highly dense pyramid sites as less cost-effective than smaller, more widely distributed sites. Nonetheless, in this study, cost-benefit was determined by overall fish density and EIS density. Large piles of structure such as the 4000-ton railroad tie pile in this

study may be more effective as a sport-diving site, rather than a site intended for fish attraction or production. These results highlight the importance of clear goals when constructing artificial reefs to determine their success using methods specific to that goal.

In general, the overall success of RGV reef is largely determined by its goal. According to *Friends of RGV Reef*, the organization responsible for the construction of the reef, the primary objectives of the reef were to restore lost nursery reef and provide stepping-stones to carry fish species through multiple life stages. Although this study was not designed to classify nursery habitat, the large presence of juvenile lutjanids and carangids suggests that the first objective of the reef is being met. However, the lack of mature individuals of these species may indicate that the reef is not carrying all life stages. Several factors including the time-of-year of sampling, fishing pressure, or bottom depth could have influenced the spectrum of life stages that were observed in this study. For example, sampling was conducted during the months that are typically most popular for recreational anglers on the Texas coast (Ditton et al. 1978). Rio Grande Valley reef is also a widely publicized fishing spot (tpwd.texas.gov, rgvreef.org) and is within state waters, meaning that recreational harvest of Red Snapper and other species over a certain size is open year-round. Persistent fishing of harvestable size-classes could be limiting the presence of adult life stages. The presence of younger life stages could also be due to the shallow depth of the reef. Red Snapper and other reef fishes have demonstrated ontogenetic movements to deeper depths and to different suitable habitats with depth (Eggleston 1995, Hujibers et al. 2015, Dance & Rooker 2019). More specifically, Red Snapper exhibit shifts from inshore (20 m depth) to offshore (100 m depth) across the Texas continental shelf with age (Dance & Rooker 2019). Although nearshore reefs are more accessible to fishers, if future reefs of this nature intend on harboring a higher abundance of Red Snapper, a reefing location further

offshore may be more appropriate. Several studies have reported higher abundances of age 2+ Red Snapper in 41-100 m depths (Gallaway et al. 2009, Karnauskas et al. 2017, Stunz et al. 2021), which may be more suitable for artificial reefs on the Texas continental shelf if the success of the reef is being defined by commercial and recreational fishing.

This study provides an in-depth analysis of different reefing materials and quantities as artificial habitat on the Texas coast, but more data are needed to determine the “best” reefing materials. In general, species richness increased with increased railroad tie pile tonnage; however, the 4000-ton railroad tie pile was the least cost-effective and, as a result, I do not recommend exceedingly large piles of materials for fisheries-based reefing designs. However, results may have been confounded due to the close proximity of other structures at this location, which exemplifies the importance of developing artificial reefs with science in mind. Science-based reefing designs (e.g., Port O’Connor nearshore reef) allow for unbiased sampling designs to determine the ecological benefits of the reef more accurately. Proper evaluation of reef success could lead to more cost-effective reefs in the future and further the understanding of the role of artificial reefs in the life histories of economically important fishes. Future reefing projects should consider reef construction with science in mind, which may include placing different materials further distances apart while still occupying similar geography, depth, and proximity to shore to properly examine their efficacy.

This study provides a case study of how to ensure best reefing practices. For fisheries-based reefing projects, managers may want to: 1) define clear goals for the artificial reef, 2) examine the life histories and ecology of the primary target species to determine ideal reefing locations, 3) evaluate the proximity of the reef to other nearby natural or artificial reefs, and 4) design the reef configuration that minimizes pseudoreplication and other confounding factors. As

artificial reefing practices improve, so too will the science examining their effectiveness and the scientific understanding of how artificial reefs affect fish communities.

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CHAPTER V:
INDIVIDUAL-BASED MODEL OF RED SNAPPER AND GREATER AMBERJACK
POPULATION RESPONSE TO DIFFERING DENSITIES OF TWO TYPES OF ARTIFICIAL
REEFS

Abstract

Artificial reefs are one of the most beneficial resource enhancement tools, but there is a lack of understanding of how artificial reefs affect fish populations and community dynamics. With advances in computing power over the last few decades, modeling has become one of the most important tools used to confront the different biological and technical interactions among fish species in a system. The purpose of this study was to build on the individual-based modeling techniques to assess these dynamics around manmade reefs. Specifically, I sought to evaluate Red Snapper, Greater Amberjack, and four competing species, population responses to changing artificial reefs in more complex, but realistic, habitat scenarios like those observed on the Texas continental shelf. The model simulated growth, mortality, and movement over 50 years on a 2-dimensional spatial grid composed of natural, prefabricated pyramid, and rig reefs. Simulation experiments consisted of various combinations of no pyramids, no rigs, or differing densities of rigs or pyramids with a natural reef always present. Model results indicated an overall increase in Red Snapper and Greater Amberjack biomass and average weight-at-age with increasing number of structures. Additionally, distance traveled, forage distance, and number of hours at risk, which are indicators for predation risk, decreased with increasing number of structures and mirrored patterns in connectivity on the spatial grid. These results suggest that habitat connectivity plays an important role in population stability, mixing, and for other important life stages in life history for species that use artificial and natural habitats. This model also validated age- or size-based

transitions in habitat for Red Snapper that have been observed in other studies, which suggests a transition from habitat-centric to forage-centric movement for this species. Furthermore, I incorporated value and cost-benefit into the model to assess artificial reef efficacy over differing reef configurations. This model provides resource managers and stakeholders with a tool to guide future reefing projects to discern the impact of placement and density of artificial reefs on population dynamics of economically important fish species.

Introduction

Artificial reefs are perhaps one of the most lucrative resource enhancement tools (Bohnsack & Sutherland 1985), yet proper placement, and proximity to other reefs is highly region specific, difficult to ascertain (Charbonnel et al. 2002, Strelchek et al. 2005, Vivier et al. 2021), and, scientifically, poorly understood. Primarily, this is due to a lack of understanding of how artificial reefs affect fish populations and community dynamics (Campbell et al. 2011), which can be quite complex (Carpenter 1988). A fish will grow several orders of magnitude, experience intra- and interspecific competition, reproduce, and occupy many different trophic niches in its lifetime (McDermot & Rose 2000). Multiple forms of mortality including natural, predation, or fishing mortality also increase the complexity of the community interactions. Meanwhile, the same fish will be interacting with these reef environments and responding differently to different environmental stressors.

With advances in computing power over the last few decades, simulation modeling has become one of the most important tools enabling elucidation of these different biological and technical interactions among fish species in a system (Rose et al. 1996). Yet, fisheries stock assessment models continue to focus on single-species models (Quinn 2003, Quinn & Collie 2005, Marasco 2007, Kinzey & Punt 2009), which can be attributed to guidelines for the

implementation of the Magnuson-Stevens Act that focused fisheries stock assessments on the exclusive use of population models (Rose & Cowan 2003). Recent advances in ecosystem-based fisheries management (EBFM) plans have since shifted the focus from species-specific approaches to ecosystem-based approaches (Moore et al. 2016, O'Farrell et al. 2017). Ecosystem-based approaches can lead to more informed predictions on how the structure and dynamics of fish production will respond to changes in multiple stressors including, but not limited to, climate change and fishing pressure (Collie et al. 2016, Doyen et al. 2017). New holistic or ecosystem models for EBFM rely on descriptions of species interactions to account for the indirect effects of harvesting one species on another species and emphasize the importance of understanding energy flow through systems from primary production to tertiary components of food webs (Jurado-Molina et al. 2005, Collie et al. 2016). Other fisheries models include budget models, coupled single-species models, and individual-based bioenergetics models (Rose & Sable 2009). Budget models typically explore biomass or energy flows of a food web, while coupled models form community models from several single-species models (Rose & Sable 2009). Individual-based models (IBMs), on the other hand, simulate populations and communities by following individuals and their properties through time as they are exposed to environmental gradients and other critical components to systems (DeAngelis & Grimm 2014). Applying this modeling technique improves the scientific understanding of both positive and negative anthropogenic impacts on fish communities.

Unlike traditional empirical models, IBMs are bottom-up models in which population-level behaviors emerge from the interactions of autonomous individuals and their environment (DeAngelis & Grimm 2014). Aspects usually ignored in an analytical model such as variability among individuals, local interactions, complete life cycles, and individual behavior adapting to

their changing internal and external environments can be incorporated into an IBM (Grimm et al. 2006). IBMs also allow researchers to study how system-level properties affect the behavior of individuals from multiple species (Campbell et al. 2011, Grüss et al. 2016, Campbell et al. 2018). Thus, IBMs are thought to better simulate an individual's natural responses to changing scenarios than analytical models, which is particularly useful for fisheries managers and especially those undertaking EBFM (Boyd et al. 2018).

Recent examples of IBMs include PISCATOR and OSMOSE (object-oriented simulator of marine ecosystems exploitation), which were developed to simulate fish community dynamics and trophic interactions (van Nes et al. 1996, Shin & Cury 2001, 2004). Other IBMs have been developed to simulate fish community responses to various environmental or anthropogenic stressors (Rose et al. 1996, McDermot & Rose 2001, Rose & Sable 2009, Campbell et al. 2011). One such study developed an IBM to address fish community responses to various artificial reef configurations (Campbell et al. 2011). This novel study integrated movement, bioenergetics, reproduction, and mortality of Red Snapper (*Lutjanus campechanus*), and two competitor species, to evaluate their response to changes in oil and gas platform (henceforth, rig) design (Campbell et al. 2011). If appropriately adapted and updated to current understanding for the modeled species, this model could also lead to further understanding of the influence of different artificial reef designs (i.e., different materials) on economically important fish species populations dynamics, thus providing resources managers and stakeholders with a tool to improve reefing practices.

The purpose of this study was to build on the model described in Campbell et al. (2011). I sought to evaluate Red Snapper, Greater Amberjack (*Seriola dumerili*), and competing species, population responses to changing artificial reefs in more complex, but realistic, habitat scenarios

like those observed on the Texas continental shelf. More specifically, this study answered questions regarding artificial reef efficacy with regards to Red Snapper and Greater Amberjack using commercial valuation and a cost-benefit ratio. This study explored rig and prefabricated pyramid density and placement to inform future reefing projects.

Methods

Model Description

The model evaluated the artificial reef efficacy by simulating the population dynamics of four focal fish species over 50 years on a 2-dimensional spatial grid. My four focal species were Red Snapper, Pinfish (*Lagodon rhomboides*), a croaker-like species, and Greater Amberjack. I also included Bluefish (*Pomatomus saltatrix*) and a jack-like species to act as predators and compete for prey and resources with the other species. Similar to Campbell et al. (2011), the model year was from July 1 to June 31 and movement, growth, and mortality were computed each hour for Red Snapper, Pinfish, croaker-like species, and Greater Amberjack. Bluefish and jack-like individuals were only simulated for hourly movement and only Bluefish acted to remove prey in their spatial cell based on their consumptive rates (Figure 5-1; Campbell et al. 2011). The mortality rates of Red Snapper, Pinfish, and croaker were adjusted based on the biomass of Bluefish, jack, and Greater Amberjack to mimic spatially dynamic predation (Campbell et al. 2011).

The model uses a superindividual approach to represent the model individuals (Scheffer et al. 1995, McDermot & Rose 2000, Campbell et al. 2011). This approach solves the need for many model individuals to account for the variability in randomly drawn population subsets, while solving for the common IBM issue of dynamic allocation of array space as fish experience birth and mortality through time (Scheffer et al. 1995). The number of super-individuals included

in the model was fixed and represented relative predator-prey ratios. I assigned a worth value to represent the survivability of each cohort. Worth was assigned when each cohort entered the model as newly recruited age-1 individuals. Initial worth was calculated by dividing the population number of age-1 individuals by the total number of model individuals. Each hour, worth was multiplied by the fraction surviving for the four focal species.

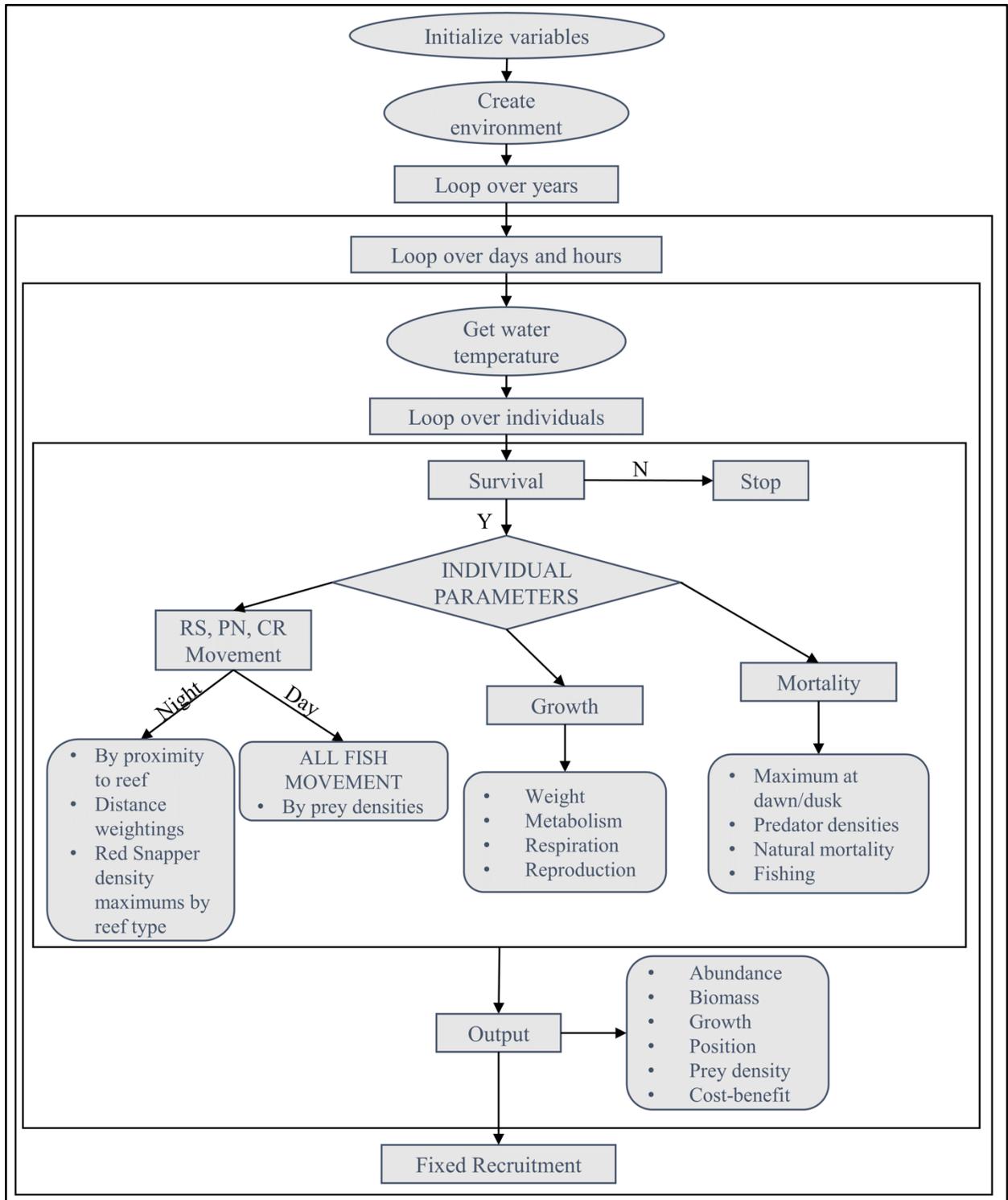


Figure 5-1. Conceptual diagram of the model design, components, and outputs. *RS*, *PN*, and *CR*, refer to Red Snapper, Pinfish, and croaker, respectively.

Model Environment

The model domain was 100 km² comprised of 10,000 cells, each measuring 100 m side⁻¹. A smaller spatial grid than Campbell et al. (2011) was selected to more accurately represent the area of influence of a reef composed of prefabricated pyramids. The number of natural bank cells represented the area of a typical natural bank on the Texas continental shelf. The number of daylight hours in a day varied throughout year to represent normal cyclical changes in daylight. Water temperature was assumed to vary daily and spatially across the grid, where the left (or west) side of the grid represented shallow depths and the right (or east) side of the grid represented deeper depths. Daily water temperature was calculated hourly using the following equations generated from data presented in Turner et al. (2017) to get a column-, or depth-specific temperature:

$$(1) \quad b = 37.098 + (dailytemp_d - (-3.801 \cdot (\ln(0.13 \cdot \frac{ncol}{2} + 20))) + 37.098)$$

$$(2) \quad temp_{d,i} = -3.801 \cdot \ln(0.13 \cdot i + 20) + b$$

Where d is the model day, $ncol$ is the total number of columns, $dailytemp_d$ is the computed overall daily temperature, i is the column number, and $temp_{d,i}$ represents the temperature of the i^{th} column on d day. In essence, the computed daily temperature was used to find the temperature of the center of the grid (Eq. 1). This temperature was then used as the center point for which the temperatures for the other columns fluctuated around (Eq. 2). Designed to represent the contours of the Texas continental shelf, depth increased 0.13 m every grid cell, or every 100 m, and the 0th column represented 20 m bottom depth. Depth was assumed to be the same within a single column from North to South along the grid (Figure 5-2).

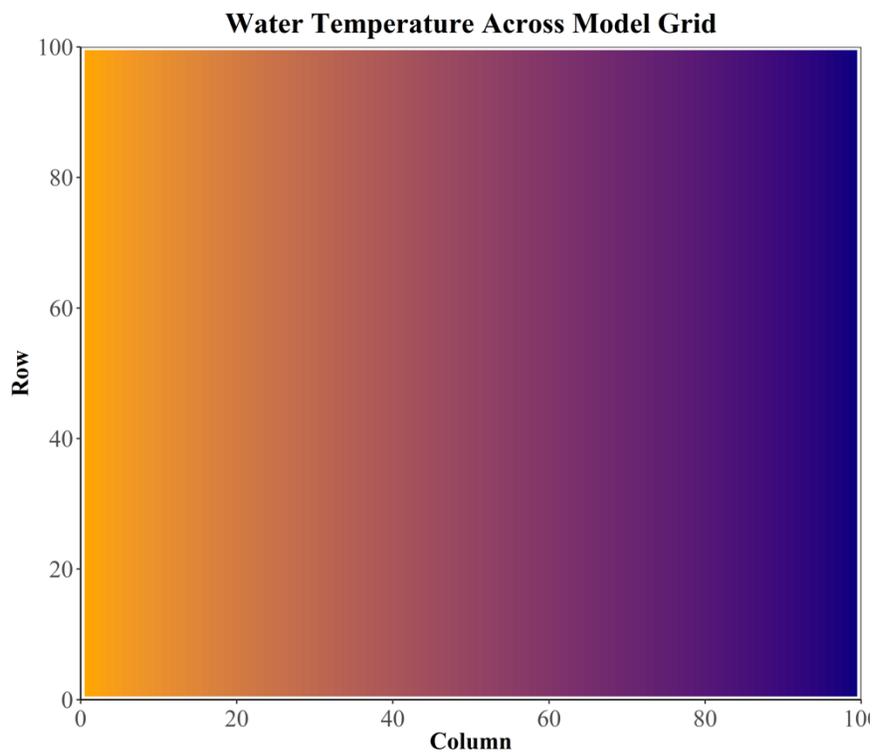


Figure 5-2. Illustrates how water temperature varied from column 1 to column 100 across the spatial grid. Temperature ranges differed depending on the model day (Eqs. 1 & 2). Orange coloration indicates warmer temperature and blue coloration indicates cooler temperatures.

Habitats

Cells were assigned one of four habitat types: benthic (non-reef), rig, pyramid, or natural.

All reef habitats were assumed to provide maximum refuge for Red Snapper, Pinfish, and croaker (i.e., only natural mortality). However, natural reef habitats did not provide refuge for those same species less than 5 years of age (Gallaway et al. 2009). Five prey groups were simulated on every cell and the density was updated each hour on each cell using a logistic growth equation (Campbell et al. 2011). These prey groups were zooplankton, crabs, shrimp, pelagic fish, and benthic fish (Table 5-1).

Table 5-1. Different parameters injected into the model for each of the five prey types. E_{prey} refers to the energy density.

Prey Type	Biomass (g ww/m ²)	Production Rate (1/yr/hr)	E_{prey} (J/g ww)
Zooplankton	6.7	0.00197	3511
Crabs	4.0	0.00046	3138
Shrimp	6.0	0.00046	3894
Pelagic Fish	20.0	0.00046	4947
Benthic Fish	20.0	0.00046	4947

Growth

As described in Campbell et al. (2011), growth for the four focal species was calculated hourly in weight (W , g ww). A modified Wisconsin bioenergetics model computed growth for each focal species (Eq. 3). Bioenergetics models describe an individuals' growth as a balance between consumption and energy losses such as respiration, specific dynamic action, excretion, egestion, and reproduction (Hatzonikolakis et al. 2021):

$$(3) \quad W_{t+1} = W_t + \left[(C - R - Ex - F) \cdot \frac{E_{prey}}{E_{fish}} - S \right] \cdot W_t$$

where C is hourly consumption rate, R is the respiration rate (standard metabolic rate + costs for activity), Ex is excretion, and F is egestion. E_{prey} and E_{fish} are the energy densities (J g ww⁻¹) of the prey and predator, and S is weight loss due to spawning. C , R , Ex , and F and were computed in units of g ww prey * g ww fish⁻¹ * h⁻¹ and were converted to g ww fish * g ww fish⁻¹ * h⁻¹ by multiplying by the ratio of the prey and predator (fish) energy densities. S was computed as g ww fish * g ww fish⁻¹ * h⁻¹. I was unaware of a bioenergetics model for Greater Amberjack, and thus, one was created using parameters from a similar species, Yellowtail Kingfish (*Seriola lalandi*; Brodie et al. 2016, Donohue et al. 2021).

Table 5-2. Parameter values for the bioenergetics, movement, natural mortality, and number of model individuals for each of the six modeled species.

	Parameters	Red Snapper	Pinfish	Croaker	Bluefish	Jack sp.	Greater Amberjack
Maximum Consumption	a_G	0.182	0.180	0.140	0.5197	-	0.240
	b_G	-0.274	-0.274	-0.250	-0.288	-	-0.320
	T-max	35	40	38	38	-	34
	T-opt	27.4 - 26.4	27.5	28	28	-	26
	θ	2.3	1.8	2.0	3.0	-	3
Respiration	a_R	0.0045	0.0200	0.0130	0.0540	-	0.0067
	b_R	-0.200	-0.202	-0.202	-0.200	-	-0.210
	T-max	35	40	38	38	-	34
	T-opt	32	27.5	28	28	-	30
	θ	2.1	1.8	2.0	3.0	-	2.0
	Act	2.5	1.5	1.25	1.25	-	2.5
Egestion	a_F	0.212	-	-	-	-	0.212
	b_F	-0.222	-	-	-	-	-0.222
	g_F	0.631	-	-	-	-	0.631
	p_F	-	0.20	0.20	0.15	-	-
Excretion	a_U	0.031	-	-	-	-	0.031
	b_U	0.58	-	-	-	-	0.58
	f_U	-0.299	-	-	-	-	-0.299
	p_U	-	0.10	0.07	0.10	-	-
Movement	Distance (m/h)	100	90	120	150	150	150
Mortality	Natural (1/yr)	0.226	0.361	0.323	-	-	0.280
Super-individuals	-	2000	5000	1000	800	800	600

Table 5-3. Species-specific vulnerabilities (v), half-saturation ($ksat$) values, and energy densities (E_{fish} ; J/g ww) for each of the five species that consumed prey.

Parameter	Prey Type	Red Snapper	Pinfish	Croaker	Bluefish	Greater Amberjack
v	Zooplankton	0.2	0.9	0	0	0
	Crab	0.1	0.9	0.6	0	0
	Shrimp	0.1	0.9	0.4	0	0.03
	Pelagic Fish	0.05	0	0.06	0.2	0.2
	Benthic Fish	0.1	0	0.15	0.2	0.25
$ksat$	Zooplankton	160	150	0	0	0
	Crab	160	156	145	0	0
	Shrimp	150	156	145	0	25
	Pelagic Fish	115	0	145	100	130
	Benthic Fish	115	0	145	100	135
E_{fish}	-	4999	5200	5050	7498	5500

I calculated maximum daily consumption rate (C_{max}) and respiration (R) using species-specific bioenergetics parameters (Table 5-2), cell-specific water temperatures, and individual weight (g ww). Both C_{max} and R are computed as allometric functions of weight:

$$(4) \quad C_{max} = a_C \cdot W^{b_C} \cdot f_C(T)$$

$$(5) \quad R = a_R \cdot W^{b_R} \cdot f_R(T) \cdot act \cdot occ$$

where a_C , b_C , a_R , and b_R are the intercepts and exponents for consumption and respiration, act is the activity multiplier and occ is an oxy-caloric coefficient:

$$(6) \quad occ = \frac{13569 \text{ joules}}{\text{g oxygen}} \cdot \frac{1 \text{ g fish}}{\text{joules/g ww fish}}$$

The effect of temperature was modelled through a species-specific temperature dependent function (determined by Φ) that is maximized at a species optimal temperature (T_{opt}) and minimized at their maximum temperature (T_{max}):

$$(7) \quad V = \frac{T_{max} - T}{T_{max} - T_{opt}}$$

$$(8) \quad Z = \ln(\Phi) \cdot (T_{max} - T_{opt})$$

$$(9) \quad Y = \ln(\Phi) \cdot (T_{max} - T_{opt} + 2)$$

$$(10) \quad X = \frac{Z^2 \cdot \left(1 + \sqrt{\left(1 + \frac{40}{Y}\right)}\right)^2}{400}$$

$$(11) \quad f(T) = V^X e^{X(1-V)}$$

where T is the current cell-specific temperature and Φ is the species-specific slope for temperature dependence. Hourly respiration was increased by 30% every hour that a Red Snapper, Pinfish, or croaker individual was not on a reef cell during nighttime hours.

Hourly individual consumption of each of the five prey groups was calculated using a type 2 functional response from the individuals' C_{max} , cell-specific prey densities, and prey

vulnerabilities (v) and half-saturation values ($ksat$) specific to each species and age class (Table 5-3; Campbell et al. 2011):

$$(12) \quad C_j = \frac{C_{max} \cdot \frac{P_j \cdot v_j}{ksat_j}}{1 + \sum_{k=1}^5 \frac{P_k \cdot v_k}{ksat_k}}$$

where P is the cell density of prey type j , and v is the vulnerability and $ksat$ is the half saturation parameter of prey type j and k to that individual. C_j values were then summed across all prey types to represent the total hourly consumption. Consumption ended if the total hourly consumption exceeded maximum hourly consumption (C_{max}) or nighttime hours began.

Vulnerabilities (v) for each prey type were determined based on literature diet information for each species and half-saturation ($ksat$) values were then calibrated to reflect appropriate weights-at-age for each species (Table 5-3; Manooch & Haimovici 1983, Hartman & Brandt 1995, Pipitone & Andaloro 1995, Thompson 2000, Patterson et al. 2001, Wilson & Nieland 2001, Nelson 2002, Murphy 2006, Shipley 2008, Gallaway et al. 2009, Campbell et al. 2011, SEDAR 2018, 2020).

Egestion (F) and excretion (Ex) were calculated as a constant proportion of consumption for Pinfish and croaker:

$$(13) \quad F = p_F \cdot C$$

$$(14) \quad Ex = p_{Ex} \cdot (C - F)$$

where p_F is the proportion of consumption and p_{Ex} is the proportion of assimilation food that is excreted. On the other hand, egestion and excretion for Red Snapper and Greater Amberjack were calculated relative to their experienced water temperature (T), consumption rate (C_{max}) and proportion of C_{max} realized:

$$(15) \quad F = \left(\frac{a_F \cdot T^{b_F} \cdot e^{g_F \cdot \frac{C}{C_{max}}}}{0.9} \right)$$

$$(16) \quad Ex = \left(a_{Ex} \cdot T^{b_{Ex}} \cdot e^{g_{Ex} \cdot \frac{C}{C_{max}}} \right) \cdot (C - F)$$

where a_F , b_F , a_{Ex} and b_{Ex} are the respective intercepts and exponents that define the effect of temperature, and g_F and g_{Ex} are the coefficients for the consumption rate effect.

Spawning (S) occurred on a single day (day 201) during daytime hours and was calculated as weight loss by subtracting a proportion of an individuals' biomass. Proportion values varied based on gonadosomatic index values for each species. Red Snapper and croaker eggs were 8% of body weight, Pinfish eggs were 4% of body weight, and Greater Amberjack eggs were 3.35% of body weight (Harris et al. 2007). No fish were mature at the age of one and 50% were assumed to be mature at the age of two. After the age of two, all individuals were assumed to be mature and reproducing.

Movement

Each individual was tracked in continuous, x-y space as described in Campbell et al. (2011). Positions were recorded every hour as x and y distances from the lower left corner of the grid. The individual then experienced the prey and predator biomass and environmental conditions of that cell for that hour. Daytime movement began at hour 1 (sunrise) and spanned a number of hours that varied with the model day according to a function generated from national oceanic and atmospheric administration (NOAA) daily daylight hours data (gml.noaa.gov). Nighttime (sunset) then began once the number of daytime hours had ended.

Recent studies have found that Red Snapper have larger home ranges during the day than at night (Piraino & Szedlmayer 2014, Williams-Grove & Szedlmayer 2017, Gibson Banks et al. 2021). Furthermore, this behavior is thought to be indicative of prey-like reef fish species who

rely on sight to avoid predators (Piraino & Szedlmayer 2014). Thus, for Red Snapper, Pinfish, and croaker, movement consisted of two main behaviors: daytime foraging and nighttime refuge-seeking. Nighttime refuge-seeking movement operated under the assumption that these three species sought refuge as protection from predators during nighttime hours. As a result, if individuals were not on a reef cell (pyramid, rig, or natural), they would travel towards the closest designated reef cell. Once they were on a reef cell, they would remain there until nighttime ended. After nighttime ended, movement for the first two hours off the reef cells were in a random direction and then foraging behavior commenced. Daytime foraging movement operated based on a 25-cell hourly neighborhood search (2-cells in eight directions). The purpose of this neighborhood search was to calculate the projected growth potential of the neighboring cells. An individual would move towards a cell in the neighborhood if the projected growth was 5% greater than their current cell. Bluefish, jack, and Greater Amberjack were assumed to be predatory species and did not exhibit prey-like refuge-seeking behavior. As a result, hourly movement for these species was solely based on foraging using the same criteria described above and did not change with day and night transitions. Optimal foraging cells were selected based on the biomass of the three prey-like species. Habitat designation and distance relative to the closest reef cell were updated hourly for each individual.

Individual movement towards the best cell was performed according to Campbell et al. (2011). Trajectory in radians (θ) towards the best cell was calculated:

$$(17) \quad \theta = \arctan\left(\frac{|Y^c - Y_t|}{|X^c - X_t|}\right)$$

where X_t and Y_t are the current x and y positions and X^c and Y^c x and y positions of the center of the best cell. The trajectory (θ) was then randomly adjusted ± 0.5 rad from a uniform distribution, which prevented all individuals of a certain species from moving identically. I then applied the

following correction to account for any negative values, implying south (down) or west (left) directions, that were lost due to the absolute values from Eq. 14:

$$(18) \quad \theta = \begin{cases} \pi - \theta & \text{if } Y^c - Y_t > 0 \text{ and } X^c - X_t < 0 \\ \pi + \theta & \text{if } Y^c - Y_t < 0 \text{ and } X^c - X_t < 0 \\ 2\pi - \theta & \text{if } Y^c - Y_t < 0 \text{ and } X^c - X_t > 0 \end{cases}$$

The x-axis and y-axis positions were then updated using the calculated trajectories and a random value was applied to adjust distance travelled:

$$(19) \quad x_{t+1} = x_t + ((dist \cdot ranv) + dist) \cdot \cos(\theta)$$

$$(20) \quad y_{t+1} = y_t + ((dist \cdot ranv) + dist) \cdot \sin(\theta)$$

where *dist* is the distance travelled along the trajectory and *ranv* is the random value pulled from a uniform distribution (± 0.3). The same random value was applied to update both the x-axis and y-axis positions. If an individual was projected to travel off the grid at any point, then the individual would move in the exact opposite direction but travel the same distance. In summary, individuals would move towards the best cell in their neighborhood search in unique directions and distances.

Mortality

Four sources of instantaneous mortality occurred every hour: natural, dawn/dusk distance-from-reef, predation, and fishing. Natural mortality was based on data presented in the most recent ‘SouthEast Data Assessment and Review’ (SEDAR) for each species. Dawn/dusk distance-from-reef mortality occurred two hours after daytime began and two hours after nighttime began to represent increased predation mortality during dawn and dusk (Hobson 1972, Helfman 1986, Danilowicz & Sale 1999, Holbrook & Schmitt 2002). This mortality was represented as a function that decreased with increasing distance from the nearest reef cell. Thus, distance mortality was maximized in the eight cells surrounding a reef cell and refuge was

reached if an individual made it to a reef cell. Mortality due to predation occurred during daytime and nighttime hours and was calculated as the density of predatory fish (i.e., Bluefish, jack-like, and Greater Amberjack) in their current cell. Mortality due to predation was maximized at 30% to prevent high densities of predators imposing too large of a mortality effect. Predation was assumed to not occur on reef cells to represent maximum refuge.

Initial Conditions and Recruitment

Initial weights for each species were assigned using mean length-weight relationships from the literature and latest SEDAR (Patterson et al. 2001, Nelson 2002, Barger 1985, Robillard et al. 2009, SEDAR 70). All simulations used a total of 102,000 super-individuals split up according to species for each of the 10 age classes (Table 5-2). Once a cohort reached age-11, the super-individuals of the four focal species were removed and replaced by age-1 super-individuals for each species. The newly introduced super-individuals were assigned initial worths set to the recruitment abundance divided by 1000. Unlike Campbell et al. (2011), my model did not incorporate varying levels of recruitment. Instead, annual recruitment for the four focal species was fixed across the grid at the beginning of each model year (day 1, or July 1). New recruits were placed randomly on the grid and assigned initial weights as defined by the mean weight of age-1 fish for each species (111.5 g ww for Red Snapper, 36.87 g ww for Pinfish, 88.71 g ww for croaker, and 1448.1 g ww for Greater Amberjack).

Cost-Benefit

A cost-benefit ratio was calculated at the end of each model year for reefs based on the total density (biomass/cell size) present on pyramid cells. I also calculated an economically important species (EIS) cost-benefit ratio at each pyramid cell by summing the weighted density (biomass/cell size) of Red Snapper and Greater Amberjack:

$$(21) \quad EIS_j = Pyramid\ Cost \div \left[\sum_1^i \left((B_{ij} \cdot 0.0022046) \left(\frac{Value_i}{Landings_i} \cdot \frac{Value_i}{\sum_1^i Value_i} \right) \right) \right]_j$$

where i represents each species, n_{pyr} is the number of pyramids in the j^{th} pyramid cell, B is the cell biomass (g ww) of species i in the j^{th} pyramid cell (converted from g ww to lbs), and $value$ and $landings$ are the 2020 Texas commercial value (in US\$) and landings (in lbs) for the i^{th} species (fisheries.noaa.gov). Pyramid cost was set to \$1600 based on the estimated cost of constructing and deploying a single prefabricated pyramid (Shively 2014, Gibson Banks et al. 2021).

Simulations

Calibration

This model was calibrated using actual artificial and natural reef configurations and pyramid densities over a 20-year model run. Natural reef cells comprised 128 cells and were oriented to emulate a typical natural reef on the Texas continental shelf. Rig cells comprised 14 cells and were oriented according to an actual rig design (Campbell et al. 2011). Conversely, actual pyramid cells were arranged only in “nearshore” locations (i.e., only on the left-third of the grid) with densities of 4, 23, and 57 pyramids to imitate Port O’Connor nearshore reef, Mustang Island 775, and Rio Grande Valley (see chapter 3).

Calibration began with bioenergetics and then incorporated mortality and movement in a stepwise fashion. I calibrated the bioenergetics of each focal species by adjusting the bioenergetics parameters and half-saturation ($ksat$) values until weights-at-age were similar to those reported in the literature (Patterson et al. 2001, Wilson & Nieland 2001, Nelson 2002, Shipley 2008, Shipley & Cowan 2010, SEDAR 2018, 2020). Bioenergetics parameters were initially calibrated to final weights using simulated temperatures until the ratio of consumption to maximum consumption was around 0.7 (Deslauriers et al. 2017). I then ran the bioenergetics of

each species in the model and adjusted their *ksat* values accordingly until I achieved realistic diets, growth rates, and weights-at-age. Biomass (g ww) and abundance were exported daily to confirm model stability from years 10 to 20.

Basal mortality rates were initially set to the natural mortality rate for each focal species based on their most recent SEDAR but were then adjusted until I obtained realistic densities and stable population biomasses (Table 5-2). I then evaluated movement of each focal species to confirm the desired response to day and night transitions (i.e., daytime foraging and nighttime refuge-seeking) for Red Snapper, Pinfish, and croaker. I also confirmed that Bluefish, jack, and Greater Amberjack movement exhibited signs of tracking their prey biomasses. Finally, I calculated Red Snapper proportional habitat use by dividing the abundance at each reef by the pyramid density, number of rigs, or number of natural reef cells by the abundance and calculated the proportion at each reef type.

Simulations

Overall, I ran 25 simulations across three different simulation experiments to test the presence, density, and placement of artificial reefs on Red Snapper and Greater Amberjack population dynamics (Table 5-4). I focus my results on Red Snapper and Greater Amberjack due to their economic significance and their overwhelming presence at artificial reefs in the Gulf of Mexico (GOM; see chapters 2, 3, and 4). All simulations included 128 natural bank cells to represent a typical natural bank in the GOM. Additionally, all simulations included “actual” reef arrangements in the GOM for both pyramids and rigs. These reef cells were manually placed on the spatial grid with pyramid cells in the left one-third of the grid. For all other reef arrangements (low, medium, and high), reef cells were randomly placed on the spatial grid (Figure 5-3).

Table 5-4. The 25 simulations and the respective number of rigs or pyramid density described as the number of pyramids representing each of the three pyramid cells. *Actual* refers to actual reef arrangements from the Gulf of Mexico for both pyramids and rigs. *High*, *medium*, and *low* refers to the number of rigs or the number of pyramids represented in the three pyramid cells.

Simulation Experiment	Simulation Category	Simulation	Pyramid Density/Number of Rig Cells
Control	No Pyramid	No Rig	-
1	No Rig	Low Pyramid	4, 4, 4
		Medium Pyramid	23, 23, 23
		High Pyramid	57, 57, 57
		Actual Pyramid	4, 23, 57
2	No Pyramid	Low Rig	9
		Medium Rig	16
		High Rig	20
		Actual Rig	14
3	Low Pyramid (4, 4, 4)	Low Rig	9
		Medium Rig	16
		High Rig	20
		Actual Rig	14
	Medium Pyramid (23, 23, 23)	Low Rig	9
		Medium Rig	16
		High Rig	20
		Actual Rig	14
	High Pyramid (57, 57, 57)	Low Rig	9
		Medium Rig	16
		High Rig	20
		Actual Rig	14
Actual Pyramid (4, 23, 57)	Low Rig	9	
	Medium Rig	16	
	High Rig	20	
	Actual Rig	14	

Differences in artificial reef type and density (pyramids only) were expressed as different weighted distances (Table 5-5). As aforementioned, nighttime refuge-seeking movement was based on distances from a reef cell. Distance weightings were applied to each reef habitat to represent the relative “discoverability” of each habitat by Red Snapper, Pinfish, and croaker. I assumed that prefabricated pyramid reefs would be more “discoverable” with a higher density of

pyramids due to a larger surface area for fish to seek refuge (Chandler et al. 1985, Charbonnel et al. 2002, Quimbayo et al. 2019). Rig cells were weighted the highest, or most discoverable, and a natural cell was weighted the lowest, or least discoverable (Table 5-5). Along with distance weightings, I imposed density caps for Red Snapper on each reef type to mirror Red Snapper densities observed on these reefs in the Gulf of Mexico (Stanley & Wilson 2001, 2003, Gallaway et al. 2009, Ajemian et al. 2015a, Streich et al 2017a,b, Gallaway et al. 2021, see chapters 2, 3, and 4). While the density caps represented in Table 5-5 are not realistic due to the number of model individuals that were chosen, the ratios of rigs to pyramids to natural cells are representative of observed ratios in the GOM (see chapters 2, 3, and 4).

Table 5-5. Defines the structure density (units per cell), distance weighting, Red Snapper density caps (fish/m²) for each reef type.

Reef Type	Structure Density (units per cell)	Distance Weighting	Red Snapper Density Cap (fish/m²)
	4	0.95	0.49
Pyramid	23	0.85	0.49
	57	0.80	0.49
Rig	1	0.60	4.19
Natural	1	1	0.35

Experiments 1 and 2 were designed to isolate each artificial reef type to better understand the impact of reef presence as well as the influence of structure density. In experiment 1, I simulated the sole presence of three pyramid cells, in conjunction with the natural cells, of four different structure densities (low, medium, high, and actual; Table 5-4). In experiment 2, rigs were also examined exclusively under four different scenarios representing differing placement and number of structures (Table 5-4). Finally, experiment 3 was constructed to test the interaction between differing number of rigs and pyramids.

Similar to Campbell et al. (2011), I summarized differences among simulation experiments by comparing Red Snapper and Greater Amberjack average weights at age-5, total grid-wide abundance and biomass, and daily distance traveled (Table 5-6). Daily distance traveled was calculated as total distance traveled during daytime hours beginning at sunrise. I also summarized foraging distance, hours at risk (i.e., number of hours off a reef cell during nighttime hours), and unique number of rig, pyramid, and natural cells visited per week for Red Snapper (Table 5-6). Forage distance was calculated as the total distance traveled from the originating reef cell during daytime hours. Finally, total biomass of all six species, economically important species (i.e., Red Snapper and Greater Amberjack) value, and age-based habitat use was evaluated for each reef type (Table 5-6). I calculated habitat use as the average total number of unique natural, pyramid, or rig cell visited for Red Snapper and Greater Amberjack by the number of natural or rig cells or by the number of pyramids. A proportion for each reef type was then generated within each age class. For each parameter comparison, I plotted the average values for each simulation as rig-dependent plots. Rig-dependent plots were designed to evaluate changes in parameters over differing number of rig cells (Table 5-4). If rig-dependent plots identified an interaction with changes in pyramid density, then a pyramid-dependent plot was generated.

Table 5-6. Model outputs for both the calibration (above gray line) and simulation experiments (below gray line). *EIS* refers to economically important species, *RS* is Red Snapper, *PF* in Pinfish, *CR* is croaker, and *GA* is Greater Amberjack.

Output Variable	Species	Frequency Exported	Computation
Weight at Age	RS, PF, CR, GA	Daily from years 10 to 20 at hour 24	Average of all values for each age (1-10)
Diet	RS, PF, CR, GA	Hourly during day 56 of year 20	Proportion of average biomass eaten of each prey type by each species for every 100th model individual
Abundance	RS, PF, CR, GA	Daily from years 10 to 20 at hour 24	Abundance of age 2+
Biomass	RS, PF, CR, GA	Daily from years 10 to 20 at hour 24	Biomass of age 2+
Hourly Position	RS, GA	Hourly from day 200 to 204 of year 20	Position on the model grid
Abundance by Age by Habitat Type	RS	Hour 24 from day 20 of year 20	Proportion of total abundance of each age (1-10) by habitat type
Biomass Snapshot	All six species	Hours 3, 9, 15, 21 from day 56 of year 20	Natural log of biomass
Prey Biomass Snapshot	All five prey types	Hour 24 from day 56 of year 20	Proportion of carrying capacity
Weight at age-5	RS, GA	Daily from years 20 to 50 at hour 24	Average of all values for each species age-5
Abundance	RS, GA	Daily from years 20 to 50 at hour 24	Average of age 2+ abundance each species
Biomass	RS, GA	Daily from years 20 to 50 at hour 24	Average of age 2+ abundance each species
Day Distance Traveled	RS	Daily from years 20 to 50 at hour 24	Average of total distance traveled for each species for that day during daytime hours
Day Forage Distance	RS	Daily from years 20 to 50 at hour 24	Average of total distance traveled from the originating reef cell for that day during daytime hours
Hours at Risk	RS	Daily from years 20 to 50 at hour 24	Average of the total number of hours spent off a reef cell during nighttime hours
Unique Rig Cells Visited	RS	Weekly from years 20 to 50	Average of weekly total number of unique rig cells visited
Unique Pyramid Cells Visited	RS	Weekly from years 20 to 50	Average of weekly total number of unique rig cells visited
Unique Natural Cells Visited	RS	Weekly from years 20 to 50	Average of weekly total number of unique rig cells visited
Total Biomass per Habitat Type	All six species	Hourly from day 56 of year 50	Average of total biomass of each species (in lbs) within each hour by habitat type
EIS Value	RS, GA	Hourly from day 56 of year 50	Average of total weighted value of each species (in US\$, based on commercial landings) within each hour by habitat
Habitat Use by Age	RS	Weekly from years 20 to 50	Average of proportion of unique types of reefs visited each week within each age

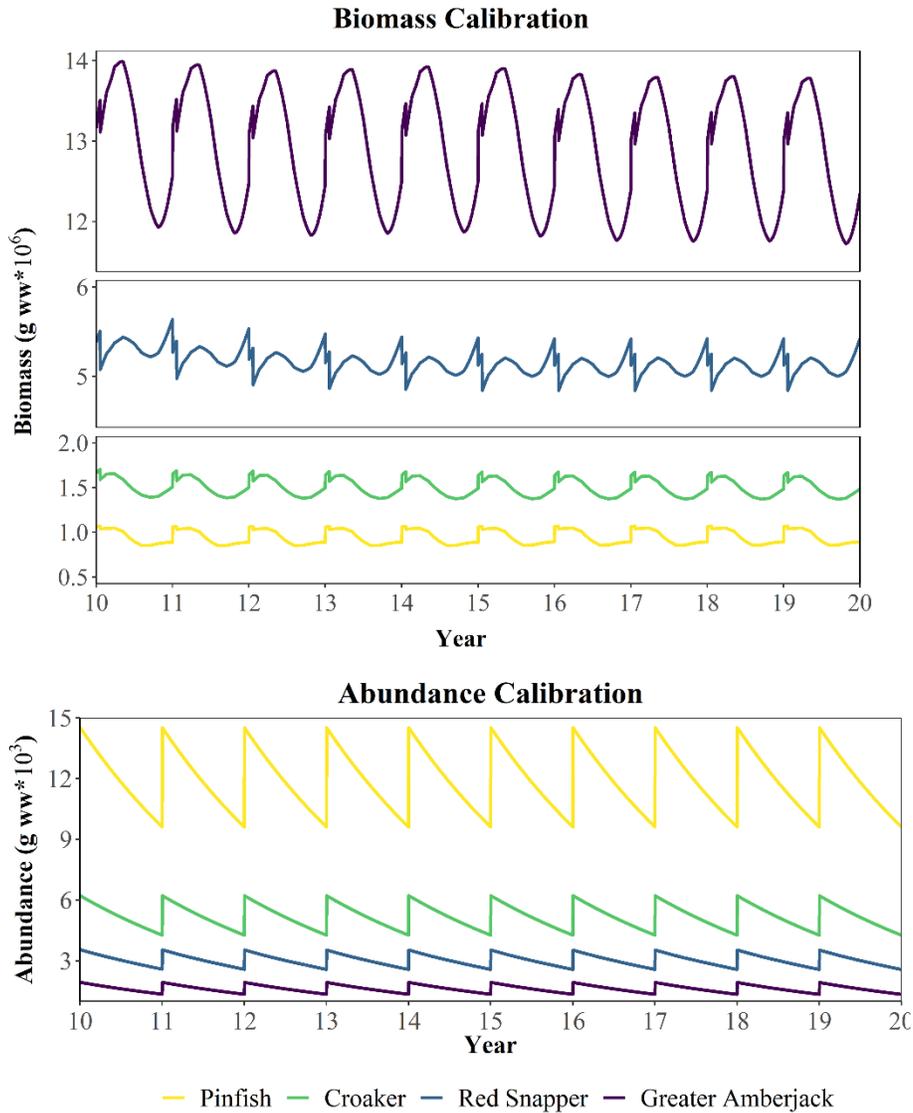


Figure 5-3. Daily values from years 10 to 20 of Pinfish, croaker, Red Snapper, and Greater Amberjack (a) biomass and (b) abundance.

Results

Calibration

Abundance and biomass for each of the four focal species confirmed a steady-state in the model after year 12 (Figure 5-3a,b). Of the four focal species, Greater Amberjack dominated the biomass (63%), followed by Red Snapper (25%), croaker (7.3%), and Pinfish (4.6%). Biomass for Greater Amberjack, Pinfish, and croaker decreased within each year, indicating that mortality

overrode growth for these species (Figure 5-3a). Conversely, Red Snapper growth exceeded mortality, which was indicated by an increase in biomass within each year (Figure 5-3a).

I adjusted half-saturation values until simulated mean weights-at-age were similar to published data for each focal species (Figure 5-4; Patterson et al. 2001, Wilson & Nieland 2001, Nelson 2002, SEDAR 2018, 2020). Weights for each species also appeared to have similar functional responses to age to observed data (Figure 5-4). Diets for each species represented appropriate proportions from published data and were competitors to Red Snapper (Figure 5-5). Red Snapper consumed about 41% benthic fish, 21% pelagic fish, 20% zooplankton, 12% shrimp, and 5% crab, which were similar representations of diet for this species in the GOM (Szedlmayer & Lee 2004, Shipley 2008, Wells et al. 2008). Greater Amberjack consumed about 89% fish and 11% shrimp, which were also similar to observed data (Figure 5-5; Manooch & Haimovici 1983, Piptone & Andoloro 1995, Foss 2016). Likewise, Pinfish and croaker exhibited diets similar to published data for these species (Figure 5-5; Overstreet & Heard 1978, Thompson 2000, Murphy 2006, Campbell et al. 2011, Faletti et al. 2019).

Similar to Campbell et al. (2011), prey halos enveloped each reef type as fish foraging reduced the biomass for each prey type (Figure 5-6). Comparatively, biomass distributions of the focal species mirrored prey halos around reef cells (Figure 5-7). Additionally, Red Snapper demonstrated more widespread biomass from reefs than Pinfish or croaker due to the imposed density caps (Figure 5-7). Hourly movement for both the prey (Red Snapper, Pinfish, and croaker) and predator (Bluefish, jack, and Greater Amberjack) species were confirmed to exhibit desired responses to day/night transitions and to prey biomass (Figure 5-8). Red Snapper also exhibited a response to density caps imposed on each reef type by changing direction to a different reef cell during nighttime hours when the density cap of the originally targeted reef had

been reached (Figure 5-8). Red Snapper showed loose shifts in abundance at reef cells with age that are indicative of natural habitat shifts along the Texas continental shelf (Figure 5-9; Gallaway et al. 2009, Dance & Rooker 2019, see chapters 3 and 4).). More specifically, Red Snapper increased their use of natural cells and decreased their use of pyramid cells with age (Figure 5-9). Greater Amberjack illustrated prey-tracking behavior by remaining around reef cells (Figure 5-8).

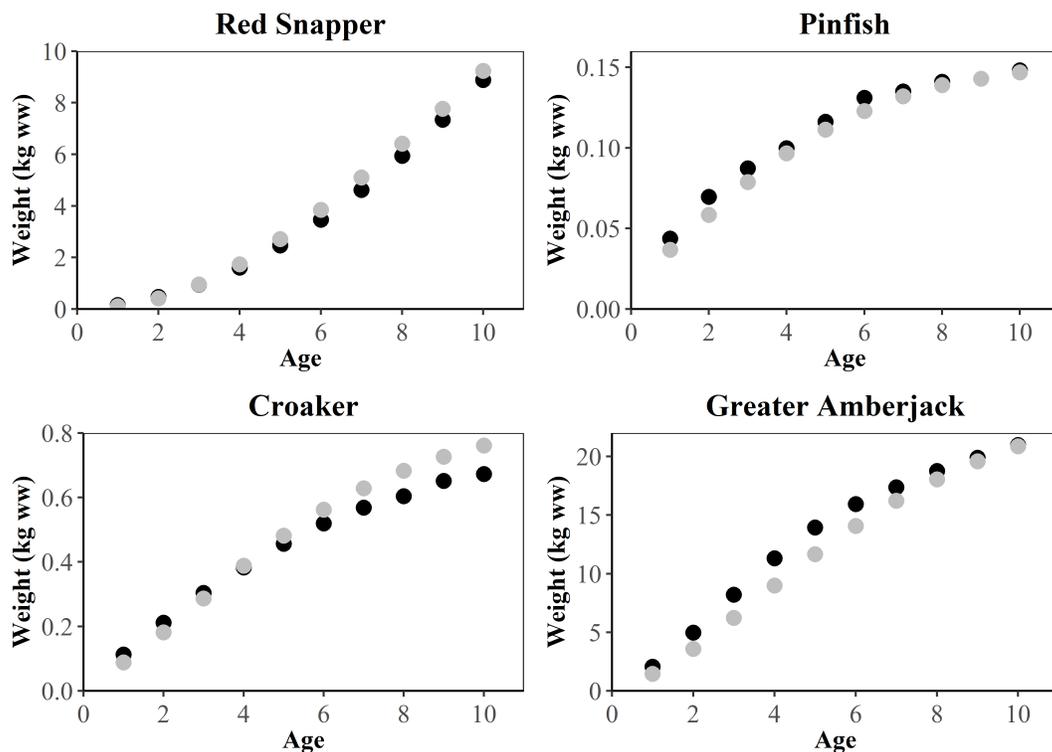


Figure 5-4. Average modeled (black) and observed (gray) weights-at-age for each of the four focal species (in kg ww).

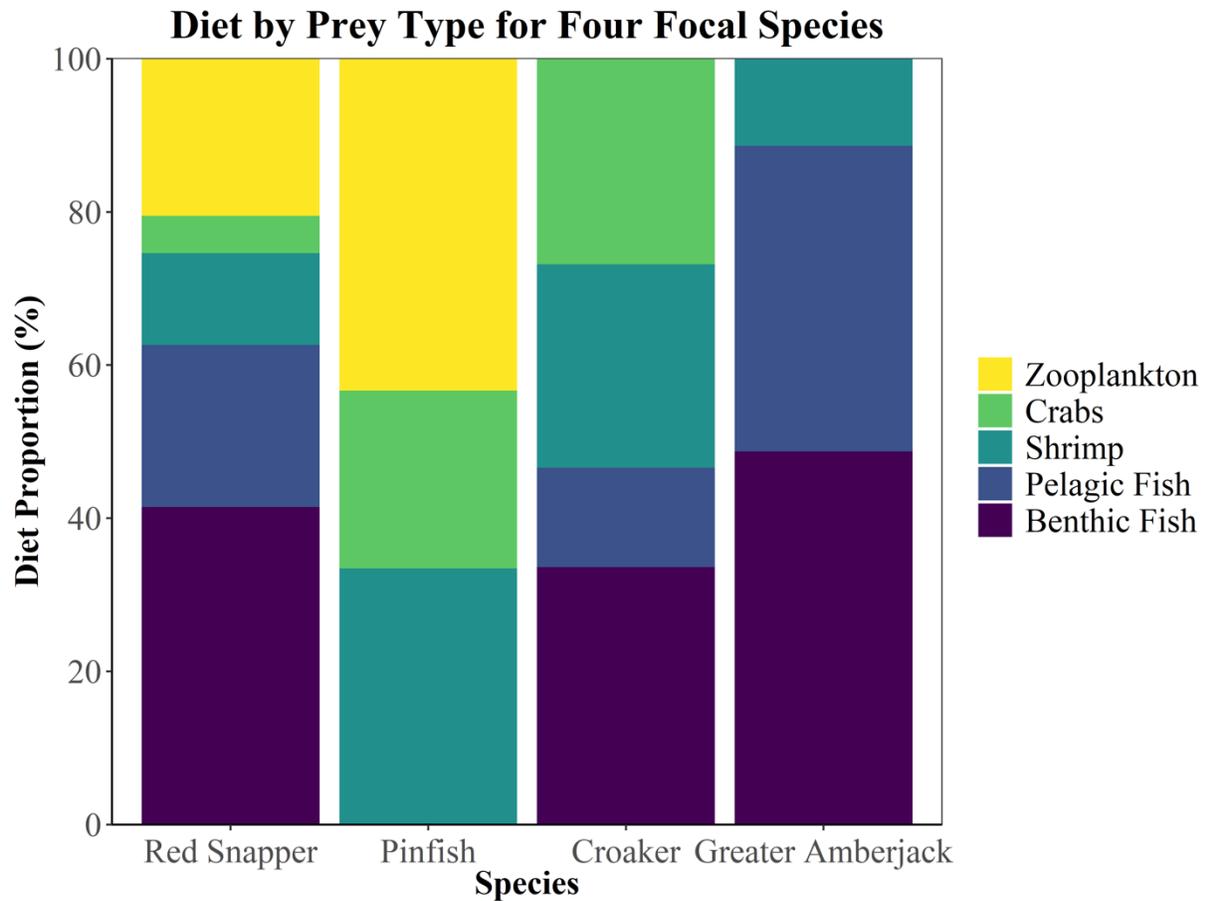


Figure 5-5. Proportion (in %) of each of the five prey types consumed by the four focal species.

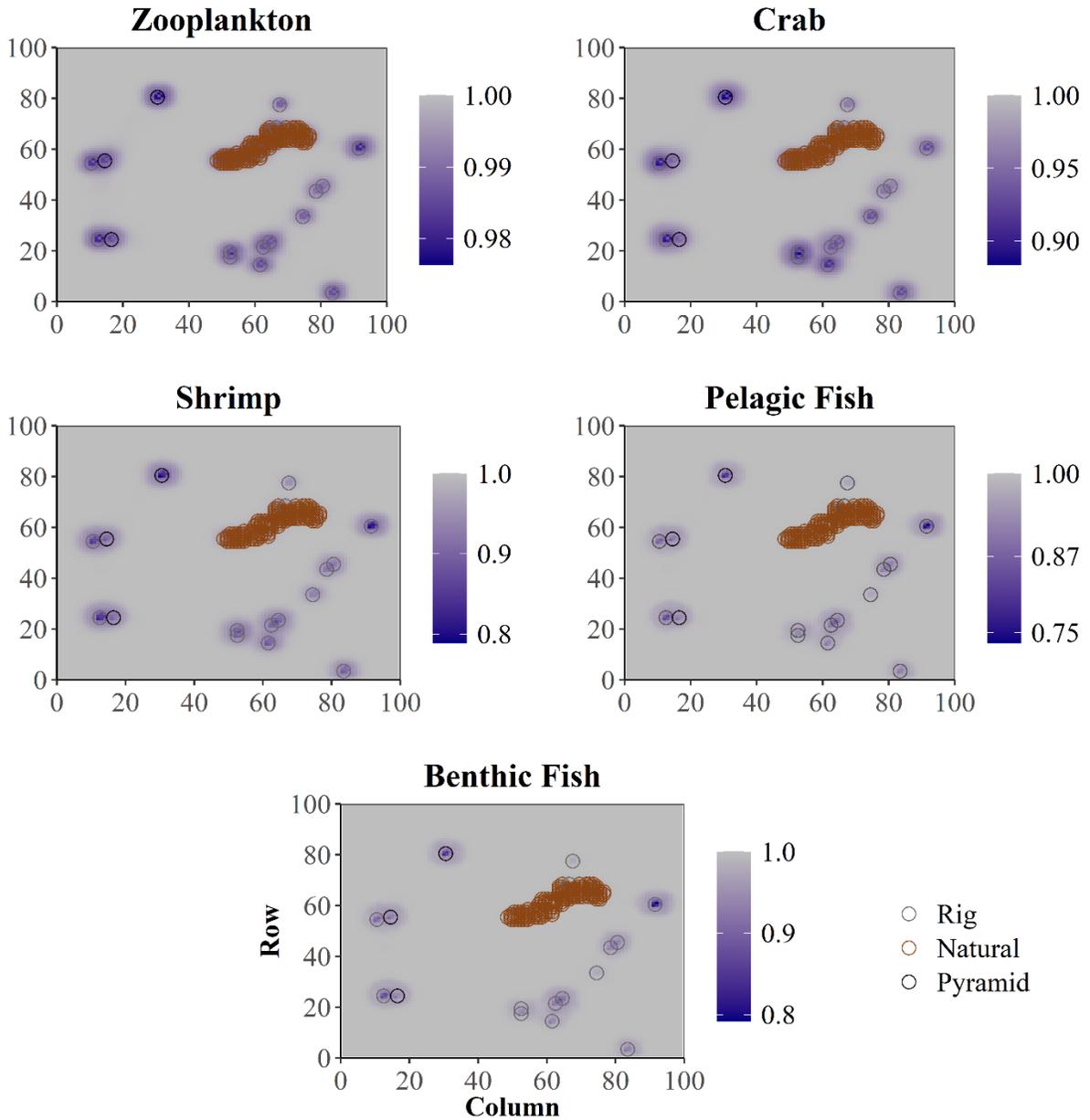


Figure 5-6. Biomass of each prey type divided by their respective carrying capacities and mapped on the model grid. Prey halos were identified around each artificial structure. Colored circles represent different types of reefs.

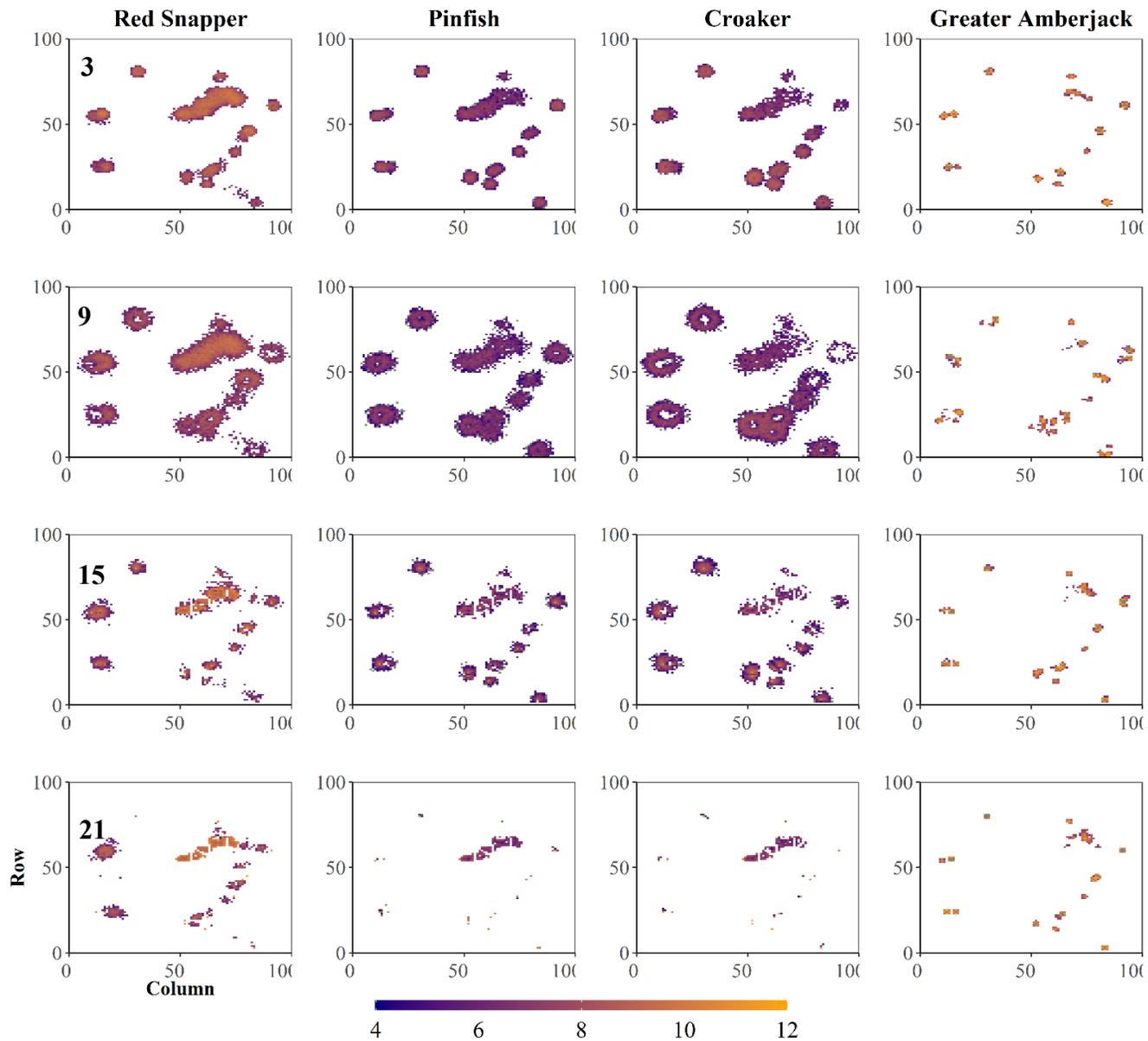


Figure 5-7. Log-transformed biomass for each of the four focal species at hours 3, 9, 15, and 21 during day 56 of year 20.

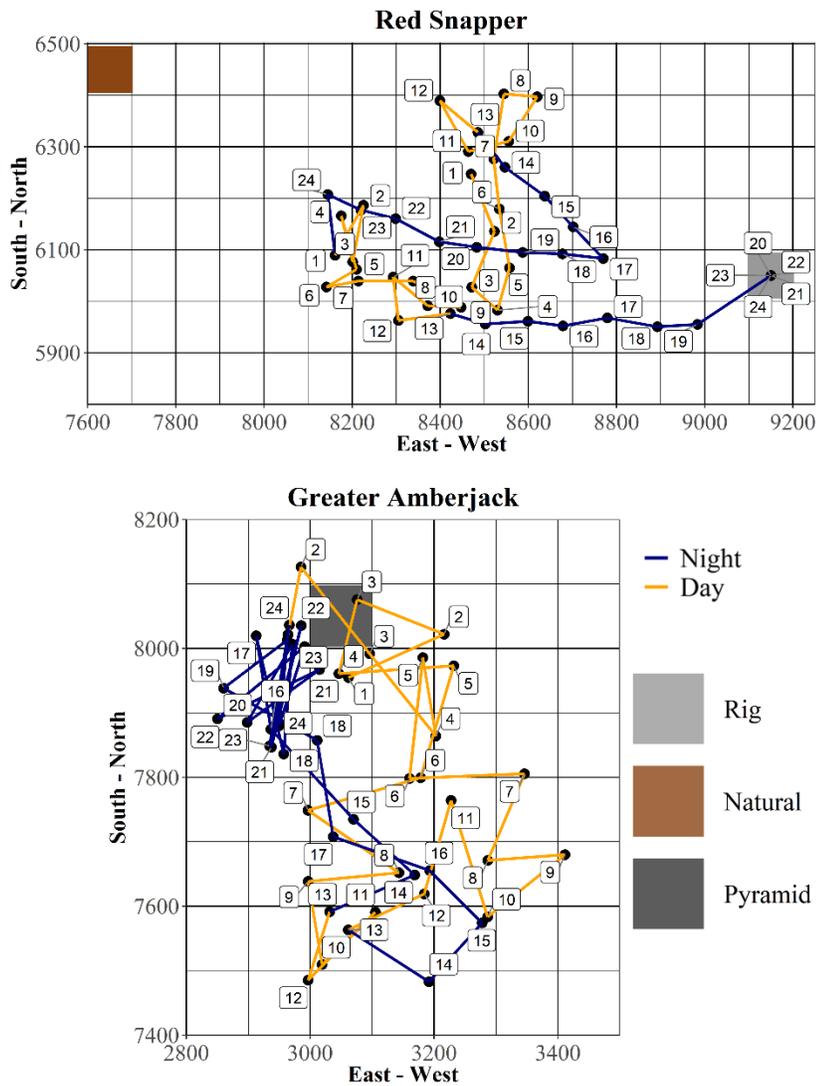


Figure 5-8. Hourly movement for day (orange) and night (blue) over a two-day period for Red Snapper (top) and Greater Amberjack (bottom). Colored squares indicate either a rig (gray), pyramid (black), or natural (brown) reef cell. Numbered boxes indicate the hour for the corresponding point location.

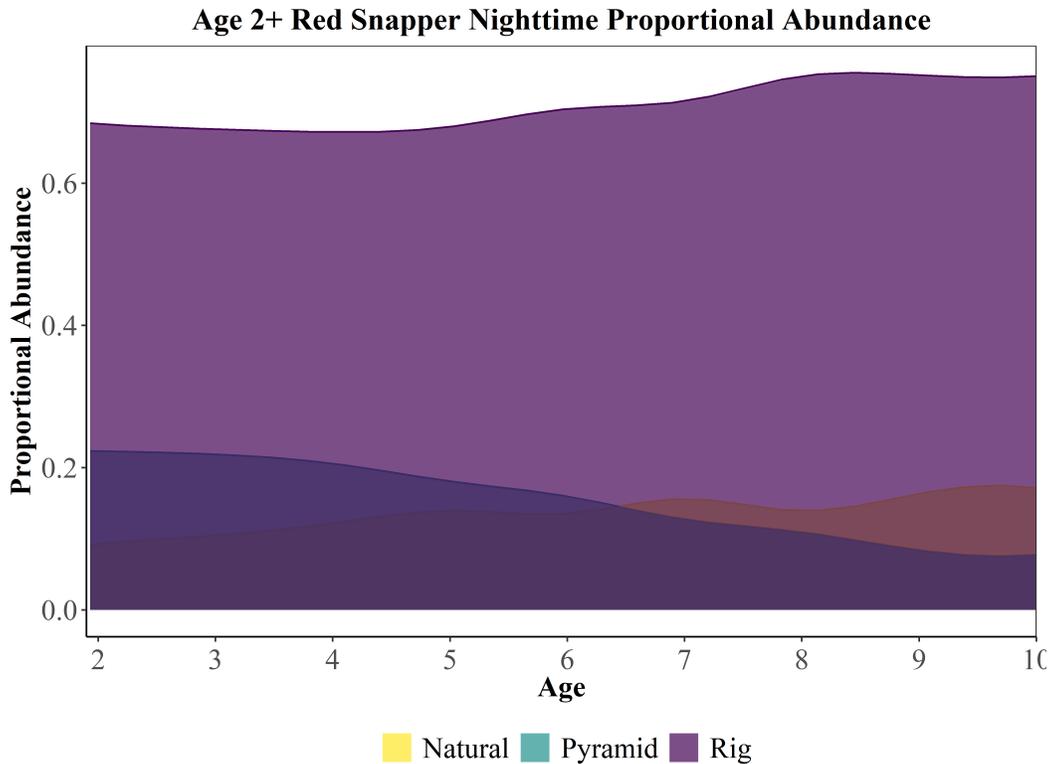


Figure 5-9. Age 2+ Red Snapper proportional abundance at each habitat type within each age class. Proportional abundance values are on a scale of 1 to 10 within each age class. Spline interpolation was used to smooth point connections between ages.

Simulation Experiments

Overall, increasing the number of rigs resulted in an increase in the average weight-at-age 5 for both Red Snapper and Greater Amberjack (Figure 5-10). Grid-wide age 2+ Red Snapper abundance decreased with increasing number of rigs (Figure 5-10). Conversely, Greater Amberjack abundance exhibited an initial decrease in abundance from “none” to “low”, but then increased with increasing number of rigs. Similar to average age-5 weights, both Red Snapper and Greater Amberjack grid-wide biomass increased with increasing number of structures. Interestingly, the actual pyramid simulations had relatively low abundances for both species. However, patterns across pyramid simulations appeared to be the similar for age-5 weights, abundance, and biomass, indicating that pyramid densities did not have profound effects on these factors (Figure 5-10).

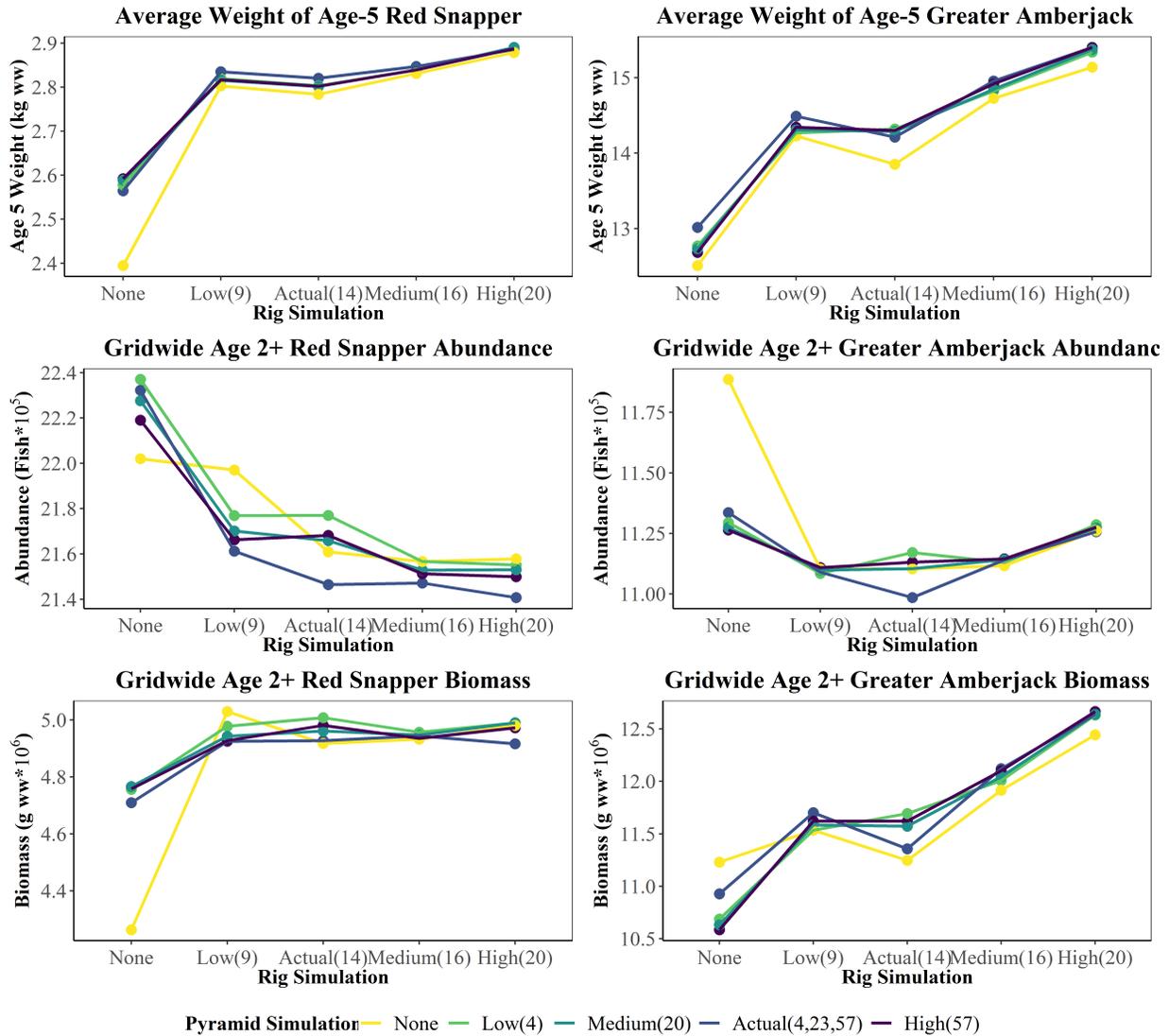


Figure 5-10. Red Snapper and Greater Amberjack average weight (kg ww) at age 5 and grid-wide abundance (fish*10⁵) and biomass (g ww*10⁶) for each pyramid simulation (colored lines) over differing rig simulations. Rig simulations on the x-axis increase in number of structures from left to right.

Average Red Snapper daily distance traveled and number of hours at risk was maximized during the “low” rig simulation and was minimized with no rigs (Figure 5-11). Red Snapper in the “low” rig simulations were required to travel further distances to meet their growth requirements, which resulted in an increase in the number of hours at risk, which carried a respiration penalty. These results are reflected in the comparatively low Red Snapper biomass observed for a low number of rigs compared to other simulations where rigs were present (Figure

5-10). Furthermore, the “actual” rig simulation appeared to have the lowest daily distance, forage distance, and number of hours spent off a reef cell at night among simulations with rigs present (Figure 5-11). In general, forage distance decreased with increasing number of structures (Figure 5-11). Average Greater Amberjack daily distance did not vary by more than 1 m across all simulations and was thus excluded from this portion of the analysis.

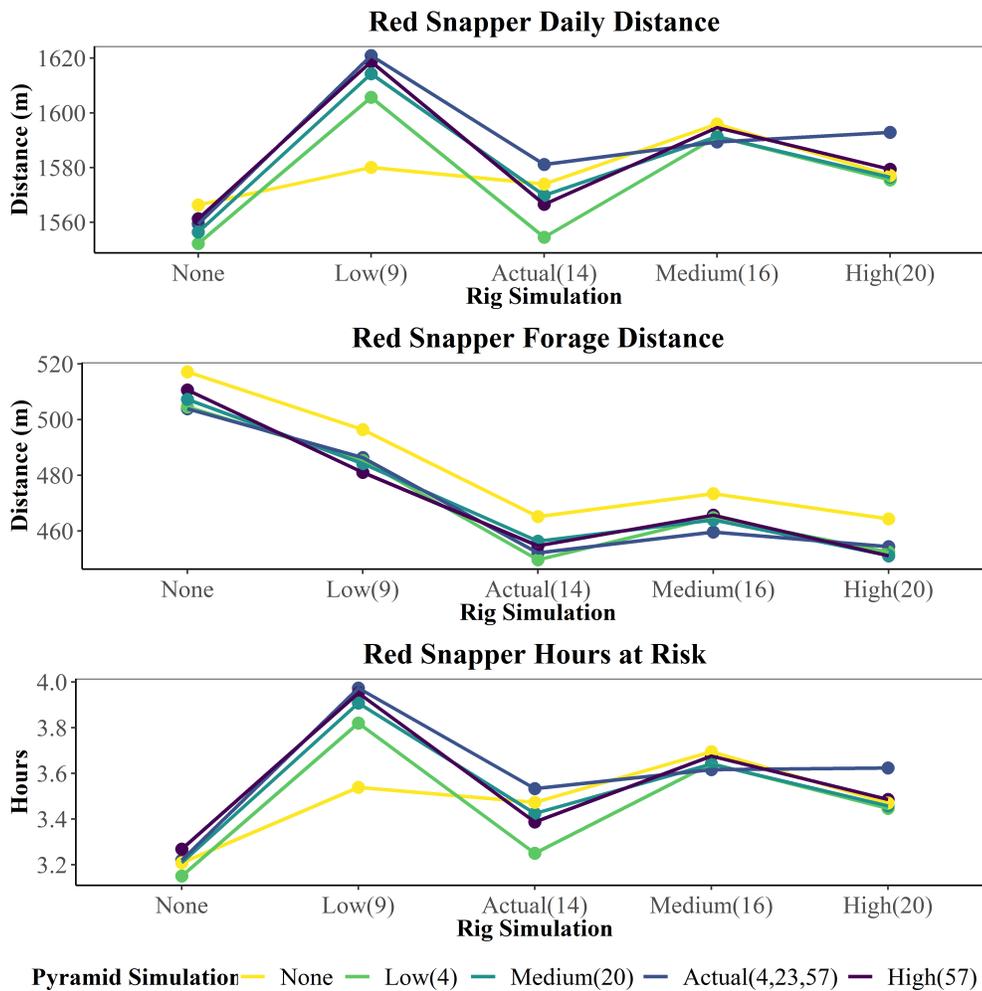


Figure 5-11. Red Snapper average daily distance traveled (m), forage (daytime) distance traveled (m), and average number of hours at risk during daytime for each pyramid simulation (colors) over differing rig simulations. Hours at risk refers to the average daily total number of hours spent off a reef cell during nighttime hours. Rig simulations on the x-axis increase in number of structures from left to right.

Average total biomass for all six species was divided by the number of rig, pyramid, or natural cells within each simulation to represent a value (in lbs) per number of reef cells for each

reef type. Overall, rigs had higher total biomass and EIS value than pyramid or natural reefs. Total biomass and EIS value at rigs decreased with increasing number of rigs. Total biomass and EIS value at natural reefs appeared unchanging with increasing number of rigs and pyramids did not exhibit an identifiable trend (Figure 5-12). Pyramid cost-benefit ratio (US\$/pyramid) exhibited an overall decrease with increasing pyramid density (Figure 5-13). These results reflect the distance weightings that were applied to each pyramid cell according to their density (Table 5-5).

Overall, EIS proportional use of artificial reef cells decreased with age and use of natural cells increased with age (Figure 5-14). Furthermore, the “actual” rig configuration appeared to exact the highest proportional use of rigs for EIS (Figure 5-14). Pyramid density (top to bottom in Figure 5-14) did not appear to have an effect on proportional habitat use; however, there was a general increase in rig use with increasing number of rig cells (left to right in Figure 5-14). These results are mirrored in Red Snapper and Greater Amberjack grid-wide biomass (Figure 5-10), where I observed an increase in biomass with increasing number of rig cells. Because the “actual” rig configuration appeared to respond differently to numerous parameters, I calculated the minimum distance between reef cells and averaged the values across the grid for each model grid. This value acted as an overall indicator for habitat connectivity for each simulation. Unsurprisingly, the “actual” rig configuration had the shortest average distance between reef cells for all simulations (Figure 5-15). In general, closest reef distance decreased with increasing number of rig cells (Figure 5-15).

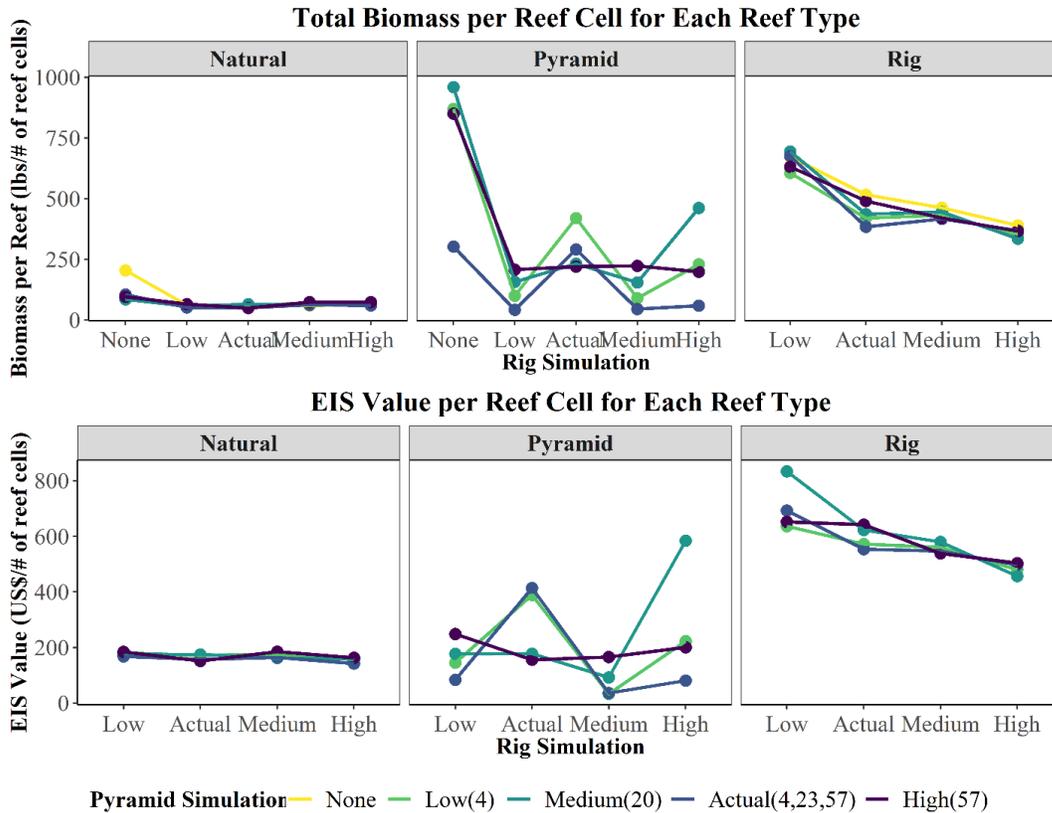


Figure 5-12. Average total biomass (in lbs) for all six model species (top) and average total commercial value (in US\$) for economically important species (Red Snapper and Greater Amberjack; bottom). Both values were divided by the number of cells for each reef type to represent a value per reef cell for each reef type. *EIS* refers to economically important species.

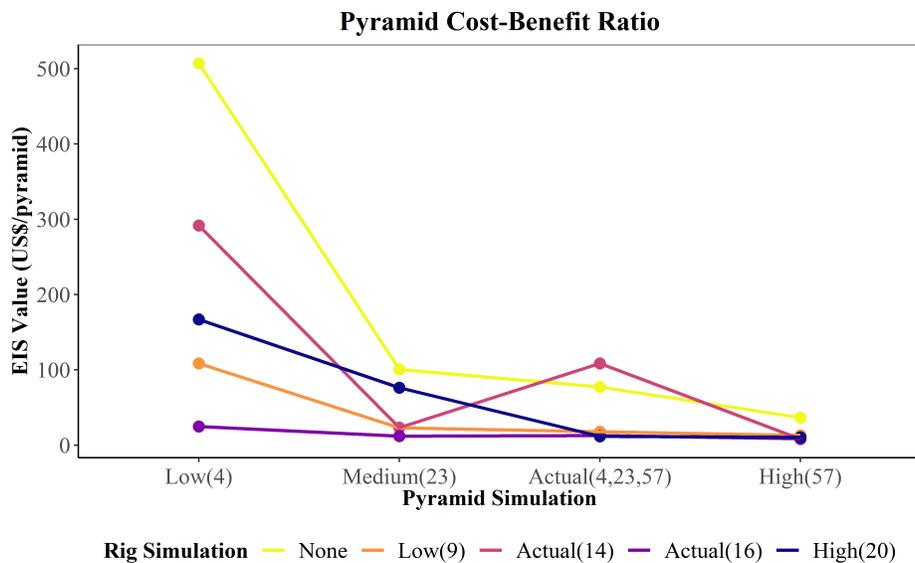


Figure 5-13. Economically important species cost-benefit ratio (in US\$/pyramid) for each rig simulation (colored lines). Pyramid simulations on the x-axis are arranged by increasing density from left to right.

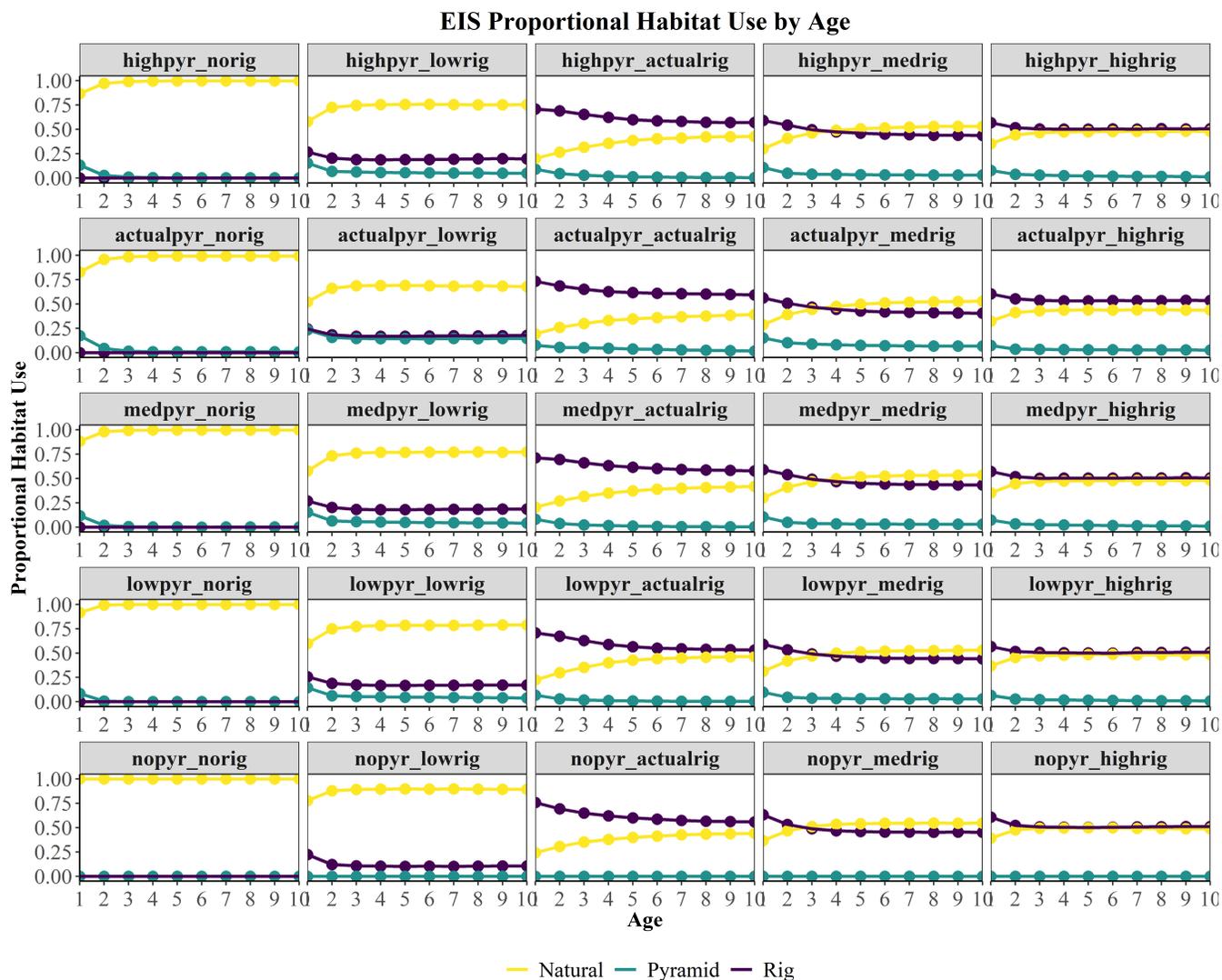


Figure 5-14. Proportional habitat use for economically important species (*EIS*) by age for each of the 25 simulations. Habitat use was calculated by dividing the total number of unique rig, pyramid, or natural cells by the number of cells present for each reef type in the simulation. Proportional use was then calculated for each reef type within each age class. Number of rig cells increases from left to right, while pyramid density increases from bottom to top.

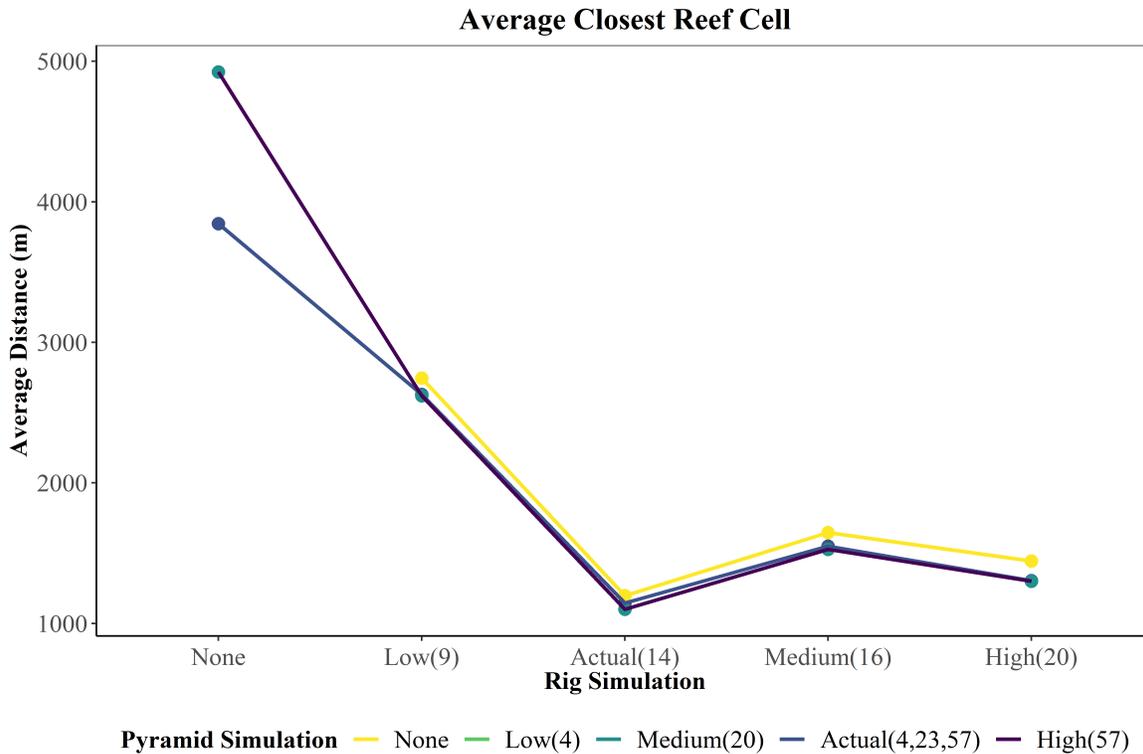


Figure 5-15. Average distance (in m) of the closest reef cell to each reef cell as an indicator for habitat connectivity across each model grid. Number of rig cells increases from left to right across the x-axis.

Discussion

In this study, I developed an individual-based model to assess the population dynamics of two economically important fish species over differing densities of artificial reefs. I expanded the model described in Campbell et al. (2011) to simulate natural behaviors and capture Red Snapper age-based habitat partitioning similar to those observed in the GOM (Dance & Rooker 2019). My results identify the importance of artificial reefs in the growth of Red Snapper and Greater Amberjack, particularly in areas, like the GOM, where there are vast expanses of sand and mud bottom with relatively little structured habitat available for fishes. While the ideal artificial reef configuration remains unclear, my model illustrates the importance of habitat connectivity, which should be considered in future reefing projects.

Comparisons to Original Model

Although obtaining the ideal artificial reef configuration was outside the scope of this study, the importance of artificial reef placement is supported by comparing these results to those from Campbell et al. (2011). For example, Red Snapper abundance and biomass consistently increased with increasing number of rigs when rigs were fixed in a grid-like orientation (Campbell et al. 2011); however, random or actual rig placement do not express the same consistent increase with increasing number of rigs. In fact, I observed a decrease in Red Snapper abundance with increasing number of structures. Actual reef layouts in Campbell et al. (2011) also resulted in inconsistent relationships of abundance and biomass to the number of structures. These results indicate that Red Snapper are not habitat limited, but it is likely that other factors such as reefing location and orientation may be just as impactful as number of structures. Fewer rigs resulted in higher abundances of Red Snapper with no rigs experiencing the highest abundances of both EIS. Differences between Campbell et al. (2011) and this study can be explained by the numerous changes that were made to update the model based on recent literature and to address different research questions.

While this model used the same foundation as the model described in Campbell et al. (2011), several key parameters were included or adjusted to fit the desired research questions. First, my model included pyramid and natural reefs in addition to rigs and randomly placed them on the grid, with the exception of the “actual” simulations. With the addition of pyramid habitats, I adjusted the model cell size from 200 to 100 m*side⁻¹ to more accurately reflect the area of influence that a pyramid reef only containing four pyramids may have. Smaller cell sizes resulted in a smaller spatial grid than the original model and the number of model individuals was reduced. Decreasing the number of model individuals required reductions in natural mortality for

Pinfish and croaker, while Red Snapper mortality was increased based on the most recent stock assessment for this species (SEDAR 2018). I also imposed different density caps and distance weightings on reef cells to limit Red Snapper densities by reef type to replicate population distributions in the GOM (Ajemian et al. 2015a,b, Streich et al. 2017 a,b, Gallaway et al. 2021, Stunz et al. 2021, see chapters 2 and 3). Furthermore, because Red Snapper have been observed displaying cross-shelf movement thought to be temperature driven (Dance & Rooker 2019), I included a temperature gradient to simulate a change in depth from west to east. Likewise, I adjusted Red Snapper optimal temperature to decrease with age to reflect ontogenetic behavior (Dance & Rooker 2019). This model also fluctuated the number of daylight hours in a day throughout the year and adjusted Red Snapper movement patterns based on recent evidence that this species ventures further from the reef during the day than at night (Piraino & Szedlmayer 2014, Williams-Grove & Szedlmayer 2017, Gibson Banks et al. 2021). Greater Amberjack was included as an additional predatory economically important fish species, for which a bioenergetics model had to be built. With the inclusion of an additional predator, I limited the distance-from-reef mortality to 2-hours after sunrise and 2-hours after sunset to simulate increased mortality due to predation at dawn and dusk (Hobson 1972, Helfman 1986, Danilowicz & Sale 1999, Holbrook & Schmitt 2002). Finally, I used the commercial value of both EIS along with a cost-benefit ratio to evaluate economic benefits over differing rig and pyramid densities.

Red Snapper Movement

This study intrinsically imitated Red Snapper age-based habitat partitioning, which typically resulted in the majority of the age 5+ population occupying natural reef cells. These results may indicate that, due to their large spatial area, natural reefs more efficiently distribute the population, which reduces competition for prey and optimizes foraging. For example, prey

halos around the natural reef were only present in simulations with no rigs or artificial reefs. Conversely, prey halos were always present on the pyramid or rig cells but decreased with an increase in the number of rig cells. Likewise, Red Snapper were less reliant on the natural reef with an increase in the number of rigs. Prey halos are known to occur at artificial reefs in the GOM (Lindberg et al. 1990, Strelchek et al. 2007, McCawley & Cowan 2007) and because Red Snapper are attracted to artificial reefs due to behavioral preference rather than foraging potential, their fitness may be negatively affected by these prey halos (McCawley & Cowan 2007, Daigle 2011).

The negative impacts of prey halos can lead to reduced growth potential around artificial structures, which may result in habitat transitions with size and age to more profitable areas. While Red Snapper less than age 10 are well known to display site fidelity towards standalone artificial reefs or artificial reef complexes in the GOM (Patterson & Cowan 2003, Topping & Szedlmayer 2011, Piraino & Szedlmayer 2014, Williams-Grove & Szedlmayer 2017, Froehlich et al. 2019, Gibson Banks et al. 2021), they are also known to increase their home-range and time at liberty with size and age (Diamond et al. 2007, Piraino & Szedlmayer 2014). These findings suggest a decrease in the reliance of complex structures for protection and/or an increase in growth requirements with size and age (Diamond et al. 2007, Piraino & Szedlmayer 2014). While my model did not adjust movement speed, distance, or home range with age, Red Snapper still gravitated towards the large area of the natural reef over time simply due to the inclusion of a density cap on all reef cells; thus, indicating the importance of competition and growth requirements as a driver for Red Snapper movement. Therefore, Red Snapper movement may transition from habitat-centric to forage-centric as their growth requirements increase and their need for protection decreases.

Aiding in the transition to meeting fish growth requirements, habitat connectivity affects the potential movement to more profitable habitat as fishes are able to more easily migrate to different structures the closer they are together. When rigs were present, patterns in Red Snapper daily distance, forage distance, and hours at risk nearly mirror the average closest reef, or the indicator for habitat connectivity across the grid. These results are the product of two important functions: meeting growth requirements and competition. As previously discussed, larger prey halos were observed at artificial reef cells with fewer artificial structures present. This directly resulted in Red Snapper traveling further distances from the reef cell to fulfill their energy requirements. Likewise, if a fish returning to the reef cell at night encountered a density cap, it would target a different reef cell on the grid and attempt to reach that cell before sunrise. Both of these functions are governed by habitat connectivity. The shorter the distance between reefs, the more evenly distributed the population, the smaller the prey halos, and the fewer number of hours at risk either from foraging or to transition away from highly dense reefs. Therefore, future artificial reef projects should consider the relationship of the newly constructed reef to other reefs (natural or artificial). Moreover, the placement of reefs in appointed reefing blocks may increase habitat connectivity and allow for greater distribution of prey; however, considerations must be made to avoid overlapping prey halos, which could negatively affect fitness (McCawley & Cowan 2007).

Artificial Reef Value

Despite the potential growth and distribution limitations with fewer rigs present, total biomass and EIS value were higher per rig with fewer rigs and decreased with increasing number of rigs. I would like to clarify that this does not directly imply that overall value is higher with fewer rigs, rather that value per rig decreased with increasing number of rigs due to fish

populations being more concentrated around fewer rigs. Grid-wide Red Snapper and Greater Amberjack biomass generally exhibited a positive relationship with number of rigs suggesting that the overall EIS populations are healthier with more rigs. Furthermore, pyramid cost-benefit ratio indicated that the most efficient pyramid configuration contained the highest number of pyramids. Pyramid efficacy was also “best” during high-rig simulations regardless of pyramid density. These results may also support the significance of habitat connectivity and future pyramid reefing projects should consider the relationship and density of other reefs (natural or artificial) when deciding placement of the reef (see chapters 3 and 4). Additionally, prefabricated pyramid reefs have been observed playing a role in providing habitat for younger cohorts of Red Snapper and Greater Amberjack (Streich et al. 2017b, Dance & Rooker 2019, see chapters 3 and 4), but assessing the impact of pyramids on recruitment was outside the scope of this study. To further explore prefabricate pyramid artificial reefs, future studies should anticipate adapting the model to assess pyramid placement within a reef as opposed to simply the pyramid reef as a whole.

In summary, this model identified the importance of artificial reef placement and potential impact of distance between artificial reefs and other reefs on population dynamics of Red Snapper and Greater Amberjack. More refined data on juvenile recruitment of Red Snapper to pyramid reefs could improve future attempts to capture age-based habitat partitioning. In the meantime, incorporation of varying levels of habitat-based recruitment at reefs may lead to further understanding of the mechanisms driving Red Snapper transitions to different habitats. More data on predation risk around artificial and natural reefs and ontogenetic shifts in habitat are also needed to improve the model. Until more data are available on Greater Amberjack bioenergetics, movement, and distributions, this model provides a baseline for typical predatory

movement and captures general patterns in Greater Amberjack population dynamics. I observed an overall increase in Red Snapper and Greater Amberjack population health (e.g., biomass and weight-at-age) with increasing number of structures. Additionally, distance traveled, forage distance, and number of hours at risk, which are indicators for predation risk, decreased with increasing number of structures. These results may suggest that habitat connectivity plays an important role in predation risk and time at liberty for Red Snapper. I also validated age- or size-based transitions in habitat for Red Snapper that have been observed in other studies. Furthermore, I incorporated value and cost-benefit into my model to assess artificial reef efficacy over differing reef configurations. This model provides resource managers and stakeholders with a tool to guide future reefing projects to discern the impact of location, orientation, and density of artificial reefs on populations of economically important species.

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CHAPTER VI: SUMMARY

In the northern Gulf of Mexico (GOM), artificial reefs have become an important resource enhancement tool (Bohnsack & Sutherland 1985). These structures act as fish aggregation sites on a relatively barren continental shelf region that contains little to no vertical structure for fishes to seek refuge (Dennis & Bright 1988, Rooker et al. 1997, Hobday & Campbell 2009). Generally, a lack of fishery-independent studies comparing artificial reefs to their natural counterparts has led to uncertainty surrounding the role of artificial reefs (Streich et al. 2017). Furthermore, with increasing legislative regulations on the types of materials permitted for artificial reefs, it is important to examine which materials provide the most benefit (Becker et al. 2018). Thus, the goal of this dissertation was to present stakeholders and managers with the tools to understand how artificial reef placement and proximity to other reefs, which are difficult to ascertain, affect population dynamics to guide future reefing projects.

In Chapter 2, I used a combination of remotely operated vehicle and hydroacoustic surveys to characterize fish communities at artificial reefs and natural banks and provide absolute abundance estimates of four economically important fish species: Red Snapper, Greater Amberjack, Gray Triggerfish, and Vermilion Snapper. I identified differing patterns of fish communities between artificial reefs and natural banks and among three depth strata. Furthermore, although artificial reefs were more speciose and contained higher densities, natural banks contributed more to the overall abundance. My results show high abundance of reef fish, particularly Red Snapper, Vermilion Snapper, and Greater Amberjack. Additionally, age 2+ Greater Amberjack extrapolated estimates were 221% of the most recent stock assessment for this species and identify a need for reevaluation of the habitat use and abundance estimates for

this species. My findings improve the scientific understanding of the role of these habitats in fish communities and provide fisheries managers with abundance estimates for stock assessments.

In Chapter 3, I evaluated the efficacy of differing densities and orientations of three nearshore prefabricated pyramid reefs on the Texas continental shelf using stereo-remote underwater video and hydroacoustic surveys. Although reefing location nor orientation had a significant effect on species richness, diversity, and evenness, I identified differing communities between both variables (reefing location and orientation) with Red Snapper, Greater Amberjack and Gray Triggerfish among the main contributing species to these differences. Fish measurements identified primarily juvenile individuals which emphasizes the importance of lower relief reefs as habitat for younger cohorts. Absolute abundance estimates were derived from combining the video surveys and hydroacoustic surveys and were used to calculate two types of cost-benefit ratios based on overall fish density and the densities of economically important species such as Red Snapper, Greater Amberjack, Gray Triggerfish, Vermilion Snapper, and Almaco Jack. Port O'Connor nearshore reef was considered the least cost-effective reefing location in terms of overall fish density, but not for economically important species density. Port O'Connor was configured with science in mind and produced the most consistent results compared to the other reefing locations, which highlights the importance of developing a reef with science in mind to evaluate reef benefits more competently.

In Chapter 4, I examined fish communities at different reefing materials and quantities at Rio Grande Valley reef using a combination of stereo-remote underwater video systems and hydroacoustic surveys. I assessed fish community differences and calculated two different types of cost-benefit ratios based on overall fish density and the densities of economically important species such as Red Snapper, Greater Amberjack, Gray Triggerfish, Vermilion Snapper, and

Almaco Jack. Species richness was found to increase with increasing tonnage of railroad ties and is likely due to differences in structural complexity. Evaluation of the two different types of cost-benefit ratios identified a massive 4000-ton railroad tie pile as the least cost-effective in terms of overall fish density and economically important species density. As a result, I do not recommend considerably large piles of reefing material if the goal is to provide habitat for economically important fish species, rather spread this material in smaller widely dispersed groupings. My findings identify the need for specifying clear objectives when designing artificial reefs to maximize efficiency. Future reefing projects should also consider placing different materials further distances apart while still occupying similar geography, depth, and proximity to shore to properly examine their efficacy.

In Chapter 5, I developed an individual-based model to evaluate Red Snapper, Greater Amberjack, and four competing species, population responses to changing artificial reefs in more complex, but realistic, habitat scenarios like those observed on the Texas continental shelf. The model simulated growth, mortality, and movement over 50 years on a 2-dimensional spatial grid composed of natural, prefabricated pyramid, and rig reefs. Model results indicated an overall increase in Red Snapper and Greater Amberjack biomass and average weight-at-age with increasing number of structures. Additionally, distance traveled, forage distance, and number of hours at risk, which are indicators for predation risk, decreased with increasing number of structures and mirrored patterns in connectivity on the spatial grid. These results suggest that habitat connectivity plays an important role in population stability, mixing, and for other important life stages in life history for species that use artificial and natural habitats. This model also validated age- or size-based transitions in habitat for Red Snapper that have been observed in other studies, suggesting a transition from habitat-centric to forage-centric movement for this

species. Furthermore, I incorporated value and cost-benefit into the model to assess artificial reef efficacy over differing reef configurations. This model presents resource managers and stakeholders with a tool to guide future reefing projects to discern the impact of placement and density of artificial reefs on population dynamics of economically important fish species.

Collectively, my findings suggest that artificial reefs in the western GOM play an important role in shaping fish communities and in the population dynamics of economically important species. Furthermore, I developed a unique approach to quantifying the efficacy of artificial reefs and revealed two types of reefs that were significantly less cost-effective than the others explored in this study. Future artificial reefing projects should avoid large, highly dense reefs and should consider proximity to other artificial or natural habitats. I also present a tool for resource managers and stakeholders to guide future reefing projects to discern the impact of differing artificial reef designs on populations of economically important species. This dissertation improves the scientific understanding of the role of natural and artificial habitat in fish communities and provides fisheries managers with key abundance estimates needed for stock assessment models for several economically important species on the Texas coast.

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BIOGRAPHICAL SKETCH

Kelsey Lynn Martin was born in 1991, in Frederick, Maryland. She graduated from Governor Thomas Johnson High School in 2009 and attended Coastal Carolina University (CCU) where she majored in Marine Science with a minor in Applied Mathematics. Her love for the ocean and her passion for science stem from a combination of spending her summers in the Outer Banks, North Carolina and from her excellent professors at CCU. During her undergraduate, she engaged in several research opportunities, one of which investigated the commensal relationship between the Ghost Crab and a parasitic nematode. She also researched reef fish communities among various benthic habitats in Discovery Bay, Jamaica. Upon graduating Magna cum Laude with her Bachelor of Science in 2012, Kelsey volunteered at the Bimini Biological Field Station Foundation in Bimini, the Bahamas. She then spent a year as a North Pacific Groundfish Observer on the Bering Sea, Alaska before attending graduate school in 2015 to obtain her Master of Science under Dr. Dan Abel at CCU. Kelsey's thesis research involved investigating the habitat use of Blacktip Sharks at fishing piers in Myrtle Beach, SC. Immediately following obtaining her master's degree in December 2017, Kelsey moved to Texas to begin working on her doctorate at the Fisheries and Ocean Health Laboratory under Dr. Greg Stunz at the Harte Research Institute for Gulf of Mexico Studies (HRI) at Texas A&M University-Corpus Christi (TAMUCC). Kelsey is also a National Oceanic and Atmospheric Administration Center for Coastal and Marine Ecosystems Scholar and has presented research at their 10th Biennial conference in which she won an award for best oral presentation in the category of Healthy Oceans. Kelsey will earn a Doctor of Philosophy degree in Marine Biology from TAMUCC in August 2022. Following graduation, she will begin a position as a post-doctoral associate at the Center for Sportfish Science and Conservation at HRI.