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## **Fish Biomass and Community Structure Around Standing and Toppled Oil and Gas Platforms in the Northern Gulf of Mexico Using Hydroacoustic and Video Surveys**

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FISH BIOMASS AND COMMUNITY STRUCTURE  
AROUND STANDING AND TOPPLED OIL AND GAS PLATFORMS  
IN THE NORTHERN GULF OF MEXICO  
USING HYDROACOUSTIC AND VIDEO SURVEYS

A Thesis

Submitted to the Graduate Faculty of the  
Louisiana State University and  
Agricultural and Mechanical College  
in partial fulfillment of the  
requirements for the degree of  
Master of Science

in

The Department of Oceanography and Coastal Sciences

by  
Emily Marissa Reynolds  
B.S., Emory University, 2012  
August 2015



To my parents, thank you for all of your support and guidance throughout everything.  
I would not be where I am today without you.

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## ABSTRACT

There has been relatively little study of the efficacy of decommissioned oil and gas platforms as artificial reef habitats for fish assemblages in the northern Gulf of Mexico (GOM). A variety of fish species have been reported on these structures, but the species biomass distribution and community structure has not been studied thoroughly. Hydroacoustic and video surveys were conducted quarterly from June 2013 to June 2014 at three standing and two toppled oil and gas platforms located approximately 130 km off the coast of Louisiana at 90 m depth; to gain information about the spatial biomass distribution and community structure at standing and toppled platforms. The toppled platforms in our study became a part of the Louisiana Artificial Reef Program in 2002. Stereo and Go-Pro® cameras were utilized for video surveys, allowing us to record counts and lengths of fishes. Hydroacoustic surveys were used to define the spatial distribution of fish biomass (MVBS,  $S_v$ ) in relation to distance to 500 m from the center of each site and the depth distribution of the observed biomass. MVBS of fishes was highest when near the structure and declined rapidly as distance from the structure increased, leveling off to a background-noise level at 100 m from the structure. Additionally, fish MVBS was highest in the lower water column (>60 m) compared to the upper and middle water columns. The fish communities differed between depth layers in the water column, seasonally, and between structure types (standing/toppled), with red snapper (*Lutjanus campechanus*) as a dominant species present at both site types. Coupling the data from these methods revealed that even though fish MVBS remained consistent throughout all seasons, different species comprise the MVBS observed during different seasons and between different layers in the water column.

Coupling non-destructive methods allows for rapid monitoring which can be helpful in determining how to effectively manage both the fishes around the structure and the structures themselves.

## CHAPTER 1: GENERAL INTRODUCTION

The northern Gulf of Mexico (GOM) encompasses a range of complex and unique habitats that provide shelter and resources to a variety of fish communities; as well as being home to the world's largest de-facto artificial reef complex (Dauterive 2000). The northern continental shelf of the GOM was once thought to be primarily soft bottom with little to no vertical relief (Ludwick 1964, Kennicutt II et al. 1995, Wells 2007). However, natural hard bottom banks are common at the edge of the continental shelf (Bright 1977, Parker et al. 1983).

Natural banks and artificial reefs in the GOM provide different types of hard bottom habitat and vertical relief for fishes. Natural banks provide shallow slopes, whereas artificial reefs (ex. toppled and standing oil and gas platforms) provide steeper vertical relief (Figure 1.1). The plan and side views of these structures have different footprints, creating varied relief in the water column. Due to varied vertical relief and footprints, ecologically and economically valuable fishes utilize both the natural and artificial habitats; therefore it is important to manage these resources.

The definition of an “artificial reef” varies from state to state in the GOM, and materials designated for building such structures may include commonplace items such as concrete, rock, stone, FADs (Fish Aggregating Devices), offshore platforms, plastic, shipwrecks, steel, automobiles, and rope (Baine 2001). Artificial hard substrates in Louisiana largely consist of standing and toppled oil and gas platforms.

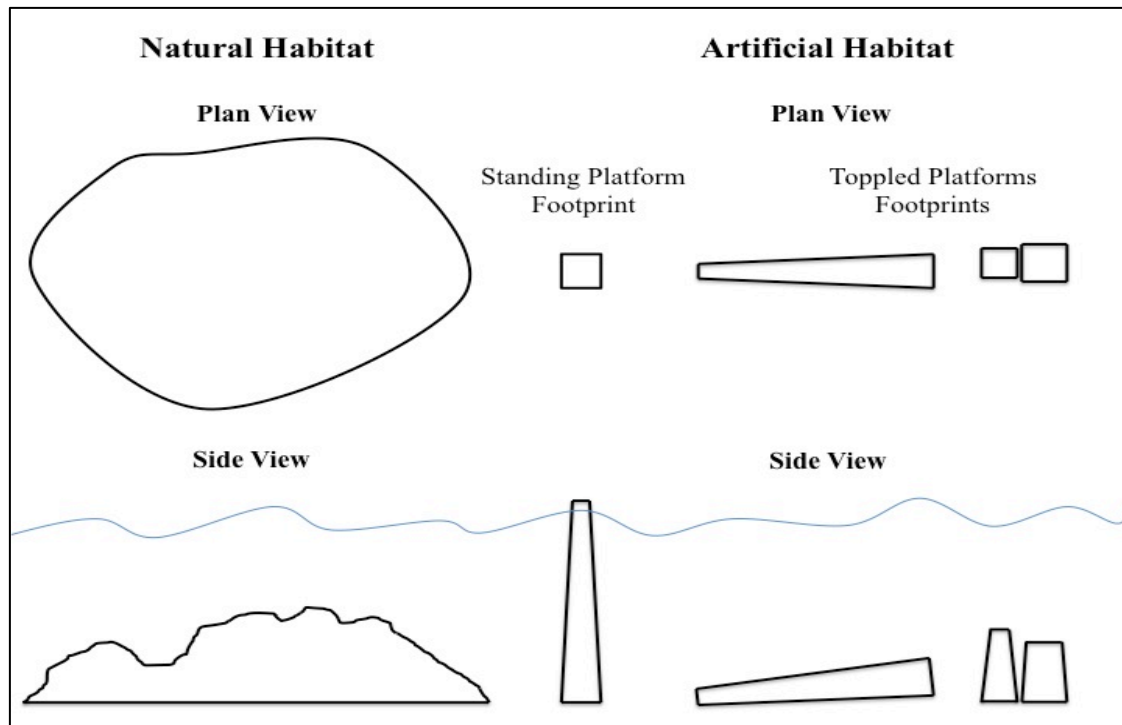


Figure 1.1. Schematic of vertical and horizontal footprint and relief comparing natural and artificial reefs (reproduced from Glenn 2014). These figures are not to scale.

The first oil platform in the GOM was installed in 1942 on the Outer Continental Shelf (OCS) (Pulsipher et al. 2001, Westmeyer et al. 2007). Natural hard substrate in the northwestern and north central GOM OCS region is limited and it is estimated that offshore platforms contribute additional “reef” habitat. Since the 1940s, the oil industry in the GOM has been estimated to add between 0.4-28% of hard artificial structure in the form of platforms (Gallaway and Lewbel 1982, Scarborough Bull and Kendall Jr 1994, Stanley and Wilson 1996, Gallaway et al. 1998, Scarborough-Bull et al. 2008).

Over 5,500 platforms have been constructed in the GOM since 1942, and over 4,000 have since been decommissioned and/or removed (Dauterive 2000, Simonsen 2013, Cowan Jr., J.H.<sup>1</sup> personal comm). Most platforms are located on the outer

<sup>1</sup> Cowan, Jr., J.H. 2014. Louisiana State University, Department of Oceanography and Coastal Sciences.

continental shelf in the north central and northwestern GOM off of Texas, Louisiana, and Mississippi, however many of the platforms off the coast of Louisiana have since been removed in the past year (Figure 1.2). These platforms provide hard substrate in a mostly soft bottom habitat, providing habitat for species that rely on these surfaces for shelter and as prey sources.

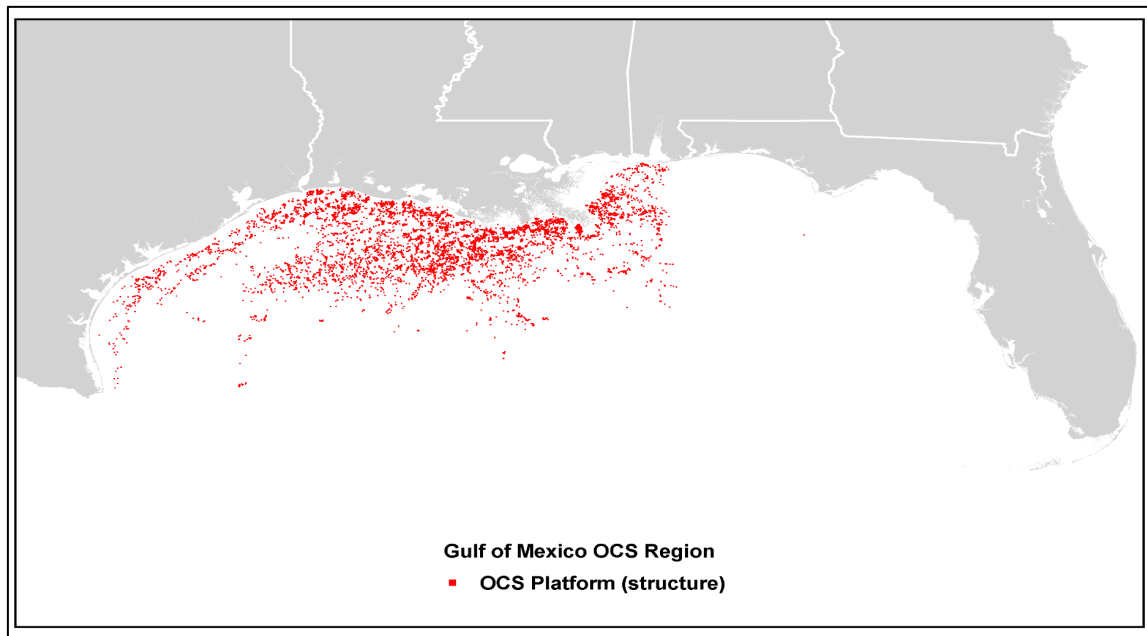


Figure 1.2. Map showing current standing oil and gas platforms as of 2015, off the coast of Louisiana, Texas, Alabama, and Mississippi (adapted from BOEM 2015).

Reefed and standing platforms are of interest to many stakeholders because they tend to aggregate and attract various species of reef-associated fishes. Productivity of an artificial reef relies on the assumption that they provide additional important habitat that increases the environmental carrying capacity of organisms associated with reefs; thus, resulting in a net positive affect on the abundance and biomasses of fishes in those ecosystems (Bortone et al. 1994, Polovina 1994, Pickering and Whitmarsh 1997).

Artificial reefs have been noted to provide various ecosystem services such as, substrata



for fouling organisms and benthic fauna, increased prey sources and feeding efficiency, shelter from currents and predation (Spanier 1997) and recruitment habitat, in addition to acting as aggregating devices (Bohnsack 1989). Since these structures have the potential to attract/aggregate demersal and pelagic organisms, they make an easy target for fishermen and larger predators (Pickering and Whitmarsh 1997). Studies have shown fish abundance to be 3-25 times higher within 16 m of the structure in the GOM and are frequented by both recreational and commercial fishers, in addition to SCUBA divers and spear fishers (Dauterive 2000, Stanley and Wilson 2003, Wilson et al. 2006, Keenan et al. 2007, Galloway et al. 2009, Harwell 2013). A typical eight-leg structure “provides a home for 12,000 to 14,000 fish” according to a study by Stanley and Wilson (2000). Studies have attempted to prove or disprove that platforms in the GOM produce contribute to primary production. The debate has been labeled as pointless and difficult because of the inability to determine these factors for all species associated with the reefs (Shipp 1999, Shipp and Bortone 2009, Cowan et al. 2011).

Artificial reefs may be essential to U.S. fisheries because of the habitat they provide for economically important fishes. These structures are prominent in all five coastal states in the GOM, which have approved artificial reef plans. However, Louisiana and Texas currently incorporate decommissioned platforms into their artificial reef programs. After oil and gas platforms retire, they are required to be decommissioned and removed, which in turn removes their capability for providing additional vertical and structural fish habitat. The artificial reef programs help to manage these structures, and the deployment of structures has become a common strategy for the management of fisheries and habitat rehabilitation worldwide (Brickhill et al. 2005, Fowler and Booth

2012). States in the US have set guidelines on the materials allowed for use as artificial reefs, as well as monitoring these structures and their placement. These programs are important because these structures appear to serve as principal and intricate habitat for fishes and reefs.

The Louisiana Artificial Reef Program (LARP) was established in 1986 as a program under the Louisiana Department of Wildlife and Fisheries (LDWF) to take advantage of decommissioned platforms that provided important habitat for many of Louisiana's coastal fishes (Wilson et al. 1988). LARP relies almost entirely on using the material from decommissioned oil and gas platforms to create artificial reefs and this process has been coined as "Rigs-to-Reef" (RTR) (Kaiser 2006). LARP created nine planning areas (PA) on the Louisiana continental shelf in which decommissioned oil and gas platforms are placed (Wilson et al. 1988) (Figure 1.3).

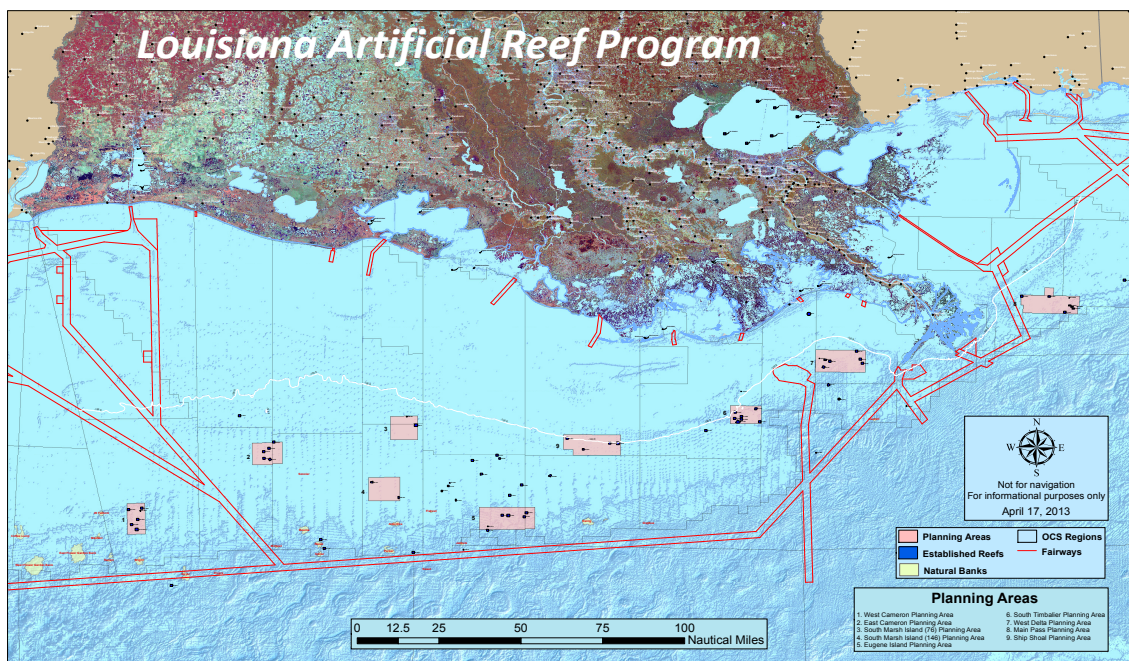


Figure 1.3. Map of nine planning areas (pink) for artificial reef placement off the coast of Louisiana through the Louisiana Artificial Reef Program (LARP). The established reefs are shown in blue and natural banks in yellow (image from LARP 2014).

The selection for these nine PAs included evaluation of available scientific information (side-scan sonar surveys) in conjunction with the consultation of participants from the offshore, penaeid shrimp fishery, affiliated oil and gas industries, and with federal and state agencies (Stephan et al. 1990). Typically, platforms are converted to artificial reefs by one of three methods: laying the platform on its side, cutting the platform at a specific depth and placing the remaining piece next to it, or by toppling the platform then moving it to a new location (Figure 1.4) (Dauterive 2000, Wilson et al. 2006). Since the mapping of the nine PAs in 2007, 71 artificial offshore reefs utilizing the jackets of 320 obsolete platforms have been created off the coast of Louisiana (LARP 2014). Conservation organizations along with both recreational and commercial fishers are supportive of this process (Dauterive 2000, Wilson et al. 2006).

Using platforms as artificial reef habitats has many benefits: their large numbers, design, longevity, and stability provide advantages over the use of traditional artificial reef materials such as concrete and plastics (Baine 2001, LARP 2014). Additionally, there is a positive relationship between the oil/gas companies and the artificial reef program, which encourages the reefing of these platforms, rather than an expensive full removal. The companies donate one half of their realized savings over a traditional onshore removal into the LARP Trust Fund, along with the physical platform material (LARP 2014). For example, if decommissioning a structure costs \$800,000 to remove, transport and leave on shore, and reefing the structure would reduce costs to \$400,000, the company would donate \$200,000 to the state program to assist with the management of the artificial reef program (Bureau of Safety and Environmental Management 2015).

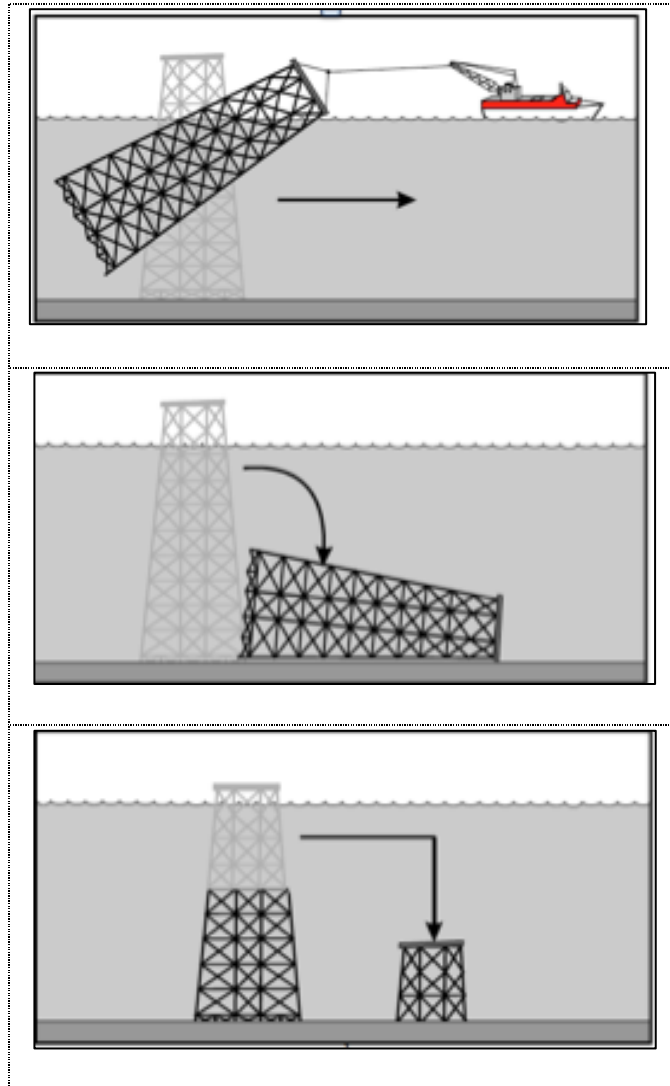


Figure 1.4. Diagram of three decommissioning methods for turning standing platforms into toppled platforms (reproduced from Dauterive, 2000).

According to the Bureau of Safety and Environmental Enforcement (BSEE), approximately 450 platforms have been converted to permanent artificial reefs in the GOM since December 2013. Even though there are many benefits of reefing platforms, the logistics involved in determining where such large structures can be safely placed can be challenging. According to LARP the waters that are deep enough ( $>30$  m) to place toppled platforms in accordance with Coast Guard regulations, are typically 30-70 miles off the coast of Louisiana (Wilson et al. 2006).

Surprisingly, there has been relatively little study of the efficacy of standing and toppled structures as fish habitat although most of the information known about reef-associated fishes in the GOM is based upon fishes collected on platforms or other artificial reefs (SEDAR 2013). Informed decisions about the use of artificial reefs as fishery management tools requires quantitative data on spatial and temporal variation in fish biomass, abundance, and community composition (Grossman et al. 1997, McCawley and Cowan 2007, Boswell et al. 2010).

Determining the community structure and biomass of fishes associated with artificial reefs as large as oil and gas platforms is difficult due to the size of the structures, the distance offshore, and the relatively high diversity of fishes found in the GOM. Furthermore, fishery independent data are often difficult to collect, especially at toppled platforms due to the depth of water in which the structures are placed. Toppled structures are not always visible to the naked eye depending on depths, increasing sampling difficulties, as the structure may have shifted due to weather and erosion (Stanley and Wilson 1996, Boswell et al. 2010).

To manage fisheries responsibly, it is important to take into account the life histories of the target species. For example, most fish species spend their juvenile life in a habitat different from that used during their adult life. Red snapper (*Lutjanus campechanus*) are known to be an abundant reef-associated fish around platforms in the northern GOM. They tend to spend their juvenile lives closer inshore on sandy or muddy bottoms, then move offshore as an adult, favoring hard reef and reef-like habitats (Stanley and Wilson 1990, Goodyear 1995, Render 1995, Nieland and Wilson 2003, Westmeyer et al. 2007). Red snapper have been found to collectively comprise more

than 95% of the total fishes at some platforms (Stanley and Wilson 2000, Scarborough-Bull et al. 2008). Additionally, studies conducted over several decades consistently show high levels of red snapper abundance around these structures, with large schools of sub-adult and adult fish comprising between 37 and 80% of the total fish abundance (Continental Shelf Associates 1982, Render 1995, Nieland and Wilson 2003, Rademacher and Render 2003, Westmeyer et al. 2007). The time of residence of red snappers is relatively short because they move off of more structure habitat after they are about 10 years old, although they can live to be >50 years old. Even though there are estimates of red snapper abundance on platforms, little is known about what role they play in red snapper ecology and life history. The decommissioning of platforms may have an effect on red snapper along with many other reef-associated species, however the long-term effects are unknown.

To determine the efficacy of these structures in relation to fish biomass and community structure, baseline data along with effective sampling methods are necessary. Baseline data can provide ecological information that is valuable to both resource managers and fishery scientists. Due to natural and human-caused disasters, such as the Deep Water Horizon oil spill, baseline data is important to create a “before” picture in case a disaster occurs. Baseline data can then be used to understand what affects the disaster has had on the current area. Therefore, it is important to the LARP that a cost-effective and relatively simple method is developed that can be used to first collect baseline data, then to monitor the community structure and biomass distribution around platforms in the PAs, both temporally and spatially (LARP 2014).

Unbiased and accurate surveys are needed to better understand the reef-associated fish distributions around these abundant structures. As such, I have combined two non-destructive survey techniques of video and hydroacoustic (acoustic) to assess biomass and community structure that provides efficient, reliable data that have been collected at three standing and two toppled platforms chosen for this study. Video data (stereo-video and Go-Pro<sup>®</sup>) provided me with size, number, and species identification of fishes. Acoustic surveys were used to estimate the spatial biomasses and distributions of fish stocks (Simmonds and MacLennan 2005, Lundgren and Nielsen 2008). I believe that the combination of these two methodologies provide insight on what species (via video sampling) are contributing to the acoustic biomass observed during surveys.

Several studies have attempted to combine both video and acoustic data or used a combination of one of these methods along with other various methods (such as trawling). For example, a study conducted by Stanley and Wilson (2000) used SCUBA visual surveys and quantitative dual beam acoustic surveys in the northern GOM to document the communities of fish associated with standing petroleum platforms. The combination of these techniques allowed for the measurement of the area around the artificial reefs paired with estimates of the abundance and species composition throughout the water column (Stanley and Wilson 2000). Additionally, a study conducted by Langlois et al. (2012) used a stereo-baited remote underwater video (BRUV) array and the results provided accurate estimates of length-frequency distributions for the fish sampled when compared to length-frequency distributions derived from fishery-independent line fishing (Langlois et al. 2012). More recently, studies conducted by Boswell et al. (2010) and Simonsen (2013) were able to estimate

the spatial distribution of fish biomass and density associated with standing and or toppled platforms in the GOM. However, these were solely acoustic estimates. If there was complementing video data to ground truth what species were contributing to the acoustic biomass observed, additional information could have been collected regarding fish community structure patterns.

Because combining video and acoustic sampling techniques has been successful in previous studies, I used similar methodology; video and acoustic surveys to determine biomass and community structure at standing and toppled platforms approximately 100 miles off the coast of Louisiana. Simonsen (2013) acoustically sampled four of the five study sites during a previous study from 2008-2010. This study is sampling these sites due to familiarity and associated baseline data. Acoustic methodology was improved by shortening the distance in which acoustic data was collected, allowing for more efficient data collection and a shorter survey time.

Chapter 2 describes the spatial biomass distribution of fishes around standing and toppled oil and gas platforms through acoustic surveys. Additionally, determining whether the spatial biomass distribution of fishes differs between types of structure (standing and toppled) in relation to season, depth of the water column, and distance away from structure was addressed. Swim-bladdered fish biomass was compared to nekton biomass to determine if differences existed in relation to structure, depth, and distance away from the structure. Acoustic data collected were three years after Simonsen (2013) and were compared to identify whether differences occurred in relation to biomass around the structure (depth and distance away). Chapter 3 describes possible differences in fish community structure at both standing and toppled oil and gas



platforms through video surveys, in relation to season and depth of the water column. Published length-weight relationships were derived using length data obtained from the video surveys and used to apportion the observed acoustic biomass (MVBS) based upon the number and size of the fishes observed in the videos. These data were also used to ground truth the acoustic biomass described in Chapter 2. Chapter 4 explains the implications and uses for fishery monitoring and management from using acoustic and video surveys for data collection in addition to addressing whether or not rapid assessments can be used throughout the GOM using these non-invasive survey methods.

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## CHAPTER 2: SPATIAL BIOMASS DISTRIBUTION OF FISH AROUND STANDING AND TOPPLED OIL AND GAS PLATFORMS IN THE NORTHERN GULF OF MEXICO

### **Introduction**

The discovery of extensive oil deposits in the northern Gulf of Mexico (GOM) resulted in large-scale oil and gas production, leading to the creation of a considerable amount of artificial reef structures. Beginning in the 1940s, the oil industry in the GOM added anywhere from 0.4-28% of additional hard artificial structure in the form of platforms (Stanley and Wilson 1996, Gallaway et al. 1998, Scarborough-Bull et al. 2008). Once platforms are no longer useful, they are decommissioned in accordance with federal law. There are many challenges associated with sampling and monitoring these structures due to their size and distance from shore (Fowler and Booth 2012, LARP 2014). To better understand how these structures can be used as fishery management tools, data collected must provide quantitative descriptions of spatial and temporal variations in fish biomass, abundance, and community composition (Grossman et al. 1997, Boswell et al. 2010). Collecting fishery independent data using hydroacoustic (acoustic) surveys is a well-established method to estimate the spatial biomass distribution around these complex structures.

The use of underwater sound dates back to the 1500s, when people listened for sounds of distant ships using underwater tubes. Later, echograms were used to locate fish schools in the 1930s, however quick advancements in sonar technology during World War II led to the first quantitative estimates of fish abundances in the 1950s. As the technology rapidly advanced, echo integration methods involved deployment of split-, dual-, and multi-beam devices (Simmonds and MacLennan 2005). Acoustic instruments

that transmit and receive sound waves are used to detect fishes and other objects beyond the human range of vision and have had a major impact on fishing, increasing fishing power significantly because of the ability to locate concentrations of target species (Simmonds and MacLennan 2005).

Acoustics are a non-intrusive and non-destructive survey tool that use high-frequency sound to locate fish and marine organisms in the water column. They are typically conducted sampling along transects in a specific pattern while recording the echoes of organisms detected by the echo sounders. They can be used to detect fish and marine organisms across all habitat types; however detectability is limited near the seafloor (Clark et al. 2014).

Acoustics are used in fisheries research to estimate the spatial biomass distribution of fish stocks (Simmonds and MacLennan 2005, Lundgren and Nielsen 2008). The unit of measurement of spatial biomass is typically expressed as mean volume backscatter strength (MVBS;  $S_v$ ), a proxy for biomass, and is the combined echo intensity from all scatterers, averaged over a given volume. It is understood to be proportional to a measure of biomass in the water column and is measured in decibels (dB) (MacLennan et al. 2002).

Previous studies have attempted to better determine and understand the spatial distribution of fish biomass associated with artificial reefs including: the North Sea (Løkkeborg et al. 2002), Mediterranean Sea (Fabi and Sala 2002), Australia (Pradella et al. 2014), and the GOM (Stanley and Wilson 1996, Boswell et al. 2010, Simonsen 2013). Results of these indicate that abundance, biomass, density, and catch rates exponentially decline with increasing distance from the reefs.



As rapidly as acoustic surveys and techniques have evolved, they are likely to continue serving as a necessary non-destructive method of studying and quantifying fishes and other marine organisms. Acoustics can be mounted on structures such as oil and gas platforms (stationary) (Stanley and Wilson 1996, 1997, 1998, 2000) or they can be towed around structures (mobile) (Gerlotto et al. 1989, Boswell et al. 2010). Not only can acoustic devices quantify abundance of fish present, they can also be used in evolutionary and behavioral studies (Pitcher et al. 1996, Nottestad and Axelsen 1999, Nottestad et al. 2002, Simmonds and MacLennan 2005).

Acoustic surveys provide solutions some of the problems that other non-destructive survey methodology may encounter. For example, when visual surveys are used to assess fish populations around platforms, limited visibility, diver avoidance, and gear bias can lead to unstandardized data which makes it difficult to compare results across multiple surveys and studies (Stanley and Wilson 2000). Acoustics can be used to sample large areas at various depths because sound can propagate over hundreds of meters. In addition, they can operate over a broad frequency range and offer a high sampling rate (Simmonds and MacLennan 2005, Lundgren and Nielsen 2008, Ressler et al. 2015).

Even though there are advantages of using acoustics, there are also factors that limit usefulness, especially in places where species diversity is high. There is little information about species-specific target strength (TS), thus hindering, species identification, making it difficult to infer patterns of species distribution (Clark et al. 2014). Additionally, sound can be attenuated by various physical factors such as

turbidity, salinity and density gradients that may hinder the data collected at great depths or in turbid environments (Simmonds and MacLennan 2005).

Due to the continued success of acoustic data collection, fish distribution, density, and biomass at artificial habitats can be quantified (Barans et al. 1997). These surveys are especially useful on artificial reefs that are either too large or too deep for visual assessments (Boswell et al. 2010). A more recent study conducted by Simonson (2013) used acoustic transects to determine spatial and temporal patterns of fish biomass distribution associated with standing and toppled oil and gas platforms in the northwestern GOM.

The goal of my study was to use acoustic surveys to determine the spatial biomass distribution of fishes at standing and toppled oil and gas platforms in relation to season, depth bins (layers) in the water column, and their distance away from the structure. In addition, spatial biomass distributions around these structures (standing/toppled) determined from the acoustic surveys will be compared to better understand the value of these artificial habitats as a management tool. I hypothesized that 1) acoustic biomass at standing platforms would be greater than at toppled platforms, and 2) acoustic biomass would be higher near structure and decrease as distance increased from structure, regardless of structure type.

## **Methods**

### **Study Area**

Five oil and gas platforms in the Eugene Island (EI) Oil and Gas Lease Block were sampled between June 2013 and June 2014. The sites consisted of three standing and two toppled oil and gas platforms located approximately 130 km off the coast of Louisiana (Figure 2.1). Standing platforms are located in blocks EI 346, EI 325, and EI

320 and toppled platforms are in blocks EI 322 and EI 324. The standing platforms, EI 346A, EI 325A, and EI 320B, are operational platforms that began production in 1989, 2000, and 2006 respectively.

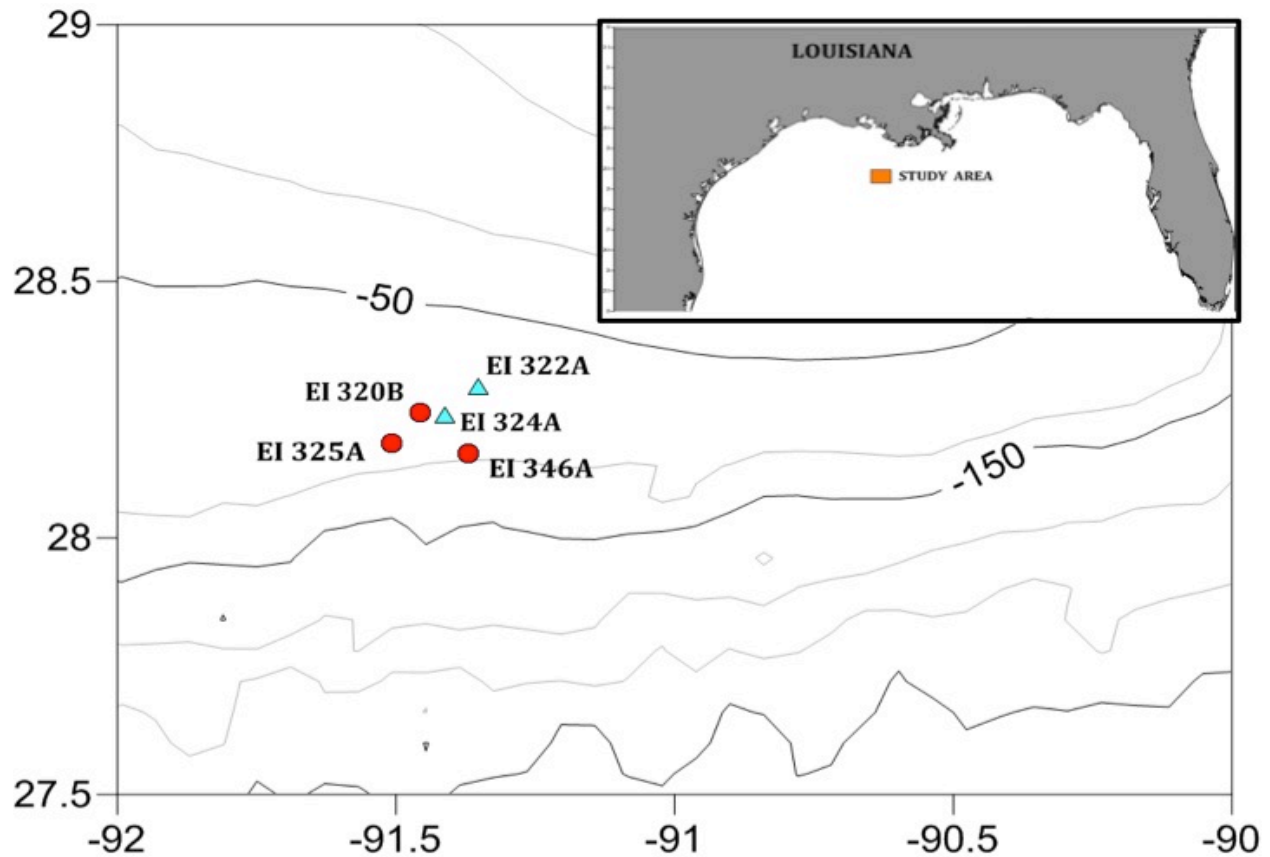


Figure 2.1. Map of study site approximately 130 km off the coast of Louisiana. Five study sites shown: two toppled platform sites (blue triangles); three standing platform sites (red circles).

Hurricane Lili damaged two standing platforms beyond repair in 2002, which led them to be decommissioned and later toppled. They were donated for biological study by British Petroleum (EI 322A) and Newfield Exploration Company (EI 324A) to the Louisiana Artificial Reef Program (LARP) (Kaiser and Kasprzak 2008). The three standing platforms are located in ~ 85 m depth and the two toppled platforms are located in ~ 70 m (Table 2.1). Study sites were chosen to facilitate direct comparisons with a

Table 2.1. Physical geography of the three standing and two toppled oil and gas platforms in the Eugene Island (EI) Oil and Gas Lease Block. Values obtained from the Bureau of Safety and Environmental Enforcement (BSEE 2014) and the Louisiana Artificial Reef Program (LARP) (Mike McDonough<sup>2</sup> personal comm.).

| SITE  | COMPANY                                    | LATITUDE      | LONGITUDE      | DEPTH | INSTALLED | TOPPLED | PILES  |
|---|--|---------------|----------------|-------|-----------|---------|--------|
| EI 346<br>“A”<br>(standing)                       | Fieldwood<br>Energy                        | 28° 9'49.97”N | -91°22'8.19”W  | 95m   | 2000      | n/a     | 4-pile |
| EI 325 “A”<br>(standing)                          | Fieldwood<br>Energy                        | 28°14'39.00”N | -91°27'26.00”W | 76m   | 1989      | n/a     | 4-pile |
| EI 320 “B”<br>(standing)                          | Arena Offshore                             | 28°15'3.28”N  | -91°26'2.60”W  | 76m   | 2006      | n/a     | 3-pile |
| EI 324 “A”<br>(toppled)                           | Newfield<br>Exploration-<br>currently LARP | 28°14'8.23”N  | -91°24'40.50”W | 80m   | 1990      | 2002    | 4-pile |
| EI 322 “A-<br>prod” and<br>“A-drill”<br>(toppled) | British<br>Petroleum-<br>currently LARP    | 28°17'22.34”N | -91°21'5.50”W  | 70m   | 1978      | 2002    | 4-pile |

<sup>2</sup> McDonough, M. 2014. Louisiana Artificial Reef Program, Louisiana Department of Wildlife and Fisheries.

previous study that sampled EI 325A, 346A, 322A, and 324A in 2009 and 2010 (Simonsen 2013). Data from Simonsen (2013), collected in 2008 to 2010, provide an opportunity for me to determine the range of variability in time and space of fish biomass around these structures.

### **Sampling Method**

Acoustic, video, and CTD surveys were conducted quarterly at each site from June 2013 through June 2014, as weather and scheduling permitted, to examine seasonal and habitat effects on fish biomass. Sites were chosen to replicate the size of structures to the degree possible (Table 2.1). All study sites are artificial reefs, however they will henceforth be identified as either standing or toppled.

Environmental data were collected with a Sea-Bird SBE 25 Sealogger CTD to measure temperature and salinity relative to depth at each site. These parameters are needed for sound speed correction during acoustic analysis and are known to influence fish distribution. The CTD was acclimated at the sea surface for three minutes before descent, and was deployed vertically at a steady rate of approximately 1 m/s. Both descent and ascent were used for verification.

Acoustic surveys collected acoustic backscatter with three downward-facing BioSonics (BioSonics, Inc.) split-beam transducers by using three different frequencies (70 kHz, 123 kHz, and 206 kHz). Transducers were calibrated with the standard sphere method periodically (Foote et al. 1987). Surveys were conducted at  $\hat{X} = 3 \text{ m s}^{-1}$  (5 knots), with each transect measuring 1 km.

Transects were completed at each site in pattern that resembled a ten-lobed tract, focused on the center of the platform with each line extending 500 m on either side of the

geographical center point (1 km diameter); lines were offset 18° from the previous line (Figure 2.2). All surveys were conducted during daylight to reduce the effects of crepuscular movements of fishes at the sites I sampled. Active pinging was recorded during the main line surveys and passive listening was recorded between lines at turns to obtain a baseline background noise level of the ship and water column.

The vessel's location was tracked by a wide area augmentation global positioning system (GPS). The transducers were mounted on a pivoting arm that was lowered when collecting data. It extended 2 m below the surface to avoid additional backscatter that is caused by bubbles created by the ship's wake.

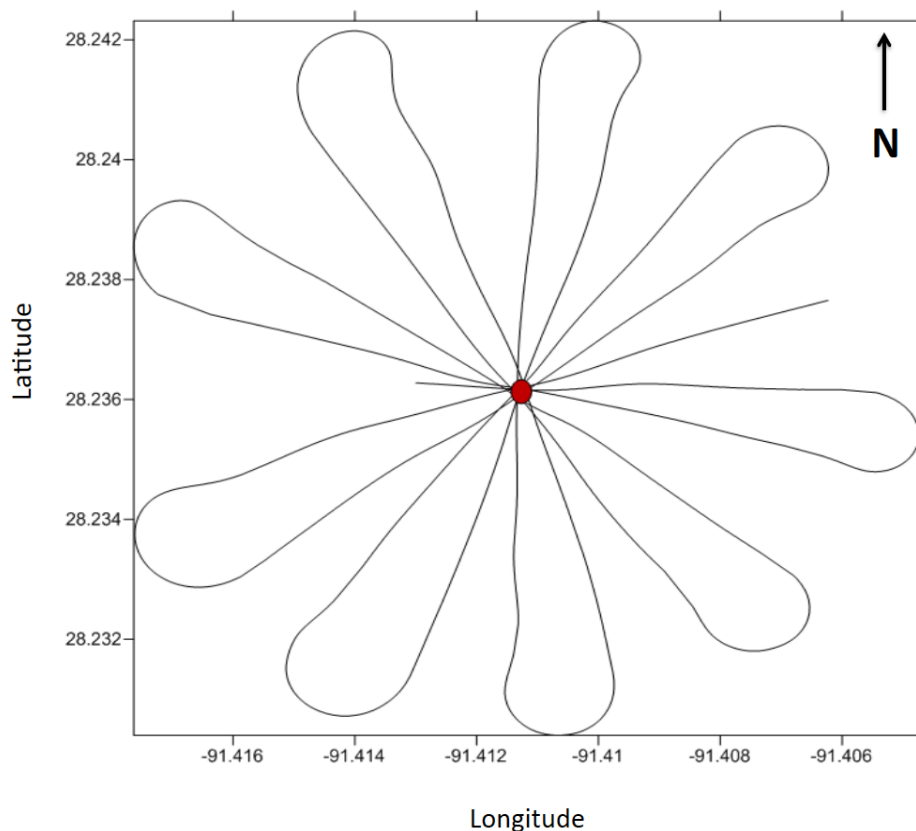


Figure 2.2. Representation of approximate GPS track in the petal hydroacoustic survey pattern around the study site; geographic center of structure represented by the red dot.

## Data Processing

Acoustics backscatter data were catalogued and post-processed in Echoview 5.3 (Myriax Pty. Ltd., Hobart, Tasmania, Australia) to obtain values of MVBS ( $S_v$ , dB). The echogram files were calibrated using data collected with the CTD to correct the speed of sound for the effects of temperature and salinity. Echograms were visually inspected to eliminate regions that I considered misleading due to factors such as excess noise, surface bubbles, loss of signal, and structure. Data within 5 m of the transducers' face were excluded to avoid surface noise. An algorithm ("best bottom candidate" bottom detection) was applied to exclude data within 1 m of the seafloor and reef structures, which were manually edited when necessary. Background noise was removed following the methods of De Robertis and Higginbottom (2007). Stochastic noise spikes that intermittently appeared in the data were suppressed using an algorithm following Anderson et al. (2005). Grids were applied to each echogram, as 10 m depth by 20 m distance cells. The 70 kHz data were exported after the volume backscatter coefficient ( $S_v$ ) was integrated over each cell (10 m x 20 m) to acquire MVBS, to analyze the spatial distribution of the fishes around the platforms (MacIennan et al. 2002), using the relationship:

$$S_v = 10\log_{10}(s_v)$$

where  $S_v$  is the mean volume backscatter (decibels) and  $s_v$  is the volume backscattering coefficient ( $\text{m}^{-1}$ ), represented by the relationship:

$$s_v = \sum \sigma_{bs} / V$$

where  $V$  is the volume occupied by a scattering medium or multiple discrete targets.  $\sigma_{bs}$  is the variance of the backscattering cross-section represented by the relationship:

$$\sigma_{bs} = [r^2 I_{bs}(r) 10^{ar/10} / I_{inc}]$$

where  $r$  is the distance of measurement position from a small target,  $I_{bs}(r)$  is the intensity of the backscattered wave,  $a$  is the area of a school echo-trace observed on an echogram, and  $I_{inc}$  is the intensity of the transmitted or incident wave at the target (MacIennan et al. 2002).

Decibel (dB) differencing was used to isolate swim-bladdered fishes within the echogram. Because different types of organisms respond differently to various frequencies, selecting for specific ranges of  $S_v$  in the difference between two frequencies allows for the differentiation between swim-bladdered fishes and other nekton in the water column. The methods for frequency differencing described in Maduerira et al. (1993), Korneilussen and Ona (2002), Mosteiro et al. (2004), Korneilussen et al. (2009), De Robertis et al. (2010) were followed.

First,  $S_v$  ping times from both the 120 kHz and 70 kHz transducers were synchronized. Differencing of frequencies followed, by subtracting each sample at 70 kHz from the corresponding sample at 120 kHz. Data were organized into two different frequency response ranges: sample  $S_v_{120-70}$  ranging from -15 to 1 db, is labeled as swim-bladdered fishes (fish) and  $S_v_{120-70}$  ranging from 3 to 25 db, is labeled as organisms lacking swim bladders (Table 2.2).



Table 2.2. Classification rules applied to acoustic backscatter based on frequencies.

| Classification | $\Delta db_{i,j}$       | Description   |
|----------------|-------------------------|---|
|                | <u>120-70 kHz</u>       |   |
| Fish           | $-15 < S_{V120-70} < 1$ | Decreasing values of Sv with increasing frequency                 |
| Zooplankton    | $3 < S_{V120-70} < 25$  | Frequency independent at lower frequencies with increasing values |

Data were returned to their original resolution, exported as MVBS, and later used in models to spatially view biomass around the study sites (Figure 2.3). The two cleaned data sets that were produced as a result included 1) data derived from the 70 kHz echogram, hereafter nekton, and 2) data from the dB differencing echogram, hereafter fish.

The nekton data set included all “noise” making organisms in the water column, such as jellyfish, copepods, zooplankton, shrimp, and fishes. The fish data set is considered a subset of the nekton data set, as it selects for a specific range of frequencies that are most likely specific to “swim-bladdered” fishes.

### Data Analysis

I compared the effects on the distribution of biomass of fish of variables including distance away from structure, layers in the water column, type of site, and season. The geographic center point of each site was determined and distance bins were created based on that center data point in SAS v 9.4 (Cary, North Carolina, USA).

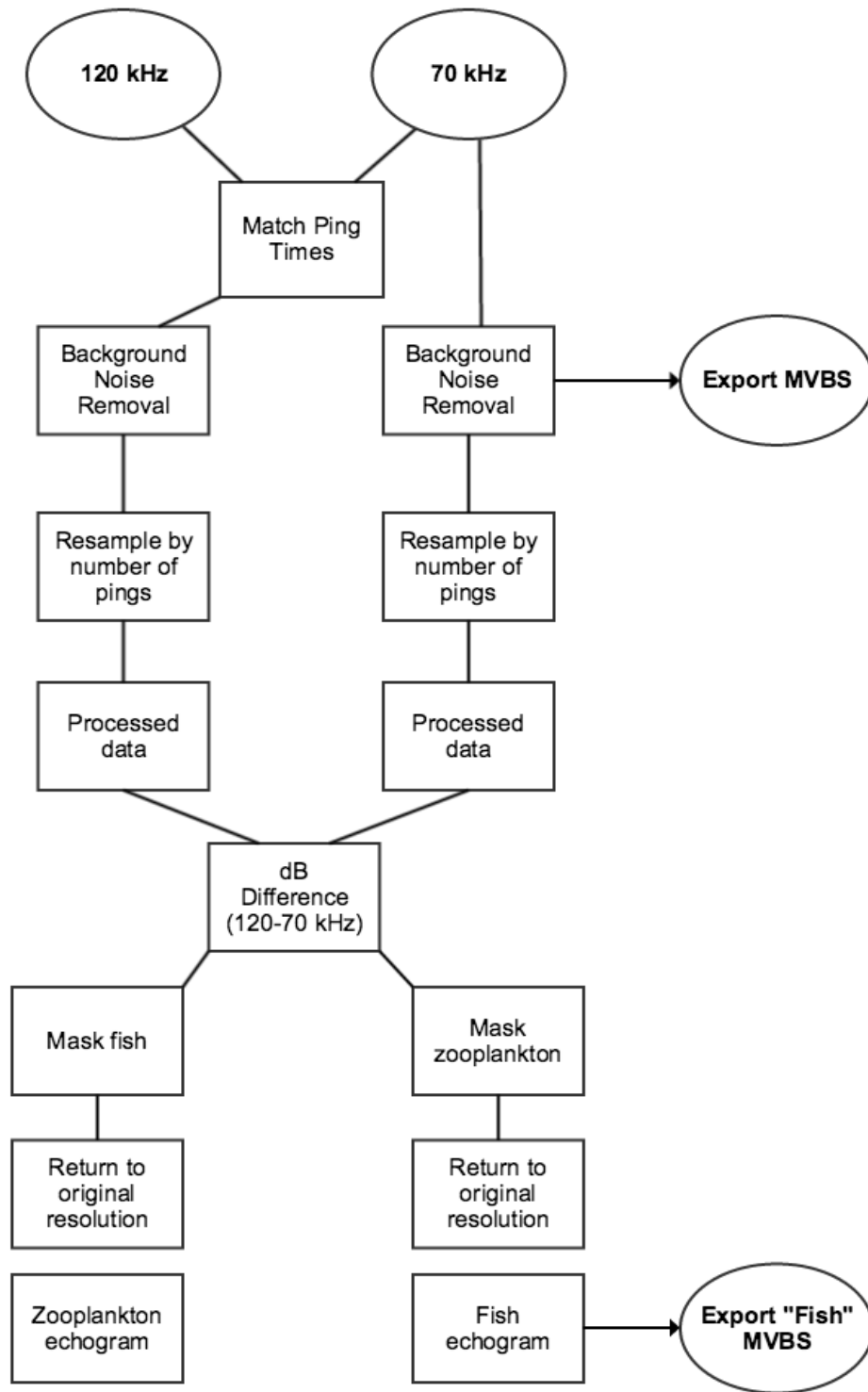


Figure 2.3. Diagram of Echoview Data Flow used for processing acoustic data for decibel (dB) differencing (120-70kHz, fish Sv, dB) and mean-volume backscattering strength (MVBS) from 70 kHz (nekton Sv, dB).

The distance bins I created are; 0-20 m, 20-40 m, 40-60 m, 60-80 m, 80-100 m, 100-200 m, 200-300 m, 300-400 m, 400-500 m, and >500 m, referred to as distance bins 1-10 respectively. Layers were created by dividing the water column into thirds for analysis to determine if biomass changed with depth. Depth bins are; 0-30 m, 30-60 m, and >60 m, referred to as layers 1 to 3, respectively. Season is defined as summer (June, July, August), fall (September, October, November), and winter (December, January, February). Spring was not sampled due to inclement weather. Sites are grouped into types; standing or toppled, with replicates within each group. The variance-to-mean ratio ( $\sigma^2 / \mu$ ) was calculated for each site, where  $\mu$  = mean  $S_v$  estimate for each site to determine whether individual sites were true replicates.

The spatial biomass distribution of nekton and fishes around all standing and toppled sites was examined with a generalized linear mixed model (GLIMMIX). All statistical tests were performed with SAS with a significance level of  $\alpha=0.05$ . The linearized version of MVBS, the volume backscattering coefficient ( $s_v$ ), was modeled using a GLIMMIX with a lognormal distribution (MacLennan et al. 2002). MVBS values were linearized by using:

$$S_v = 10 \log_{10}(s_v)$$

where  $S_v$  is the volume backscattering strength (decibels).  $S_v$  estimates were used to calculate a linearized version of the MVBS data (mean volume backscatter coefficient),  $s_v$  ( $m^{-1}$ ), defined by the relationship:

$$s_v = (10^{S_v/10}) * 10^6$$

where  $\overline{S_v}$  is the individual values of MVBS (or MVBS when  $s_v$  is averaged over a finite volume) in a log form. After the linearized values were used in the model, they were converted back into log form for comparison and visual representation purposes.

A full model including season, type, distance, depth and their interactions was simplified to retain only significant independent variables. The reduced model used for both data sets excluded season and its interactions. A random effect for season was included to account for variability between seasons that may exist.

The residuals from both reduced models were examined for normality. A Tukey adjusted post-hoc test was used to reveal pairwise combinations resulting in the highest significant interaction (depth by distance by type). Additionally, I determined the area of influence, defined as the area near the structure with significantly higher MVBS than greater distances, and compare across layers, and type. CTD data from 25 profiles (temperature and salinity) were plotted as  $\bar{X} \pm SE$  at discrete depths (every 10 m, from 0-100 m) to reveal differences in temperature and salinity based upon the factors season and water depth at the five sites.

## **Results**

In 2013 and 2014, data were collected during cruises in June 2013, August 2013, October 2013, December 2013, February 2014, and June 2014. Twenty-five acoustic surveys were conducted during this study.

CTD indicated a clear halocline and thermocline during the summer months; however the water column remained well mixed during the other two seasons (Figure 2.4). The CTD data show similar patterns for temperature (°C) and salinity (PSU) at toppled and standing platforms.

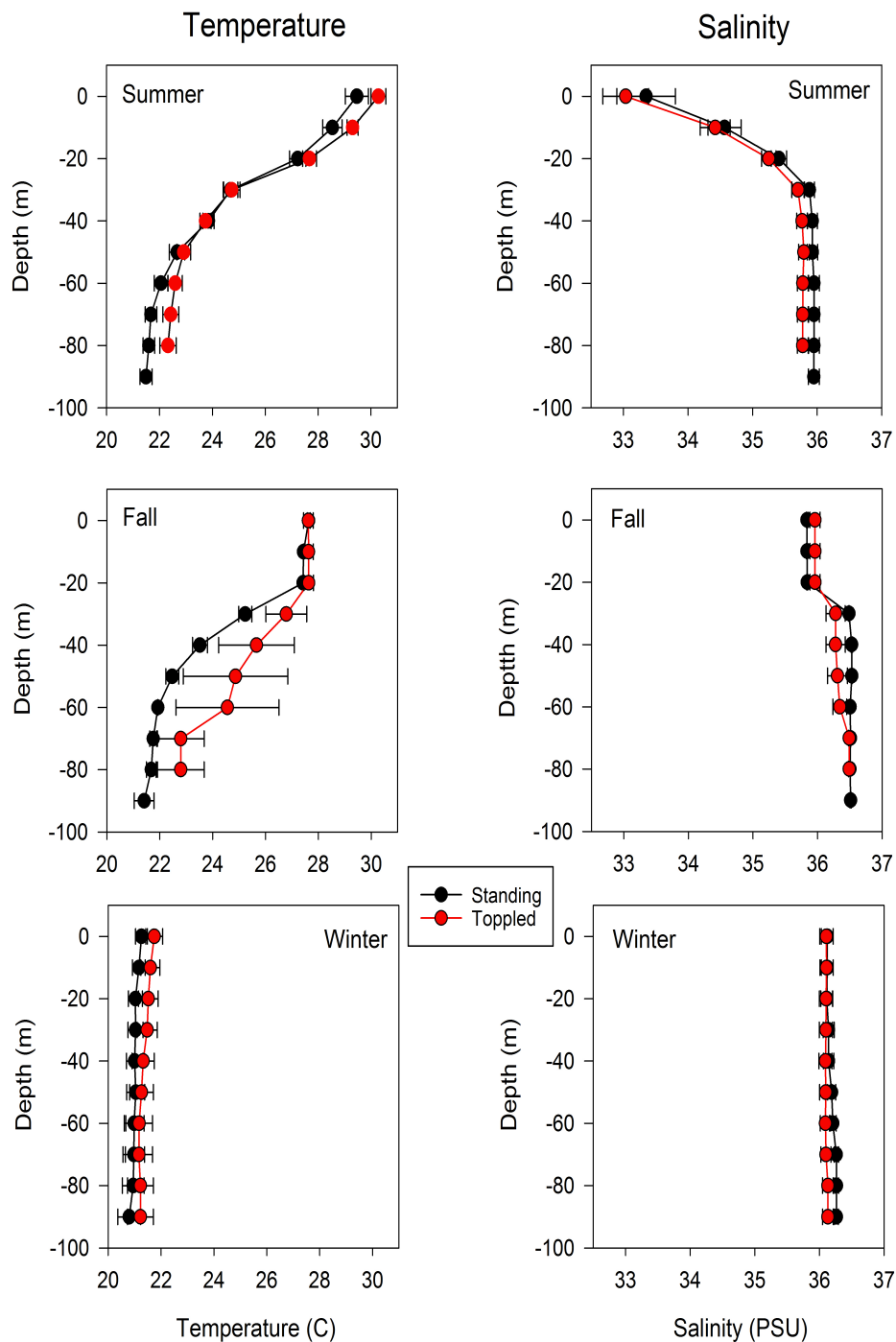


Figure 2.4. Average temperature ( $^{\circ}\text{C}$ ) (left column) and salinity (PSU) (right column) profiles at standing (black) and topped (red) platforms in the Eugene Island Oil and Gas Lease Block in the northern Gulf of Mexico. The top row is summer samples, middle row is fall samples and the bottom row is winter samples. Standard error bars are shown but may not be visible.

Individual sites were not different from each other based on calculation of the variation-to-mean ratio  $\sigma^2/\mu$  (Table 2.3). The ratios were almost identical and only vary by  $\sim 2\%$  from one another.

Table 2.3. Variance-to-mean ratios  $\sigma^2/\mu$  for each individual site sampled,  $\mu$  = mean  $S_v$  estimate for each site.

| Variable Type | Replicate Sites | $\sigma^2/\mu$ |
|---------------|-----------------|----------------|
| Standing      | EI 325A         | -389.86        |
|               | EI 346A         | -395.48        |
|               | EI 320B         | -389.53        |
| Toppled       | EI 322A         | -399.53        |
|               | EI 324A         | -404.37        |

#### **Nekton data set**

Results of the MVBS GLIMMIX model indicated a significant three-way interaction between depth, distance, and type ( $p < 0.0001$ , Table 2.4). Each factor and interaction contributed significantly to the variation in MVBS observed. LSmean MVBS values from standing platforms ranged from a low of -79.3 dB to a high of -64.3 dB, and values from toppled platforms ranged from a low of -79.1 dB to a high of -63.1 dB. It is important to keep in mind that a higher negative value of MVBS indicates a lower relative acoustic biomass. LSmean MVBS values were plotted for each unique three-factor interaction combination. For both platform types, LSmean MVBS values were highest overall depth layer 3 ( $>60$  m) and lowest overall in depth layer 1 (0-30 m) (Figure 2.5). Regardless of depth layer, LSmean MVBS values were highest in distance bins 1-2 (0-40 m) from the geographic center point of the structure and declined rapidly with distance, reaching a consistent level of noise after approximately 100 m from the structure ( $\sim -77$  to  $-80$  db).

Table 2.4. ANOVA table type III fixed effects from the GLIMMIX model for the nekton dataset comparing MVBS with depths, distance, and type of structure. Significance was set at  $\alpha=0.05$  for all tests. Model DF=model degrees of freedom. Error DF= error degrees of freedom.

| Source                          | Model DF | Error DF | F-value | p-value |
|---------------------------------|----------|----------|---------|---------|
| Depthlayer                      | 2        | 98141    | 1652.24 | <0.0001 |
| Distancebin                     | 9        | 98141    | 1288.10 | <0.0001 |
| Depthlayer x Distancebin        | 18       | 98141    | 176.76  | <0.0001 |
| Type                            | 1        | 98141    | 459.23  | <0.0001 |
| Depthlayer x Type               | 2        | 98141    | 8.93    | <0.0001 |
| Distancebin x Type              | 9        | 98141    | 25.86   | <0.0001 |
| Depthlayer x Distancebin x Type | 18       | 98141    | 16.33   | <0.0001 |

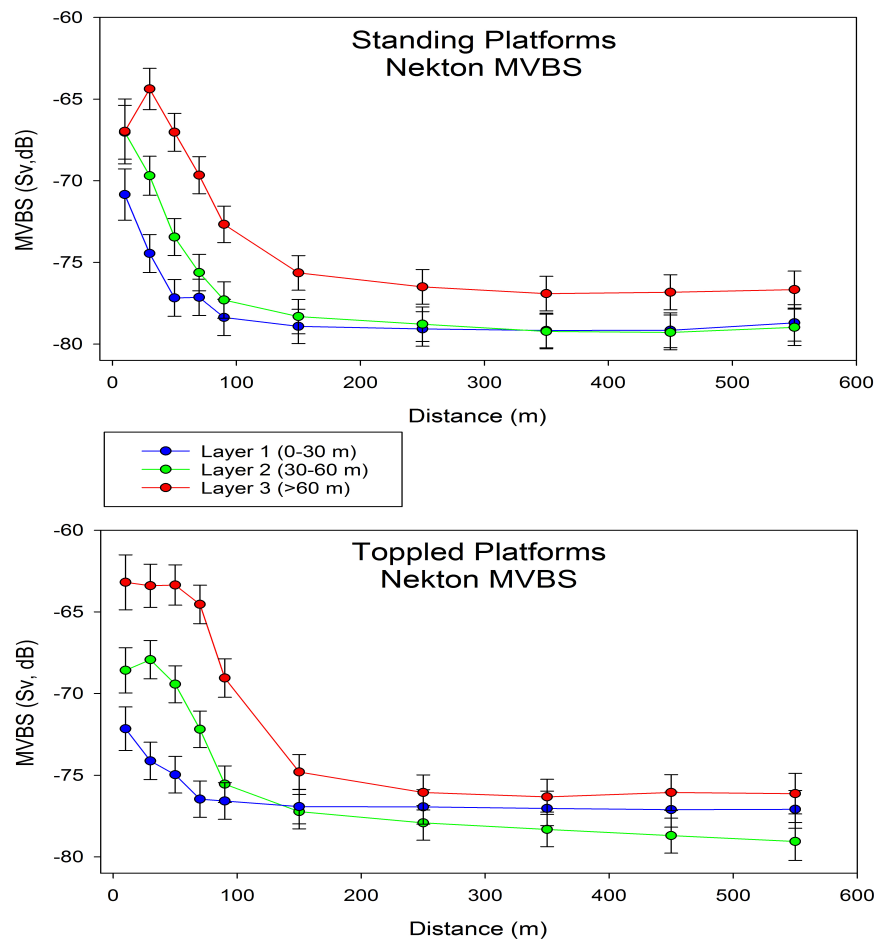


Figure 2.5. LSmean-volume backscatter (MVBS, Sv) of nekton data set versus distance from the structure at both types of habitat (standing and toppled platform) in the northern Gulf of Mexico. Top graph shows standing platforms, and bottom graph shows toppled platforms. The blue line represents depth layer 1 (0-30 m), green line represents depth layer 2 (30-60 m) and red line represents depth layer 3 (>60 m). Error bars indicate 95% confidence intervals.

Post-hoc multiple comparison indicate that MVBS was highest in distance bins 1-4 (0-80 m) in depth layer 1 (Tukey:  $p < 0.0001$ ). In depth layers 2 and 3, MVBS was highest in distance bins 1-6 (0-200 m) of the structure (Tukey:  $p < 0.0001$ ) (Appendix A).

At toppled platforms, MVBS was highest in distance bins 1-3 (0-60 m) in depth layer 1 (Tukey:  $p < 0.0001$ ). However, in depth layers 2 and 3, MVBS was highest in distance bins 1-5 (0-100 m) of the structure (Tukey:  $p < 0.0001$ ) (Appendix A).

### **Fish data set**

Results of the MVBS GLIMMIX model indicated a significant three-way interaction between depth, distance, and type ( $p < 0.0001$ , Table 2.5). All factors and interactions contributed significantly to the variation in MVBS observed. LSmean MVBS values from standing platforms ranged from a low of -84.8 dB to a high of -77.2 dB, and values at toppled platforms ranged from a low of -84.8 dB to a high of -77.7 dB. When compared to the nekton data set, the range of MVBS values were smaller. LSmean MVBS values were plotted for each unique three-factor interaction combination. For both platform types, LSmean MVBS values were highest overall in depth layer 3 (>60 m) and lowest overall in depth layer 1 (0-30 m) (Figure 2.6). The LSmean MVBS values were variable at the toppled platforms within distance bins 1-5 (0-100 m) of the structure. Regardless of structure type, LSmean MVBS rapidly declines with distance, reaching a consistent level of noise after approximately 100 m from the structure (~-82 to -84 db).

Post-hoc multiple comparison indicate that MVBS was highest in distance bins 1-2 (0-40 m) in depth layer 1 (Tukey:  $p < 0.0001$ ). In depth layer 2, MVBS was highest in distance bins 1-5 (0-100 m) of the structure (Tukey:  $p < 0.0001$ ). In depth layer 3, no significant differences existed (Tukey:  $p < 0.9999$ ) (Appendix A).



Table 2.5. ANOVA table type III fixed effects from the GLIMMIX model for fish data set comparing MVBS with depths, distance, and type of structure. Significance was set at  $\alpha=0.05$  for all tests. Model DF=model degrees of freedom. Error DF= error degrees of freedom.

| Source                          | Model DF | Error DF | F-value | p-value |
|---------------------------------|----------|----------|---------|---------|
| Depthlayer                      | 2        | 98170    | 485.76  | <0.0001 |
| Distancebin                     | 9        | 98170    | 60.75   | <0.0001 |
| Depthlayer x Distancebin        | 18       | 98170    | 10.60   | <0.0001 |
| Type                            | 1        | 98170    | 45.99   | <0.0001 |
| Depthlayer x Type               | 2        | 98170    | 7.34    | <0.0006 |
| Distancebin x Type              | 9        | 98170    | 24.47   | <0.0001 |
| Depthlayer x Distancebin x Type | 18       | 98170    | 6.95    | <0.0001 |

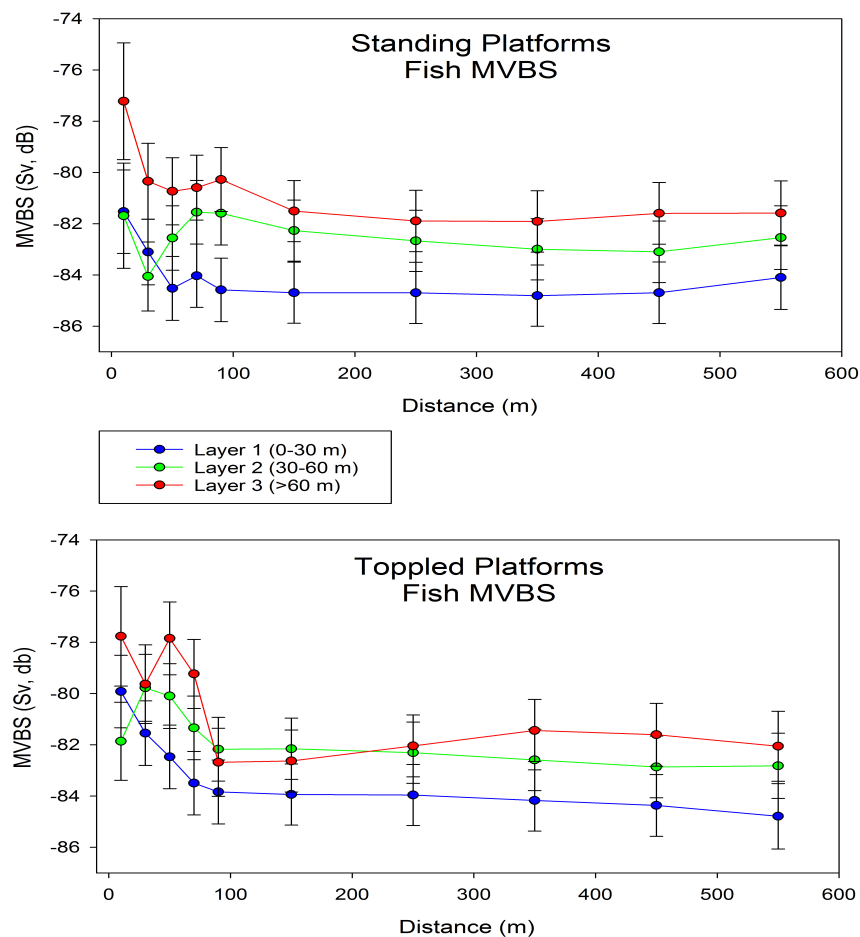


Figure 2.6. LSmean-volume backscatter (MVBS, Sv) of fish data set versus distance from the structure at both types of habitat (standing and toppled platform) in the northern Gulf of Mexico. Top graph shows standing platforms, and bottom graph shows toppled platforms. The blue line represents depth layer 1 (0-30 m), green line represents depth layer 2 (30-60 m) and red line represents depth layer 3 (>60 m). Error bars indicate 95% confidence intervals.

At toppled platforms, MVBS was highest in distance bins 1-3 (0-60 m) of the structure (Tukey:  $p < 0.045$ ), in depth layer 1. In depth layers 2 and 3, MVBS was highest in distance bins 1-4 (0-80 m) (Tukey:  $p < 0.0339$ ) and distance bins 1-5 (0-100 m) (Tukey:  $p < 0.0001$ ), respectively.

## **Discussion**

This study used acoustic data to determine the spatial biomass distribution of fishes around standing and toppled oil and gas platforms in the northern GOM. Similar studies have focused primarily on the distribution of biomass associated with artificial reefs (Stanley and Wilson 1997, 1998, 2000, 2003, Boswell et al. 2007, 2010, Simonsen 2013). Like many of these studies, this study found that fish and nekton MVBS was highest near the structure itself, but values varied in relation to depth, type of structure, and distance from structure.

The decline of acoustic biomass of fishes with increased distance from the structure appears to be consistent with those reported by Gerlotto et al. 1989, Stanley and Wilson 1996, 1997, 1998, 2003, Boswell et al. 2007, 2010, Shipley and Cowan 2010, Harwell 2013, and Simonsen 2013. Simonsen (2013) conducted transects of 2 km in length at the same sites as this study, in 2010; her results showed clear patterns of MVBS decreasing exponentially with distance, with highest levels of biomass within approximately 100 m of the structures. Transects in this study were shortened to focus on a smaller area around the structures due to consistently low levels of acoustic biomass past 200 m from the structure in Simonsen (2013). My study indicated a higher overall MVBS within distance bins 1-2 (40 m) of the geographic center point of the structures, indicating that these hard structures in an otherwise flat, muddy area may be suitable

habitat for reef-associated fishes. The background noise levels reached were what I would expect to observe over unstructured area. These background noise levels are consistent with Simonsen (2013), who also observed these similar levels around 100 m away from the structure. Due to the periodic calibration of my acoustic equipment, direct comparisons of MVBS between results of this study and others should be interpreted with caution, even as the overall trends are identifiable. These artificial reef structures may be providing habitats to fishes and organisms, which would make these artificial reef structures a valuable resource to manage. However, if these structures are simply attracting reef-associated fishes from nearby habitats, they could make these fishes more vulnerable to exploitation (Cowan et al. 2011).

Results from this study show a compact area of influence around standing structures and are similar to results reported by Stanley and Wilson (1996, 1997, 2000) and Boswell et al. (2010). The area of influence around the standing and toppled structures for the nekton and fish MVBS was variable with respect to depth and distance. Nekton MVBS, compared to fish MVBS, had a smaller area of influence in shallower depths compared to deeper ones, which leads me to believe that fishes and other organisms may be inclined to group closer together in the shallower depths due to vulnerability and foraging behaviors. The deeper depths may provide more protection due to less predators and limited light, allowing for fishes to move away from the protection the structure may be providing. Additionally, the toppled structures had a larger area of influence compared to the standing. The configuration of toppled structures is not consistent, which may be responsible for the pattern I observed. Even though the areas of influence varied by type and layers, it is important to emphasize that a

statistically significant difference in MVBS between varying distances away from the structure does not necessary imply a biological significance.

This study found that nekton and fish MVBS was significantly different between structure types and MVBS values at standing platforms were higher compared to toppled platforms, which is consistent with both Wilson et al. (2003) and Simonsen (2013). Even though MVBS at structure types were statistically different from each other, I question the biological significance of these results. There was a large number of data points generated from the acoustic surveys (98,203) that may have influenced the statistical significance of the effects. Due to this many data points, small differences can cause statistical significance. To interpret results in the context of a large dataset, I used variance-to-mean ratios to confirm that my individual sites were true replicates with each type group. Each individual site had a very similar ratio value indicating that they were true replicates with little variance between each site itself.

Based on the number of data points in this study, I would have been able to detect significance between two MVBS values that were  $<0.06\%$  different, with a confidence level of  $p=0.0005$ . Because of the large number of degrees of freedom in my statistical analyses, it is hard to conclude specific statistical results and therefore interpret this data. In the future, a random subsample from the data points would provide a more detailed and possibly more accurate estimate of the statistical significance of MVBS differences between site types. Furthermore, when determining the biological significance of these results, variables such as the structure type, distance from structure, and depth layers should be assessed in relation to the fish communities present to better understand the value of these structures as habitat for a variety of fishes, some of which are, and others

that are not, reef-associated. Statistical results should be used to support ideas instead of being used as absolute proof when evaluating a constantly changing environment with many variables. Therefore, parametric statistics may not be the most effective way to analyze and model this type of data.

The high MVBS in layer 3 (>60 m) at both structure types compared to layers 1 and 2 was the opposite of what Simonsen (2013) observed at the same sites, 3 years earlier. My MVBS for layer 3 were similar to Simonsen (2013) when comparing the fish echograms, however, layers 1 and 2 were drastically lower compared to the previous study. Less MVBS is present in 2013-2014 compared to 2009-2010. The shift in distribution in the water column may be due to prey and light availability in addition to schooling behaviors, but this does not explain the differences between my results and those of other studies (Wilson et al. 2003, Simonsen 2013). One possible explanation for the shift in biomass throughout the water column may have been caused by the unusually cold winter that occurred in-between Simonsen's (2013) and my studies. However, sea surface temperature records for those winters (2008-2014) do not show any dramatic temperature decreases that could explain behavior of fishes. Although sea surface temperatures do not reflect the entire water column temperature, I would assume that fish would move towards deeper, warmer waters. Effects of seasonal and oceanographic influences can be inconsistent, leading to varying results over time, sometimes with little known explanation.

While season was not statistically significant during this study, these results are different from Putt (1982), Stanley and Wilson (1998, 2000), and Simonsen (2013) who reported that fish density or MVBS significantly varied among seasons. The lack of

spring data and limited surveys during other seasons may have influenced my seasonal results. Additionally, a previous study (Simonsen 2013) suggested that light from structures may impact fish distribution and diet around these structures and differences observed in fish distribution during this study may have been due to this source of variability. While the effects of light were not explored during this study, work to determine whether light has an effect on fish distribution and diet is ongoing (Foss, K.L.<sup>3</sup>, Barker, V.A.<sup>4</sup>, personal comm.).

A major challenge associated with acoustic surveys is the ability to determine the species composition that comprises the acoustic biomass (Simmonds and MacLennan 2005, Simonsen 2013). Decreasing values of Sv with increasing frequency are known to belong to swim-bladdered fishes while organisms lacking swim bladders are comparatively frequency independent at lower frequencies with a dramatic increase at 200 kHz (Korneliussen and Ona 2002, Korneliussen et al. 2009, De Robertis et al. 2010). Therefore, using 120 kHz and 70 kHz for dB differencing provided the most accurate results to differentiate between fishes from other nektonic scatters from shrimp, jellyfish, and zooplankton. It is important to note that Sv difference ranges used to define/identify fishes in this study were developed based on literature from a variety of applications (Madureira et al. 1993, Korneliussen and Ona 2002, Mosteiro et al. 2004, Simmonds and MacLennan 2005, Korneliussen et al. 2009, De Robertis et al. 2010, Simonsen 2013) and therefore may not perfectly classify groups. This difficulty is exacerbated by the rich species diversity of my study sites and thus the varied morphologies and acoustic

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<sup>3</sup> Foss, K.L. 2015. Louisiana State University, MS Student, Department of Oceanography and Coastal Sciences.

<sup>4</sup> Barker, V.A. 2015. Louisiana State University, MS Student, Department of Oceanography and Coastal Sciences.

responses encountered during my surveys. For instance, sharks do not have swim bladders and their biomass return from acoustic surveys is disproportional to their actual sizes.

Moreover, I did not make the assumption that species composition was the same between structure types, therefore TS was not an appropriate metric to use for analysis. TS is sensitive to water temperature, orientation of the fishes, species present, and other factors that can influence these values (Simmonds and MacLennan 2005). Because artificial reefs in the GOM are home to many reef-associated fish species, attempting to differentiate TS for each species would have been impossible.

Using three split-beam transducers allowed for thorough data collection around these large and complex structures. Previous studies used stationary acoustic transducers to assess the species density, biomass, and composition around standing oil and gas platforms (Stanley and Wilson 1996, 1998, 2000a, 2003). While stationary acoustics are a useful monitoring tool for difficult habitats such as these structures, mobile acoustic surveys should be used in conjunction with stationary acoustic surveys to determine the total distribution and biomass of the fishes located within and directly adjacent to the structure. Because only mobile acoustic transects were conducted, the structure in addition to the potential noise coming from reflections off the structure were excluded from the analysis, along with associated biomass inside the structure. This could create bias, leading to over or under exaggeration of biomass around these structures.

Consistent future surveys of additional standing and toppled oil and gas platforms would allow for a better understanding of the spatial biomass distribution of fishes around oil structures in relation to temporal and environmental trends. Efficient data

collection is necessary when there are many structures separated by large distances, therefore the survey methods used in this study may allow for artificial reef programs such as the LARP to decrease effort and sampling time. The statistical results of these surveys should be interpreted with caution due to a large sample size and their biological significance should be examined more closely.

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## CHAPTER 3: COMMUNITY STRUCTURE AND BIOMASS APPORTIONMENT OF FISHES AROUND STANDING AND TOPPLED OIL AND GAS PLATFORMS IN THE NORTHERN GULF OF MEXICO

### **Introduction**

There are over 6,000 active oil and gas platforms throughout the world, about 2,400 of which are located in waters of the Gulf of Mexico (GOM) and approximately 1,300 within the US Economic Exclusive Zone (EEZ) (Schroeder and Love 2004, Scarborough-Bull et al. 2008, BSEE 2015). These platforms provide hard substrate where little occurs naturally, attracting/aggregating demersal and pelagic fishes (Gallaway et al. 1981, Continental Shelf Associates 1982, Bohnsack and Sutherland 1985, Pickering and Whitmarsh 1997, Stanley and Wilson 1997, 2000, Boswell et al. 2010). Researchers have hypothesized that artificial reefs have the potential to diversify habitat, modify the communities of organisms in the surrounding areas, concentrate existing fishery resources, and increase fishery resources (Bohnsack et al. 1991, Seaman and Sprague 1991, Scarborough-Bull et al. 2008). Some of these hypotheses are mutually exclusive. Studies have also shown that fish abundance is 3-25 times higher within 16 m of platforms in the GOM and attract both recreational and commercial fishers, in addition to SCUBA divers and spear fishers (Stanley and Wilson 2003, Wilson et al. 2006, Keenan et al. 2007, Galloway et al. 2009, Harwell 2013).

The scientific investigation of fishes at oil and gas platforms did not begin until the late 1950's and provided basic insight into structure-associated fish communities (Carlisle Jr. et al. 1964). Due to the challenges of limited visibility, diver avoidance, gear bias, complex architecture, and extreme water depths, the survey methods varied which made the comparison of results difficult (Stanley and Wilson 1997, 2000). Previous

studies were restricted due to these challenges and provided limited data. In 1982, a fixed array of 8-mm movie cameras was used to collect data at a platform over a three month period (Putt 1982, Stanley and Wilson 1997). This study was considered lengthy and demonstrated that the species composition and abundance of fishes varied with time. Over the last several decades, video studies have become increasingly technologically advanced and practical to use. Though technology and survey techniques have advanced, the lack of standardization between studies has resulted in limited data comparisons (Stanley and Wilson 1997).

Standing oil and gas platforms provide unique structure that extends throughout the entire water column. Toppled platforms, compared to standing, have less vertical structure due to Coast Guard regulations requiring them to sit at least 30 m below the surface (LARP 2014). Because of different levels of light availability throughout the water column, some encrusting species occupy areas based upon their light needs. Various reef-associated crustaceans, invertebrates, and fishes reside on or near platforms (Gallaway and Lewbel 1982, Dokken et al. 2000). The standing structures also tend to attract pelagic fishes such as blue runner (*Caranx crysos*), greater amberjack (*Seriola dumerili*), king mackerel (*Scomberomorus cavalla*), barracuda (*Sphyraena barracuda*), and sharks, along with demersal fishes such as grouper species, snapper species (Brown et al. 2010, Simonsen 2013). A recent study determined that even though reef-derived resources entered benthic and nektonic food webs surrounding platforms, phytoplankton was the dominant basal resource fueling platform-dwelling communities (Daigle et al. 2013). This indicated that reef function may be less dependent upon reef-associated algae and serve primarily as a sink instead of producing organic matter that fuels food

webs. Based on these results, it is very possible that these structures serve primarily as sources of food and protection, not production (i.e., the production hypothesis for artificial-reef effects) (Daigle et al. 2013).

Fishers prefer to target these sites because fishes aggregate around structure. Fishing regulations, using both input and output controls, have increased to reduced fishing power because of the mandate in 2006 reauthorization of the Sustainable Fisheries Act to end overfishing on all species under federal management by 2010 (Cowan et al. 2011). This task has been difficult because little data is available for several species that may be vulnerable to exploitation, for example, deepwater groupers (Cowan Jr., J.H.<sup>5</sup> personal comm.). In some coastal zones, artificial reefs are designated as marine protected areas and are “currently considered as an efficient tool to manage and support small-scale coastal fisheries and to restore natural habitats affected by anthropogenic impacts” (Cresson et al. 2014). It is debatable as to whether these structures are providing essential habitat for economically important fishes or whether they are merely aggregating fishes and exposing them to predators and to fishers (Claudet and Pelletier 2004, Seaman 2007, Tessier et al. 2014). Unfortunately, Fisheries Governance in the Gulf of Mexico has not designated any artificial reefs as no-take reserves. However, it is likely that such structures both provide additional habitat, and serve to aggregate fishes simultaneously, making management difficult. In addition, about 75% of the platforms that are decommissioned are removed with explosives (SEDAR 2013), causing an unquantified rate of additional mortality.

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<sup>5</sup> Cowan, Jr., J.H. 2014. Louisiana State University, Department of Oceanography and Coastal Sciences.

The available data on species biomass composition at standing and toppled platforms is limited due to the difficulty of sampling large offshore structures in deep water with limited visibility. Managing the decommissioning or removal of platforms, along with their fish communities can be challenging because of the limited amount of data scientists have on these ecosystems. Therefore, gathering additional data about the species biomass composition and community structure of fishes around structures in the northern GOM is important and necessary to better manage these habitats and species. Characterizing these fish communities can lead to future rapid assessments around these numerous structures in the GOM.

Non-destructive survey techniques such as video surveys can provide data that is useful in making informed decisions about the utilization of these structures by fishes. Recently, baited remote underwater video (BRUV) arrays have been a successful alternative to various other video systems such as remote operated vehicles (ROVs). They are cost effective, robust sampling tools that can easily and efficiently assess community composition, distribution, relative abundance, and sizes of marine fishes. BRUVs have successfully been utilized in studies in Australia, Hawaii, the GOM, and other ecologically important locations (Rooker et al. 1997, Watson et al. 2005, 2010, Brooks et al. 2011, Martinez et al. 2011, Merritt et al. 2011, Dorman et al. 2012, Harvey et al. 2013, Misa et al. 2013, Whitmarsh et al. 2014). Stereo-video systems can be incorporated in BRUVs and are frequently used because of the assumption that measurements of live fishes are as accurate and precise as those made of stationary objects under the best recording conditions (Harvey et al. 2002).

The use of BRUVs has been compared to SCUBA diver surveys to determine their effectiveness. In a recent study, the use of a stereo-BRUV array provided more accurate estimates of length-frequency distributions than those derived from fishery-independent line fishing (Langlois et al. 2012). Furthermore, Harvey et al. (2004) compared underwater visual distance estimates made by SCUBA divers to measurements made by stereo-video systems and concluded that even though SCUBA surveys are useful for a quick evaluation of the composition and abundance of reef fish communities, the data suffers from biases and errors in estimates of distance to, and the length of the fishes. Harvey et al. (2004) reported that for experienced scientific divers, distance errors may potentially result in up to an 82% underestimate or up to 194% overestimate of the actual area sampled, which affects estimates of fish density; whereas stereo-video systems under-estimated distance but to a much lesser degree with less variability than divers resulting in more consistent sampled areas (Harvey et al. 2004).

I used a baited remote underwater stereo-video (BRUSV) array along with a more traditional BRUV for non-destructive data collection around standing and toppled platforms. Previous studies have tested various camera systems for their effectiveness and precision when sampling and monitoring fish communities; they have been found to accurately identify species, size, and sex of individuals. Also, they can be used with a variety of species; Brooks et al. (2011) determined that they offer a non-destructive alternative to long line surveys for monitoring broad trends in the relative abundance of sharks. The non-destructive nature of these systems means they can also be used in place of or in addition to trawls, and are useful when sampling areas where the habitats are difficult to survey or fishing is not permitted (Wells 2007, Martinez et al. 2011).



Furthermore, a study conducted by Harvey et al. (2002) suggested that using stereo-video for measurements of fishes is more accurate than measurements of fishes taken from experienced scientific divers.

The goal of this chapter was to compare the community structure of fishes at standing and toppled oil and gas platforms in relation to season, type of structure, and depth layers in the water column using video surveys. Video data was collected by both BRUVs and BRUSVs and were as described above to accomplish my goal. I hypothesized that more species would be present at the standing platforms compared to the toppled platforms. Additionally, I hypothesized that coupling acoustic and video methods would provide me with the ability to ground-truth the acoustic data. Lastly, I hypothesized that red snapper would be the dominant species observed and contribute the most to relative biomass around these structures. Pairing acoustic surveys with video sampling (BRUV and BRUSV) provides the opportunity to ground truth acoustic estimates of biomass and provides data on the species contributing to acoustic estimates.

## **Methods**

### **Study Area**

Five oil and gas platforms in the Eugene Island (EI) Oil and Gas Lease Block were sampled between June 2013 and June 2014. The sites consisted of three standing and two toppled oil and gas platforms located approximately 130 km off the coast of Louisiana (Figure 3.1). Standing platforms are located in blocks EI 346, EI 325, and EI 320 and toppled platforms are in blocks EI 322 and EI 324. The standing platforms, EI 346A, EI 325A, and EI 320B, are operational platforms that began production in 1989, 2000, and 2006 respectively.

Hurricane Lili damaged the two standing platforms beyond repair in 2002, which led them to be decommissioned and later toppled. They were donated for biological study by British Petroleum (EI 322A) and Newfield Exploration Company (EI 324A) to the LARP (Kaiser and Kasprzak 2008). The three standing platforms are located in ~ 85 m depth and the two toppled platforms are located in ~ 70 m (Table 3.1). Study sites were chosen to facilitate direct comparisons with a previous study that sampled EI 325A, 346A, 322A, and 324A in 2009 and 2010 (Simonsen 2013).

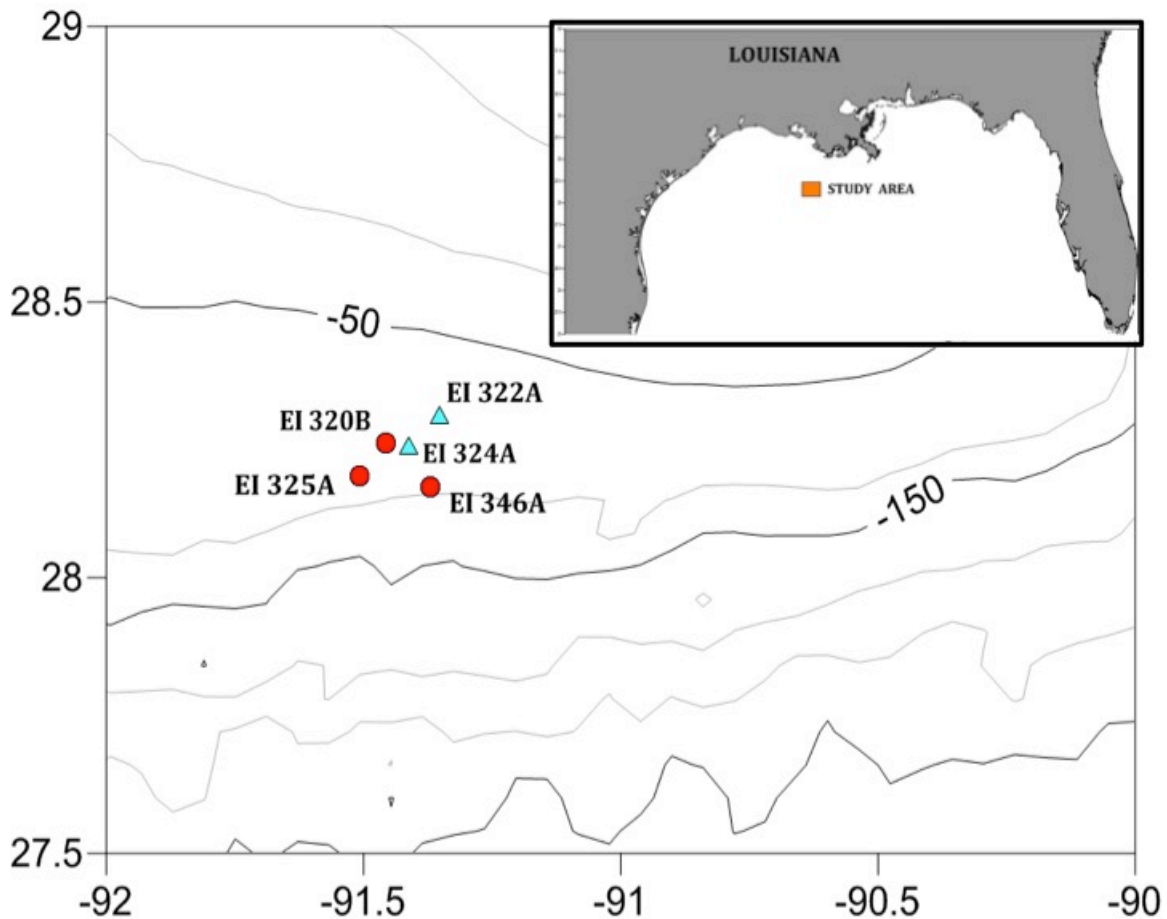


Figure 3.1. Map of study site approximately 130 km off the coast of Louisiana. Five study sites shown: two toppled platform sites (blue triangles); three standing platform sites (red circles).

Table 3.1. Physical geography of the three standing and two toppled oil and gas platforms in the Eugene Island (EI) Oil and Gas Lease Block. Values obtained from the Bureau of Safety and Environmental Enforcement (BSEE 2014) and the Louisiana Artificial Reef Program (LARP) (Mike McDonough personal comm.<sup>6</sup>).

| SITE  | COMPANY                                    | LATITUDE      | LONGITUDE      | DEPTH | INSTALLED | TOPPLED | PILES  |
|---|--|---------------|----------------|-------|-----------|---------|--------|
| EI 346<br>“A”<br>(standing)                       | Fieldwood<br>Energy                        | 28° 9'49.97"N | -91°22'8.19"W  | 95m   | 2000      | n/a     | 4-pile |
| EI 325 “A”<br>(standing)                          | Fieldwood<br>Energy                        | 28°14'39.00"N | -91°27'26.00"W | 76m   | 1989      | n/a     | 4-pile |
| EI 320 “B”<br>(standing)                          | Arena Offshore                             | 28°15'3.28"N  | -91°26'2.60"W  | 76m   | 2006      | n/a     | 3-pile |
| EI 324 “A”<br>(toppled)                           | Newfield<br>Exploration-<br>currently LARP | 28°14'8.23"N  | -91°24'40.50"W | 80m   | 1990      | 2002    | 4-pile |
| EI 322 “A-<br>prod” and<br>“A-drill”<br>(toppled) | British<br>Petroleum-<br>currently LARP    | 28°17'22.34"N | -91°21'5.50"W  | 70m   | 1978      | 2002    | 4-pile |

<sup>6</sup> McDonough, M. 2014. Louisiana Artificial Reef Program, Louisiana Department of Wildlife and Fisheries.

## **Sampling Method**

The methods for acoustic surveys will not be repeated, see Chapter 2. Acoustic, video, and CTD surveys were conducted quarterly at each site from June 2013 through June 2014, as weather and scheduling permitted, to examine seasonal and habitat effects on fish biomass. Sites were chosen to replicate the size of structures to the degree possible (Table 2.1). All study sites are artificial reefs, however they will henceforth be identified as either standing or toppled.

Environmental data were collected with a Sea-Bird SBE 25 Sealogger CTD to measure temperature and salinity relative to depth at each site. These parameters are needed for sound speed correction during acoustic analysis and are known to influence fish distribution. The CTD was acclimated at the sea surface for three minutes before descent, and was deployed vertically at a steady rate of approximately 1 m/s. Both descent and ascent were used for verification.

### Video Collection

#### Standing Platforms

BRUSVs containing six Vixia HF G10 high-definition camcorders were mounted in a custom-made camera cage, modified from a NOAA video array design, with 360° view capability. The cage consisted of two stereo camera pairs, located on opposite sides of the cage, and two single-cameras, which were mounted orthogonally to the stereo pairs (Figure 3.2). The stereo cameras within each pair were separated by 70 cm from one another and angled inward at 7 degrees to account for a wide range of individuals and measurements. Before deployment, stereo pairs were calibrated with a calibration cube

(1 m x 1 m x 0.53 m) and recorded video was processed with the computer program *Cal* (SeaGIS Pty. Ltd) to ensure accurate length measurements.

Four 50 watt HID lights (Light Monkey Enterprises LLC, Florida, USA) were mounted on the top of the cage, each one providing light to either a single camera or a stereo camera set. Each light had a self-contained battery and provided 5000 Lumen Output with a battery life of approximately two hours.



Figure 3.2. Stereo camera array with six cameras; two pairs of stereo cameras and two single cameras positioned orthogonally for a 360° view. Modified from the NOAA camera array design.

The cage was baited with gulf menhaden (*Brevoortia patronus*) or chub mackerels (*Scomber japonicas*) (both whole and ground) and deployed using a winch when the boat was approximately 20-30 m away from the structure to avoid any collisions with the structure itself. The upper, middle, and lower water columns were surveyed, purposefully avoiding the seafloor due to poor visibility caused by the nepheloid layer. Layers were created by dividing the water column into thirds for analysis to determine if community structure changed with depth. Depth bins are: 0-30

m, 30-60 m, and >60 m, referred to as layers 1 to 3, respectively. The array was suspended for 20 minutes in the middle of each depth layer (layers 1 to 3), for a total soak time of 60 minutes per site.

#### Toppled Platforms

The toppled platforms were surveyed with a BRUV array designed to minimize the chances of entanglement on any unexpected pieces of the platform that may have shifted over time. Four Go-Pro Hero 3 Silver Edition<sup>®</sup> cameras were clamped onto a copper pipe with a ¾ inch diameter, in a spiral pattern allowing for a 360° view between the top three cameras and a downward facing view from the bottom camera (Figure 3.3). Lead weights (approximately 20-30 lbs) were attached at the bottom of the array depending on the strength of the current, to ensure a vertical profile. The cameras were set on the highest resolution (1080 p), medium view frame, and placed in 5000 ft depth rating underwater housings made by GroupBInc to reach the appropriate depths.

The array was baited with gulf menhaden or chub mackerels (both whole and ground) and deployed using a winch as close to the structure as possible. The three depth layers were surveyed, purposefully avoiding the seafloor due to poor visibility due to the nepheloid layer. The array was suspended for 20 minutes in the middle of each depth layer (layers 1 to 3), for a total soak time of 60 minutes per site.

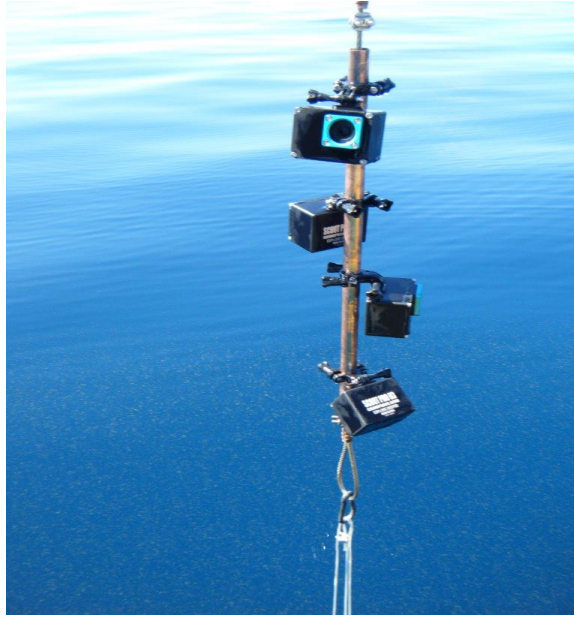


Figure 3.3. Go-Pro<sup>®</sup> camera array with four cameras, allowing for 360° view and a downward facing view. Cameras placed in GroupBInc housings, lead weight attached at bottom of the pipe to ensure a vertical profile.

### Data Processing

#### Video Data

Video from both camera arrays was post-processed in *EventMeasure* (SeaGIS Pty Ltd). Data processing followed protocols and guidelines similar to the NOAA Panama City Lab protocols (NOAA Panama City Lab Inshore NE Gulf Reef Fish Survey 2014). These guidelines outlined specific characteristics to record during the surveys including habitat, species identification, lengths, and count of fishes. The habitat guideline was modified for this study, as type of structure (standing or toppled platform) was the only habitat. My research partner, Alayna Petre, was responsible for post-processing the video data in *Eventmeasure*.

A continuous 20-minute video was examined for each layer sampled. Start time began 30 seconds after the array reached the depth layer 1, to account for possible bias caused by noise and movement disturbances that may have affected fish behavior. Data

was not eliminated from the video processing in layers 2 and 3, as it was assumed the fishes were acclimated to the camera array.

All fishes were identified to the lowest possible taxonomic level and counted. MaxN, defined as the maximum number of an individual species observed at any one time on the video, was used for all analysis and comparisons (Priede et al. 1994, Langlois et al. 2012). This parameter provides a conservative estimate of the number of an individual species that is assumed to be present at that time and is primarily used for stationary systems to account for possible recounting of fishes, which is possible due to the centralized bait system in our array (Kallayil et al. 2003). Individual camera information was combined at the survey level to account for all individual observations and to avoid replication. Estimates of fork length (FL) were recorded when individual species MaxN fishes were observed, to eliminate repeated measurements of the same fish. If a fish could not be accurately measured at time of MaxN, it was measured at the next opportunity, without being recounted. It is important to note that the downward-facing camera from the BRUV was not used for data processing, as it was too difficult to determine species and count from a downward-facing view. Rather, it aided in species identification when necessary.

Fish assemblage data were analyzed using procedures within PRIMER v. 6 (Plymouth Routines In Multivariate Ecological Research, Ltd., Ivybridge, UK). Within PRIMER, data was entered and organized by survey (site and date), species, count, and layer. Fish that could not be identified to species were excluded from all analyses.



## Biomass Apportioning

BRUSVs provide us with size, abundance, and species identification of fishes. Fish lengths taken from underwater video systems can be an accurate method to determine factors such as the biology and ecology of fish populations (Osenberg and Mittelbach 1989, Jennings and Dulvy 2005, Fisher et al. 2010, Langlois et al. 2012). Using the acoustic data from Chapter 2 and video together allows me to roughly correlate and apportion the biomass of individual species based on video data with MVBS distributions from the acoustic data.

Each length measurement recorded from the video surveys had an associated Root Mean Square (RMS) within *EventMeasure*, which is a gauge of length measurements accuracy (Seager 2014). The RMS intersection values for all points in the measured length should ideally be 0. However, values  $< 10$  mm are considered acceptable, and values  $> 20$  mm are considered problematic due to calibration variation and synchronization issues (Seager 2014). Therefore, length measurements with an RMS value larger than 20 mm for the lengths observed from our surveys were excluded from analyses.

By using published length-weight regression relationships, the length of each fish was converted into a weight to allow for a calculation of relative biomass of fishes at each site. Due to the use of BRUVs at toppled platforms, length measurements were not collected. Lengths of fishes from the standing platform were used for their respective species at the toppled platforms as a close approximation since the sites are located close to each other.

Published length-weight relationship coefficients were obtained from FishBase (Froese and Pauly 2014). The most appropriate set of coefficients was used, based on criteria: sample size, location of survey, sex of fishes, and measurement parameter. Published length-weight relationship equations used for red snapper and scamp (*Mycteroperca phenax*) were taken from other reliable sources for more accurate results (Matheson et al. 1986, Wilson and Nieland 2001).

Length-weight relationships were calculated using the general equation:

$$W = aL^b$$

where w is weight in g, a is y-intercept, b is slope, and L is fork length in cm. For a complete list of coefficients used for each species and their respective sources, see Appendix B.

If a species was counted but could not be measured during any survey due to angle of fishes or distance from camera, the species was removed from the overall length-weight calculations. Also, if a species was measured in one layer but not in another, the average weight for the species was applied to that species in the other layer with the missing length.

## **Data Analysis**

### Video Data

I compared the effects on the community structure of fish of variables including type of site, season, and layers in the water column. These variable definitions are identical to those defined in Chapter 2 to keep results comparable.

First, data were square root transformed to down weight the contribution of numerically dominant species, which could bias the results, followed by a resemblance

matrix calculation (Clarke and Gorley 2006). This step defined the zero-adjusted Bray-Curtis similarity between every pair of samples. Zero-adjusted means a “dummy species” was added to each sample which forced two samples with no content to be 100% similar and two samples with a single real individual to have some similarity.

Patterns of species diversity among type of structure, layers, and seasons were examined with multiple species diversity indices calculated for each survey (Clarke and Warwick 2001). Species diversity indices included species richness ( $S$ ), Shannon Diversity ( $H'$ ), and Pielou’s evenness ( $J'$ ). Species richness ( $S$ ) is simply the total number of species present in the survey. Diversity was calculated as the Shannon Diversity index ( $H'$ ):

$$H' = -\sum_i p_i \ln p_i$$

where  $p_i$  is the proportion of the total count of individual fish within that survey arising from the  $i$ th species (Clarke and Warwick 2001). Equability, or evenness, was calculated as Pielou’s evenness index ( $J'$ ):

$$J' = \frac{H'}{H'_{\max}} = \frac{H'}{\log S}$$

where  $H'_{\max}$  is the maximum possible value of Shannon Diversity, or the maximum that would be achieved if all species were equally abundant (Clarke and Warwick 2001). I calculated patterns of species diversity among type, layers, and season using three one-way ANOVA (SAS v. 9.4)

Permutational multivariate analysis of variance (PERMANOVA) was performed based on the resemblance matrix data set to analyze the data in response to the variables of interest. The design included variables type, layer, and season, which were all fixed

effects. Pair-wise tests were performed on any statistically significant ( $\alpha=0.05$ ) factors determined from the PERMANOVA test, to determine which levels of each factor were statistically different ( $\alpha=0.05$ ) from one another.

To evaluate species-specific contributions to the statistically significant groups from the pair-wise tests, similarity percentages analyses (SIMPER) were performed, post-hoc. Species contributing to similarities and driving dissimilarities were examined. All SIMPER analyses were one or two-way analyses with Bray-Curtis similarity test with a cutoff percentage of 90% to determine those species accounting for 90% of the total similarities and differences.

A distance-based linear model (DistLM) was performed to analyze and model the relationship between the fish communities observed and the environmental data measured: salinity (PSU) and temperature ( $^{\circ}\text{C}$ ). By analyzing the fish assemblage data first and then determining how well the information on environment matches the community structure, the linear model can attempt to explain the variation in the fish assemblage data by the environmental data (Clarke and Warwick 2001, Clarke and Gorley 2006). The models with the lowest Akaike information criterion (AICc) scores were selected.

CTD data from 25 profiles (temperature and salinity) were plotted as  $\bar{X} \pm SE$  at discrete depths (every 10 m, from 0-100 m) to reveal differences in temperature and salinity based upon the factors season and water depth at the five sites.

### Biomass Apportioning

Definitions of variables for site type, depth layers, and seasons are identical to those defined in Chapter 2. First, every individual length was converted to a weight

based on published length-weight relationships to avoid additional variation from averaging. All weights for an individual species, in any given layer during a survey, were averaged to have a mean weight per individual species per layer. For example, if species A was observed in all three layers, an average weight for species A in each of layer 1, layer 2, and layer 3 was calculated. Since the entire structure was not surveyed, relative biomass (%) and relative MaxN (%) were calculated for layer-season combinations and type.

To calculate relative biomass, a species mean weight was first multiplied by the species total count (MaxN) to estimate biomass (g) of an individual species. Then, individual species biomass values were summed both within layers by season and within site type (standing/toppled). Relative biomass (%) was defined by the relationship:

$$\text{Relative Biomass (\%)} = \frac{\text{Indiv. species biomass}}{\text{Total biomass}} \times 100$$

To calculate relative MaxN of individual species, an individual species MaxN was summed both within layer by season and within type to calculate a total MaxN for that species. Relative MaxN (%) was defined by the relationship:

$$\text{Relative MaxN (\%)} = \frac{\text{Indiv. species MaxN}}{\text{Total MaxN}} \times 100$$

Both of these calculations (relative biomass % and MaxN %) were repeated for both the layers by season and type groupings.

An estimate of the mean number of an individual species expected on any given survey were calculated by averaging a species MaxN within a given layer, season, or at a specific site type. A standard deviation was calculated for each of these values.

## Results

In 2013 and 2014, data were collected during cruises in June 2013, August 2013, October 2013, December 2013, February 2014, and June 2014. Twenty-five video surveys were conducted during this study.

CTD indicated a clear halocline and thermocline during the summer months; however the water column remained well mixed during the other two seasons (Figure 3.4).

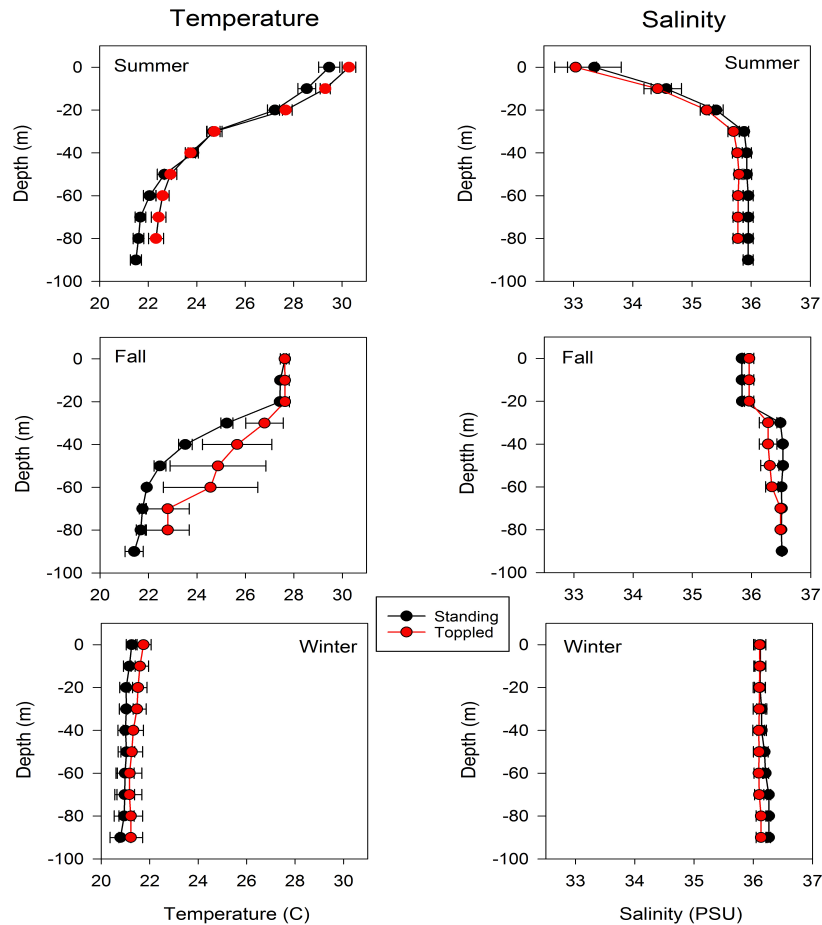


Figure 3.4. Average temperature (°C) (left column) and salinity (PSU) (right column) profiles at standing (black) and toppled (red) platforms in the Eugene Island Oil and Gas Lease Block in the northern Gulf of Mexico. The top row is summer samples, middle row is fall samples and the bottom row is winter samples. Standard error bars are shown but may not be visible.

The CTD data show similar patterns for temperature (°C) and salinity (PSU) at toppled and standing platforms.

### **Video Data**

In total, 31 species representing 15 families were identified in this study (Appendix B). Of the 31 species, 27 were observed at standing platforms, 18 were observed at toppled platforms, and 14 were observed at both site types. At toppled platforms, four species were observed that were not at standing platforms. Additionally, 13 species that were observed at standing platforms were not observed at toppled platforms. Of the 31 species observed, accurate lengths could not be recorded for 13 of the species, most of which were observed only once during the 25 surveys.

Diversity indices varied by type, season, and layers, with highest averages of species richness, evenness, and diversity associated with the standing platforms (Table 3.2). Some species richness, evenness, and diversity averages were unable to be calculated at toppled platforms due to only one observation of that species (specifically in depth layers 1 and 3).

Species richness of fish communities was significantly different between seasons and post-hoc examination indicated the LSmean species richness was lowest during winter compared to fall and summer across both types (Table 3.3, Figure 3.5). Similarly, LSmean diversity was significantly lower during the winter compared to fall and summer (Table 3.3, Figure 3.5). Evenness was not significantly different between fish communities for any of the variables of type, layer, or season (Table 3.3).

Table 3.2. Average species richness ( $S$ ), Pielou's evenness ( $J'$ ), and Shannon Diversity ( $H'$ ) indices for type of structure, depth layer in water column, and season. No calculations available certain combinations of variables due to only one species observed.

| Type     | Layer | Season | $S$   | $J'$  | $H'$  |
|----------|-------|--------|-------|-------|-------|
| Standing | 1     | Fall   | 7.000 | 0.788 | 1.534 |
|          |       | Summer | 8.000 | 0.886 | 1.797 |
|          |       | Winter | 1.333 | 0.502 | 0.116 |
|          | 2     | Fall   | 9.000 | 0.813 | 1.786 |
|          |       | Summer | 7.000 | 0.869 | 1.616 |
|          |       | Winter | 2.000 | 0.882 | 0.527 |
|          | 3     | Fall   | 6.500 | 0.916 | 1.537 |
|          |       | Summer | 6.250 | 0.877 | 1.587 |
|          |       | Winter | 2.800 | 0.914 | 0.875 |
| Toppled  | 1     | Fall   | 5.000 | 0.889 | 1.432 |
|          |       | Summer | 4.500 | 0.815 | 1.149 |
|          |       | Winter | ----- | ----- | ----- |
|          | 2     | Fall   | 2.000 | 0.626 | 0.434 |
|          |       | Summer | 4.250 | 0.876 | 0.842 |
|          |       | Winter | 2.250 | 0.951 | 0.662 |
|          | 3     | Fall   | ----- | ----- | ----- |
|          |       | Summer | ----- | ----- | ----- |
|          |       | Winter | 2.750 | 0.816 | 0.799 |

Table 3.3. Analysis of variance (ANOVA) tests of type, depth layer, and season effects for each of the diversity indices. Asterisk (\*) denotes a significant effect ( $\alpha=0.05$ ).

| Index                    | Effect | F value | p-value  |
|--------------------------|--------|---------|----------|
| Species Richness ( $S$ ) | Type   | 1.82    | 0.2705   |
|                          | Layer  | 0.11    | 0.8995   |
|                          | Season | 12.92   | <0.0001* |
| Diversity ( $H'$ )       | Type   | 1.93    | 0.2590   |
|                          | Layer  | 1.06    | 0.3560   |
|                          | Season | 13.45   | <0.0001* |
| Evenness ( $J'$ )        | Type   | 0.31    | 0.6143   |
|                          | Layer  | 1.11    | 0.3423   |
|                          | Season | 0.65    | 0.5270   |



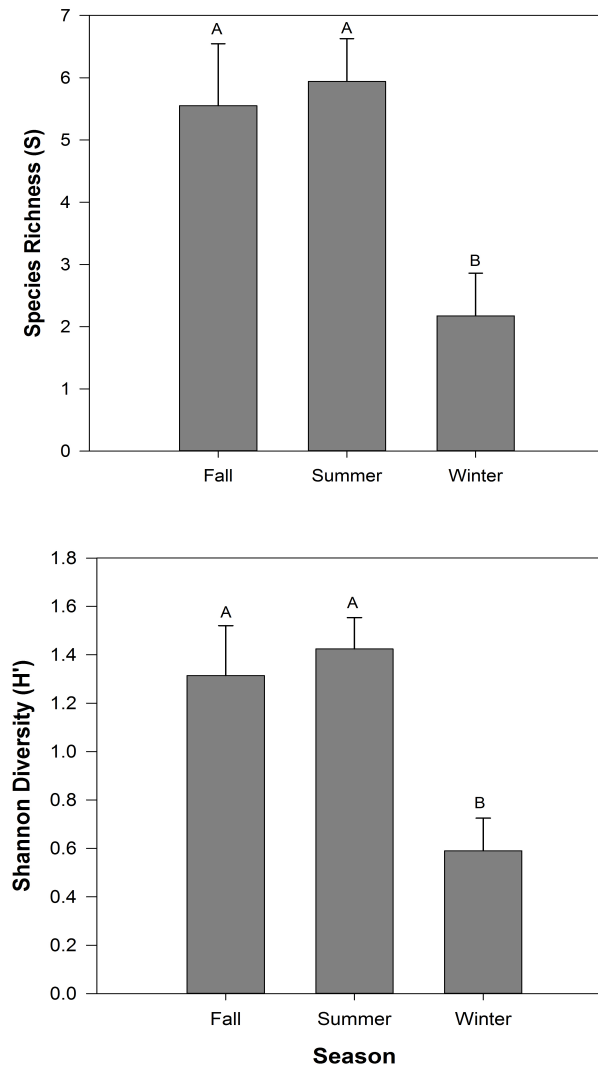


Figure 3.5. LSmean species richness ( $S$ ) and Shannon Diversity ( $H'$ ) by season. Error bars indicate standard error. Groups sharing a letter within a plot are not significantly different ( $\alpha=0.05$ ).

PERMANOVA indicated a significant two-way interaction effect between layer and season ( $p<0.005$ , Table 3.4) indicating that regardless of the type of structure, the fish communities are significantly different between at least one of the layer-season combinations. Fish communities also differ between type of structure, across all layers and seasons (Table 3.4).

Table 3.4. Permutational multivariate analysis of variance (PERMANOVA) source table comparing community structure of fishes within depth layers, seasons, and between type of structure. Asterisk (\*) denotes significant effect ( $\alpha=0.05$ ). df= degrees of freedom.

| Variable              | df | Pseudo-F | P(perm) | Unique perms |
|-----------------------|----|----------|---------|--------------|
| Type                  | 1  | 5.2959   | 0.001*  | 997          |
| Layer                 | 2  | 1.9241   | 0.019*  | 999          |
| Season                | 2  | 2.1815   | 0.006*  | 998          |
| Type x Layer          | 2  | 1.4562   | 0.123   | 998          |
| Type x Season         | 2  | 1.6148   | 0.064   | 998          |
| Layer x Season        | 4  | 1.8306   | 0.005*  | 998          |
| Type x Layer x Season | 1  | 1.3785   | 0.190   | 999          |

#### Layer by Season

Pair-wise tests were used to examine the specific combinations of layer and season that were significantly different. Statistically significant interactions occurred within depth layers 1, 2, 3, and in summer and winter seasons (Table 3.5). During fall, there were no statistically significant differences between fish communities within layers in the water column (Table 3.5). Within all three layers in the water column, fish communities were significantly different from each other between summer and winter. Fish communities were also significantly different from each other between fall and winter within layers 2 and 3 (Table 3.5).

SIMPER was used to assess which species were driving similarities within the significantly different layer and season combinations (Table 3.6). Results indicated that red snapper contributed most (48 to 80%) to the similarities within layers 2 and 3 in all three seasons (Table 3.6). Greater amberjack was the second most dominant species, contributing to the similarities in all three layers within summer, in layer 3 in the fall, and in layers 2 and 3 in the winter (Table 3.6). Additional species that contributed to the similarities of fish community structure between layers within seasons include little tunny (*Euthynnus alletteratus*), crevalle jack (*Caranx hippos*), blue runner, grey triggerfish

(*Balistes capriscus*), vermillion snapper (*Rhomboplites aurorubens*) and almaco jack (*Seriola rivirolana*).

Table 3.5. Pair-wise tests from PERMANOVA for depth layer by season; Asterisk (\*) denotes a significant effect ( $\alpha=0.05$ ), Summer= June, July, August, Fall= September, October, November, Winter= December, January, February, Layer 1= 0-30 m, Layer 2= 30-60 m, Layer 3= > 60 m.

| Factor  | Group          | t      | P(perm) | Unique Perms |
|---------|----------------|--------|---------|--------------|
| Layer 1 | Summer, Fall   | 1.1778 | 0.192   | 997          |
|         | Summer, Winter | 2.0266 | 0.001*  | 985          |
|         | Fall, Winter   | 1.8810 | 0.092   | 60           |
| Layer 2 | Summer, Fall   | 0.8214 | 0.651   | 898          |
|         | Summer, Winter | 1.7281 | 0.01*   | 999          |
|         | Fall, Winter   | 1.5785 | 0.051*  | 613          |
| Layer 3 | Summer, Fall   | 0.8621 | 0.664   | 15           |
|         | Summer, Winter | 1.5873 | 0.049*  | 993          |
|         | Fall, Winter   | 1.8535 | 0.018*  | 939          |
| Summer  | Layer 1,2      | 1.5624 | 0.018*  | 999          |
|         | Layer 1,3      | 2.6966 | 0.001*  | 997          |
|         | Layer 2,3      | 0.8146 | 0.752   | 990          |
| Fall    | Layer 1,2      | 1.7741 | 0.1131  | 15           |
|         | Layer 1,3      | 2.9331 | 0.061   | 15           |
|         | Layer 2,3      | 1.7747 | 0.246   | 12           |
| Winter  | Layer 1,2      | 1.2901 | 0.141   | 957          |
|         | Layer 1,3      | 1.9754 | 0.008*  | 998          |
|         | Layer 2,3      | 1.5641 | 0.042*  | 999          |

A Sim/SD ratio of a higher value indicates that the individual species consistently contributed to the within layer similarity across that season (Clarke and Warwick 2001). Greater amberjack consistently had a larger Sim/SD ratio compared to other species, and red snapper also contributed to similarities consistently (Table 3.6). Little tunny had the second largest Sim/SD ratio in layer 1 during summer; however, it had the lowest Sim/SD ratio in layer 2 during summer (Table 3.6).

SIMPER analyses indicated that red snapper was the dominant species driving the differences between all interactions except for Summer1 (i.e., layer 1 in summer) & Summer2, which was dominated by little tunny, and Summer1 & Winter1, which was dominated by blue runner. Red snapper contributed to the overall species dissimilarities between season and layer combinations by approximately 12 to 39% (Table 3.7). Red snapper most consistently contributed to these dissimilarities based upon the high Diss/DM values ranging from 1.11-2.16.

The same three species (little tunny, red snapper, and rainbow runner) contributed the most to the dissimilarities of Summer1 & Summer2, and Summer1 & Summer3 though absolute contributions differed (Table 3.7). Both Fall2 & Winter2 and Fall3 & Winter3 shared two species that contributed to dissimilarities (red snapper and greater amberjack; Table 3.7). Additionally, results indicated that the interaction of Summer3 & Winter3 had the smallest average dissimilarity value and Fall3 & Winter3 had the second smallest average dissimilarity value (Table 3.7).

#### Type of structure

SIMPER was used to assess which species were driving community similarities within structure type, which were significantly different from each other. Results indicated that red snapper, followed by greater amberjack, was the dominant species contributing to the similarities between type of structure (Table 3.8). Red snapper and greater amberjack contributed most (17 to 63%) to the similarities in community structure between both types of structure (Table 3.8). Based upon the Sim/SD ratio value of 0.93, red snapper at toppled structures consistently contributed to the within-type similarity among samples.

Table 3.6. Similarity percentages (SIMPER) results for the species that most contributed to the similarities between depth layer and season combinations (i.e. significant from Table 3.5). Shown are average abundance of important species within layers, the contribution to the average within layer by season similarity (Av. Sim), and the average similarity/standard deviation (Sim/SD) ratio within layer by season. The three most contributing species are shown. ## = SD could not be calculated. Summer= June, July, August; Fall= September, October, November; Winter= December, January, February. Layer 1= 0-30 m, Layer 2= 30-60 m, Layer 3= > 60 m.

| Factor | Group                        | Species                        | Av.Abund | Av.Sim | Sim/SD | Contrib% |
|--------|------------------------------|--------------------------------|----------|--------|--------|----------|
| Summer | Layer 1<br>Av.<br>Sim.=29.46 | <i>Euthynnus alletteratus</i>  | 5.38     | 7.74   | 0.73   | 25.34    |
|        |                              | <i>Seriola dumerili</i>        | 3.03     | 6.59   | 1.21   | 22.36    |
|        |                              | <i>Caranx hippos</i>           | 2.80     | 3.57   | 0.58   | 12.13    |
|        | Layer 2<br>Av.<br>Sim=29.21  | <i>Lutjanus campechanus</i>    | 4.06     | 14.16  | 0.88   | 48.46    |
|        |                              | <i>Seriola dumerili</i>        | 2.15     | 7.97   | 1.06   | 27.30    |
|        |                              | <i>Euthynnus alletteratus</i>  | 1.51     | 2.31   | 0.38   | 7.90     |
|        | Layer 3<br>Av.<br>Sim.=54.28 | <i>Lutjanus campechanus</i>    | 5.90     | 31.38  | 4.12   | 57.81    |
|        |                              | <i>Seriola dumerili</i>        | 1.87     | 11.49  | 5.78   | 21.17    |
|        |                              | <i>Rhomboplites aurorubens</i> | 2.53     | 5.25   | 0.71   | 9.67     |
| Fall   | Layer 2<br>Av.<br>Sim=24.87  | <i>Lutjanus campechanus</i>    | 4.81     | 20.12  | ##     | 80.93    |
|        |                              | <i>Balistes capriscus</i>      | 1.21     | 4.74   | ##     | 19.07    |
|        | Layer 3<br>Av.<br>Sim=56.14  | <i>Lutjanus campechanus</i>    | 7.79     | 35.35  | ##     | 62.96    |
|        |                              | <i>Seriola dumerili</i>        | 3.15     | 13.86  | ##     | 24.69    |
|        |                              | <i>Seriola rivoliana</i>       | 2.71     | 6.93   | ##     | 12.35    |
| Winter | Layer 1<br>Av. Sim=8.33      | <i>Caranx crysos</i>           | 5.39     | 8.33   | 0.58   | 100      |
|        | Layer 2<br>Av.<br>Sim=30.85  | <i>Lutjanus campechanus</i>    | 1.40     | 22.64  | 0.92   | 73.38    |
|        |                              | <i>Seriola dumerili</i>        | 1.07     | 8.21   | 0.68   | 26.62    |
|        | Layer 3<br>Av. Sim=59.14     | <i>Lutjanus campechanus</i>    | 3.96     | 22.64  | 0.92   | 73.38    |
|        |                              | <i>Seriola dumerili</i>        | 1.79     | 15.22  | 1.44   | 25.62    |

Table 3.7. Dissimilarity percentages (SIMPER) results of the species that most contributed to the dissimilarity between depth layer and season combinations that are significant from Table 3.3; shown are average abundance of important species within layers, and the average dissimilarity/standard deviation (Diss/SD) ratio within groups. Combinations of factors are denoted as the layer number, next to the season (ex: Summer1 is layer 1 in summer). The three most contributing species are shown. Summer= June, July, August; Fall= September, October, November; Winter= December, January, February. Layer 1= 0-30 m; Layer 2= 30-60 m; Layer 3= > 60 m.

| Group             | Species                        | Av. Abund | Av. Abund | Diss/SD | Contrib% |
|-------------------|--------------------------------|-----------|-----------|---------|----------|
| Summer1 & Summer2 |                                | Summer1   | Summer2   |         |          |
| Av. Dis=76.70     | <i>Euthynnus alletteratus</i>  | 5.38      | 1.51      | 1.11    | 16.23    |
|                   | <i>Lutjanus campechanus</i>    | 1.00      | 4.06      | 1.11    | 12.59    |
|                   | <i>Elagatis bipinnulata</i>    | 4.50      | 0.11      | 0.83    | 10.42    |
| Summer1 & Summer3 |                                | Summer1   | Summer3   |         |          |
| Av. Dis=79.75     | <i>Lutjanus campechanus</i>    | 1.00      | 5.90      | 1.62    | 16.09    |
|                   | <i>Euthynnus alletteratus</i>  | 5.38      | 0.25      | 1.15    | 14.20    |
|                   | <i>Elagatis bipinnulata</i>    | 4.50      | 0.56      | 0.93    | 10.17    |
| Summer1 & Winter1 |                                | Summer1   | Winter1   |         |          |
| Av. Dis=90.32     | <i>Caranx crysos</i>           | 2.84      | 5.39      | 1.01    | 19.32    |
|                   | <i>Euthynnus alletteratus</i>  | 5.38      | 0.00      | 1.06    | 18.80    |
|                   | <i>Seriola dumerili</i>        | 3.03      | 0.58      | 0.75    | 11.81    |
| Summer2 & Winter2 |                                | Summer2   | Winter2   |         |          |
| Av. Dis=79.54     | <i>Lutjanus campechanus</i>    | 4.06      | 1.40      | 1.40    | 25.35    |
|                   | <i>Euthynnus alletteratus</i>  | 1.51      | 0.00      | 0.52    | 14.06    |
|                   | <i>Seriola dumerili</i>        | 2.15      | 1.07      | 1.11    | 12.89    |
| Summer3 & Winter3 |                                | Summer3   | Winter3   |         |          |
| Av. Dis=53.13     | <i>Lutjanus campechanus</i>    | 5.90      | 3.96      | 1.44    | 20.09    |
|                   | <i>Rhomboplites aurorubens</i> | 2.53      | 0.00      | 1.26    | 18.29    |
|                   | <i>Caranx hippos</i>           | 0.79      | 0.57      | 0.69    | 9.60     |

(Table 3.7 continued)

| Group             | Species                     | Av. Abund | Av. Abund | Diss/SD | Contrib% |
|-------------------|-----------------------------|-----------|-----------|---------|----------|
| Fall2 & Winter2   |                             | Fall2     | Winter2   |         |          |
| Av. Dis=78.72     | <i>Lutjanus campechanus</i> | 4.81      | 1.40      | 1.19    | 29.45    |
|                   | <i>Caranx crysos</i>        | 7.40      | 0.00      | 0.96    | 23.58    |
|                   | <i>Seriola dumerili</i>     | 1.58      | 1.07      | 1.01    | 9.27     |
| Fall3 & Winter3   |                             | Fall3     | Winter3   |         |          |
| Av. Dis=57.76     | <i>Lutjanus campechanus</i> | 7.79      | 3.96      | 1.99    | 25.54    |
|                   | <i>Seriola rivoliana</i>    | 2.71      | 0.22      | 1.35    | 18.23    |
|                   | <i>Seriola dumerili</i>     | 3.15      | 1.79      | 1.74    | 12.33    |
| Winter1 & Winter3 |                             | Winter1   | Winter3   |         |          |
| Av. Dis=86.22     | <i>Lutjanus campechanus</i> | 0.00      | 3.96      | 2.16    | 39.95    |
|                   | <i>Caranx crysos</i>        | 5.39      | 0.74      | 1.15    | 34.98    |
|                   | <i>Seriola dumerili</i>     | 0.58      | 1.79      | 1.27    | 13.30    |
| Winter3 & Winter2 |                             | Winter3   | Winter2   |         |          |
| Av. Dis=58.89     | <i>Lutjanus campechanus</i> | 3.96      | 1.40      | 1.96    | 39.96    |
|                   | <i>Seriola dumerili</i>     | 1.79      | 1.07      | 1.27    | 21.10    |
|                   | <i>Caranx latus</i>         | 0.31      | 0.86      | 0.44    | 10.83    |

SIMPER was also used to determine which individual species were driving the dissimilarities in fish communities between types. Results indicated that red snapper and little tunny contributed greatly to the dissimilarities in community structure of fishes between type of structure (Table 3.9).

Table 3.8. Similarity percentages (SIMPER) results of the species that most contributed to the within-type similarities for the two types, toppled and standing. Shown are average abundance of important species within layers, the contribution to the average within type similarity (Av. Sim), and the average of similarity/standard deviation (Sim/SD) ratio within type. The three highest contributing species are shown.

| Group                | Species                       | Av.Abund | Av.Sim | Sim/SD | Contrib% |
|----------------------|-------------------------------|----------|--------|--------|----------|
| Toppled              | <i>Lutjanus campechanus</i>   | 3.07     | 19.85  | 0.93   | 63.27    |
| Av. Similarity=31.38 | <i>Seriola dumerili</i>       | 1.91     | 5.64   | 0.67   | 17.96    |
|                      | <i>Euthynnus alletteratus</i> | 2.25     | 3.85   | 0.37   | 12.27    |
| Standing             | <i>Lutjanus campechanus</i>   | 2.79     | 8.84   | 0.64   | 34.24    |
| Av. Similarity=25.81 | <i>Seriola dumerili</i>       | 1.83     | 8.95   | 0.90   | 31.19    |
|                      | <i>Caranx hippos</i>          | 1.63     | 2.46   | 0.50   | 9.52     |

Table 3.9. Dissimilarity percentages (SIMPER) of the species that most contributed to the within-type dissimilarities for the two types, toppled and standing. Shown are average abundances of the species within the groups, and the average dissimilarity/standard deviation (Diss/SD) ratio between type. The three highest contributing species are shown.

| Group              | Species                       | Av.Abund        | Av.Abund         | Diss/SD | Contrib% |
|--------------------|-------------------------------|-----------------|------------------|---------|----------|
| Toppled & Standing |                               | Group           | Group            |         |          |
| Av. Dis =76.46     | <i>Lutjanus campechanus</i>   | Toppled<br>3.07 | Standing<br>2.79 | 1.07    | 18.51    |
|                    | <i>Euthynnus alletteratus</i> | 2.25            | 1.09             | 0.69    | 12.69    |
|                    | <i>Seriola dumerili</i>       | 1.91            | 1.83             | 0.94    | 11.87    |

Results from the distance based linear model (DistLM) implied that both environmental parameters, temperature (°C) and salinity (PSU), explained a significant amount of the biotic data individually ( $p=0.0001$ ,  $p=0.0004$ , respectively). However, the variables were closely correlated ( $r=-0.72$ ), so using just one variable to explain the biotic data was sufficient (Table 3.10).



Table 3.10. Distance based linear model (DistLM) to show environmental parameters temperature (°C) and salinity (PSU) to explain the biotic data. Models were calculated by selecting for Akaike information criteria (AICc) with a best selection procedure.

| Variable              | Pseudo-F | P     |
|-----------------------|----------|-------|
| Temperature (Celsius) | 7.1495   | 0.001 |
| Salinity (PSU)        | 3.5588   | 0.004 |

## **Biomass Apportioning**

### Layer by Season

Red snapper and greater amberjack exhibited the highest relative MaxN and biomass for both layer by season and structure type groups. Results of relative MaxN and biomass calculations indicated that red snapper was the highest contributing species to the relative biomass within layer 3 in every season (Table 3.11). Little tunny was a large contributor to the relative MaxN and biomass within layer 1 during both summer and fall, however this species was absent during winter in layer 1 (Table 3.11).

Greater amberjack is present in every layer and season, except for during fall in depth layer 1, and its relative MaxN contribution is less than half of its relative biomass contribution. In winter layer 1, only one individual of each species present was observed, so relative contributions (both MaxN and biomass) from species are equal (Table 3.11). Results indicate that bar jack (*Caranx ruber*), crevalle jack, blue runner, almaco jack, and vermillion snapper are also important contributors to community structure based on the relative MaxN and biomass (Table 3.11). These results can also be viewed graphically with bubble plots in Appendix B.

### Type of Structure

A total of 18 species observed at the standing platforms were accurately measured and 11 species observed at the toppled platforms were accurately measured.

Table 3.11. Apportionment of the species contributing the most to percent biomass (Biomass %), ranked by highest to lowest contribution within season and depth layer, combined over type of site. Four highest ranked species are shown; Note that MaxN % is shown strictly for comparison. Summer= June, July, August; Fall= September, October, November; Winter= December, January, February. Layer 1= 0-30 m; Layer 2= 30-60 m; Layer 3= >60 m.

| Season | Layer | Species                        | Biomass% |         | MaxN% |         |
|--------|-------|--------------------------------|----------|---------|-------|---------|
|        |       |                                | Avg.     | St. Dev | Avg.  | St. Dev |
| Summer | 1     | <i>Euthynnus alletteratus</i>  | 32.1     | 36.8    | 30.7  | 34.3    |
|        |       | <i>Seriola dumerili</i>        | 23.2     | 29.1    | 11.6  | 27.0    |
|        |       | <i>Caranx hippos</i>           | 17.6     | 25.5    | 10.3  | 14.8    |
|        |       | <i>Caranx latus</i>            | 10.3     | 15.5    | 8.9   | 16.5    |
|        | 2     | <i>Seriola dumerili</i>        | 30.7     | 22.6    | 11.9  | 33.1    |
|        |       | <i>Lutjanus campechanus</i>    | 27.4     | 24.3    | 40.1  | 34.4    |
|        |       | <i>Euthynnus alletteratus</i>  | 13.9     | 32.6    | 13.4  | 32.7    |
|        |       | <i>Caranx latus</i>            | 13.1     | 33.1    | 11.9  | 33.1    |
|        | 3     | <i>Lutjanus campechanus</i>    | 59.5     | 22.2    | 61.6  | 18.4    |
|        |       | <i>Seriola dumerili</i>        | 16.5     | 3.8     | 6.7   | 2.7     |
|        |       | <i>Caranx hippos</i>           | 11.8     | 23.7    | 7.8   | 15.6    |
|        |       | <i>Rhomboplites aurorubens</i> | 4.9      | 5.9     | 13.9  | 16.2    |
| Fall   | 1     | <i>Caranx latus</i>            | 48.8     | 44.2    | 49.6  | 44.1    |
|        |       | <i>Euthynnus alletteratus</i>  | 29.8     | 51.6    | 26.2  | 45.4    |
|        |       | <i>Caranx ruber</i>            | 9.2      | 15.9    | 3.4   | 5.8     |
|        |       | <i>Lutjanus campechanus</i>    | 2.9      | 5.0     | 4.3   | 7.3     |
|        | 2     | <i>Lutjanus campechanus</i>    | 53.6     | 62.7    | 51.4  | 64.0    |
|        |       | <i>Caranx hippos</i>           | 15.9     | 22.5    | 3.5   | 5.1     |
|        |       | <i>Caranx crysos</i>           | 13.9     | 19.7    | 37.3  | 52.8    |
|        |       | <i>Seriola dumerili</i>        | 11.3     | 16.1    | 1.7   | 2.4     |
|        | 3     | <i>Lutjanus campechanus</i>    | 60.2     | 9.9     | 65.8  | 1.2     |
|        |       | <i>Seriola dumerili</i>        | 27.5     | 8.9     | 11.3  | 5.2     |
|        |       | <i>Seriola rivoliana</i>       | 7.3      | 8.1     | 10.9  | 12.8    |
|        |       | <i>Rhomboplites aurorubens</i> | 3.3      | 4.6     | 10.4  | 14.8    |
| Winter | 1     | <i>Caranx crysos</i>           | 66.2     | 57.3    | 66.1  | 57.3    |
|        |       | <i>Seriola dumerili</i>        | 33.3     | 57.7    | 33.3  | 57.7    |
|        |       | <i>Seriola rivoliana</i>       | 0.5      | 0.8     | 0.5   | 0.8     |

(Table 3.11 continued)

| Season | Layer | Species                     | Biomass% |          | MaxN% |          |
|--------|-------|-----------------------------|----------|----------|-------|----------|
|        |       |                             | Avg.     | St. Dev. | Avg.  | St. Dev. |
| Winter | 2     | <i>Lutjanus campechanus</i> | 40.2     | 40.3     | 52.6  | 40.1     |
|        |       | <i>Seriola dumerili</i>     | 36.8     | 39.8     | 23.3  | 28.2     |
|        |       | <i>Caranx latus</i>         | 9.5      | 26.9     | 11.8  | 33.2     |
|        |       | <i>Seriola rivoliana</i>    | 8.6      | 24.3     | 8.3   | 23.5     |
|        | 3     | <i>Lutjanus campechanus</i> | 56.6     | 21.8     | 71.1  | 21.9     |
|        |       | <i>Seriola dumerili</i>     | 35.0     | 25.2     | 19.3  | 18.9     |
|        |       | <i>Caranx hippos</i>        | 5.9      | 12.8     | 2.9   | 6.7      |
|        |       | <i>Caranx crysos</i>        | 1.3      | 4.2      | 5.0   | 15.1     |

Of the measured species, all observed at toppled sites were also observed at the standing sites. The eight species that were observed at standing platforms, but absent at toppled platforms, included blue runner, crevalle jack, barracuda, bar jack, scamp, yellow jack (*Carangoides bartholomaei*), and dusky shark (*Carcharhinus obscurus*).

Relative MaxN and biomass calculations indicated that there are more species (7) that contribute to 90% of the fishes observed at standing platforms as compared to those at toppled platforms (Table 3.12). The four species contributing 90% at the toppled platforms were also species that helped to contribute 90% at the standing platforms, which included red snapper, greater amberjack, crevalle jack, and horse-eye jack (*Caranx latus*).

At both types of structures, red snapper was the most dominant species in relation to relative biomass and MaxN (Table 3.12). Greater amberjack contributed the second most to relative biomass at both structure types. However, both blue runner and little tunny, at

Table 3.12. Apportionment of the species contributing the most to relative biomass (Biomass %), ranked by highest to lowest contribution within type, across all depth layers and seasons. All species are shown. Note that MaxN % is shown strictly for comparison. All Biomass % calculated for toppled platforms is from the average lengths from video at standing platforms. Asterisk (\*) denotes the species that contribute to approximately 90% of fishes observed at site type.

| Group    | Species            | Scientific name                 | Biomass% |         | MaxN% |         |
|----------|--------------------|---------------------------------|----------|---------|-------|---------|
|          |                    |                                 | Avg.     | St. Dev | Avg.  | St. Dev |
| Standing | Red snapper        | <i>Lutjanus campechanus</i>     | 27.5     | 30.4    | 33.7* | 34.7    |
|          | Greater amberjack  | <i>Seriola dumerili</i>         | 26.9     | 2.73    | 14.9* | 23.2    |
|          | Creville jack      | <i>Caranx hippos</i>            | 11.3     | 18.6    | 6.7*  | 11.3    |
|          | Horse-eye jack     | <i>Caranx latus</i>             | 10.8     | 23.2    | 10.5* | 24.4    |
|          | Blue runner        | <i>Caranx crysos</i>            | 9.7      | 25.6    | 15.3* | 31.0    |
|          | Little tunny       | <i>Euthynnus alletteratus</i>   | 4.2      | 14.1    | 3.3*  | 11.7    |
|          | Rainbow runner     | <i>Elagatis bipinnulata</i>     | 2.3      | 7.2     | 5.0*  | 12.6    |
|          | Almaco jack        | <i>Seriola rivoliana</i>        | 1.8      | 3.7     | 3.2   | 6.8     |
|          | Barracuda          | <i>Sphyraena barracuda</i>      | 1.3      | 3.6     | 0.5   | 1.2     |
|          | Bermuda chub       | <i>Kyphosus sectatrix</i>       | 1.1      | 3.1     | 2.4   | 6.1     |
|          | Bar jack           | <i>Caranx ruber</i>             | 1.0      | 4.9     | 0.4   | 1.8     |
|          | Vermillion snapper | <i>Rhomboplites aurorubens</i>  | 0.8      | 2.7     | 2.7   | 7.7     |
|          | Silky shark        | <i>Carcharhinus falciformis</i> | 0.4      | 1.1     | 0.09  | 0.3     |
|          | Gray triggerfish   | <i>Balistes capriscus</i>       | 0.3      | 0.6     | 0.4   | 0.9     |
|          | Gray snapper       | <i>Lutjanus griseus</i>         | 0.2      | 0.7     | 0.4   | 1.1     |
|          | Scamp              | <i>Mycteroperca phenax</i>      | 0.2      | 0.7     | 0.1   | 0.03    |
|          | Yellow jack        | <i>Carangoides bartholomaei</i> | 0.05     | 0.02    | 0.1   | 0.04    |
|          | Dusky shark        | <i>Carcharhinus obscurus</i>    | 0.05     | 0.02    | 0.02  | 0.01    |

(Table 3.12 continued)

| Group   | Species            | Scientific name                 | Biomass% |         | MaxN% |         |
|---------|--------------------|---------------------------------|----------|---------|-------|---------|
|         |                    |                                 | Avg.     | St. Dev | Avg.  | St. Dev |
| Toppled | Red snapper        | <i>Lutjanus campechanus</i>     | 38.5     | 35.7    | 48.8* | 38.6    |
|         | Greater amberjack  | <i>Seriola dumerili</i>         | 27.8     | 30.8    | 14.9* | 23.7    |
|         | Little tunny       | <i>Euthynnus alletteratus</i>   | 22.6     | 38.2    | 22.5* | 36.5    |
|         | Horse-eye jack     | <i>Caranx latus</i>             | 5.5      | 23.5    | 5.5*  | 23.5    |
|         | Almaco jack        | <i>Seriola rivoliana</i>        | 4.3      | 16.1    | 4.9   | 15.6    |
|         | Gray triggerfish   | <i>Balistes capriscus</i>       | 0.4      | 0.6     | 0.8   | 1.5     |
|         | Silky shark        | <i>Carcharhinus falciformus</i> | 0.3      | 1.1     | 0.1   | 0.4     |
|         | Vermillion snapper | <i>Rhomboplites aurorubens</i>  | 0.2      | 0.7     | 0.6   | 2.7     |
|         | Rainbow runner     | <i>Elagatis bipinnulata</i>     | 0.1      | 0.5     | 0.5   | 1.9     |
|         | Bermuda chub       | <i>Kyphosus sectatrix</i>       | 0.04     | 0.2     | 0.2   | 0.4     |
|         | Gray snapper       | <i>Lutjanus griseus</i>         | 0.01     | 0.007   | 0.2   | 0.01    |

standing and toppled platforms, respectively, had a greater relative MaxN compared to greater amberjack (Table 3.12). These results can also be viewed graphically with bubble plots in Appendix B.

#### Average Abundance

Red snapper was the most common species in layer 3 and the second most common in layer 2 during all three seasons (Table 3.13).

Table 3.13. Average MaxN count of species by depth layer and season. Four highest ranked species are shown; Species ranked by average from highest to lowest. Summer= June, July, August; Fall= September, October, November; Winter= December, January, February. Layer 1= 0-30 m; Layer 2= 30-60 m; Layer 3= >60 m.

| Season | Layer | Species                        | MaxN count |          |
|--------|-------|--------------------------------|------------|----------|
|        |       |                                | Avg.       | St. Dev. |
| Summer | 1     | <i>Elagatis bipinnulata</i>    | 65.4       | 161.4    |
|        |       | <i>Euthynnus alletteratus</i>  | 61.2       | 89.9     |
|        |       | <i>Caranx crysos</i>           | 22.9       | 36.4     |
|        |       | <i>Caranx latus</i>            | 15.6       | 21.6     |
|        | 2     | <i>Caranx crysos</i>           | 27.3       | 82.0     |
|        |       | <i>Lutjanus campechanus</i>    | 23.8       | 21.2     |
|        |       | <i>Euthynnus alletteratus</i>  | 8.3        | 20.6     |
|        |       | <i>Seriola dumerili</i>        | 6.3        | 5.4      |
|        | 3     | <i>Lutjanus campechanus</i>    | 36.7       | 16.7     |
|        |       | <i>Rhomboplites aurorubens</i> | 11         | 13.6     |
|        |       | <i>Seriola dumerili</i>        | 3.5        | 0.6      |
|        |       | <i>Caranx hippos</i>           | 2.5        | 5        |
| Fall   | 1     | <i>Caranx latus</i>            | 134.7      | 127.3    |
|        |       | <i>Elagatis bipinnulata</i>    | 31.6       | 51.4     |
|        |       | <i>Caranx ruber</i>            | 13.3       | 23.1     |
|        |       | <i>Euthynnus alletteratus</i>  | 12.3       | 21.3     |

(Table 3.13 continued)

| Season | Layer | Species                        | MaxN count |          |
|--------|-------|--------------------------------|------------|----------|
|        |       |                                | Avg.       | St. Dev. |
| Fall   | 2     | <i>Caranx crysos</i>           | 109.5      | 154.8    |
|        |       | <i>Lutjanus campechanus</i>    | 23.5       | 7.7      |
|        |       | <i>Caranx hippos</i>           | 10.5       | 14.8     |
|        |       | <i>Kyphosis sectatrix</i>      | 10.5       | 14.8     |
|        | 3     | <i>Lutjanus campechanus</i>    | 61         | 12.7     |
|        |       | <i>Rhomboplites aurorubens</i> | 11         | 15.5     |
|        |       | <i>Seriola dumerili</i>        | 10         | 2.3      |
|        |       | <i>Seriola rivoliana</i>       | 9          | 9.8      |
| Winter | 1     | <i>Caranx crysos</i>           | 66.3       | 110.6    |
|        |       | <i>Seriola dumerili</i>        | 1          | 1.7      |
|        |       | <i>Seriola rivoliana</i>       | 1          | 1.7      |
|        | 2     | <i>Caranx latus</i>            | 5.8        | 16.6     |
|        |       | <i>Lutjanus campechanus</i>    | 3.3        | 3.8      |
|        |       | <i>Seriola dumerili</i>        | 2.3        | 3.4      |
|        |       | <i>Caranx hippos</i>           | 0.5        | 1.4      |
|        | 3     | <i>Lutjanus campechanus</i>    | 17.3       | 9.7      |
|        |       | <i>Seriola dumerili</i>        | 5          | 7.8      |
|        |       | <i>Caranx crysos</i>           | 4.9        | 14.6     |
|        |       | <i>Caranx hippos</i>           | 1.4        | 2.9      |

Red snapper was not among the four frequently counted species in layer 1 during any season. In layer 1 during summer and fall, rainbow runner (*Elagatis bipinnulata*), horse-eye jack, and little tunny were dominant. However, in layer 1 during the winter, blue runner, greater amberjack, and almaco jack were frequently observed (Table 3.13). When specifically looking at the most abundantly observed species at standing platforms, schooling fishes such as blue runner, rainbow runner, and horse-eye jack were prevalent, whereas red snapper and little tunny were less frequently observed (Table 3.14). At toppled platforms, little tunny, red snapper, and greater amberjack were the most

abundantly observed species, and there were fewer rainbow runner observed (Table 3.14).

Table 3.14. Average MaxN count of species by type of structure, across all depth layers and seasons. All species are shown, ranked by average from highest to lowest.

| Group    | Species            | Scientific name                 | MaxN count |         |
|----------|--------------------|---------------------------------|------------|---------|
|          |                    |                                 | Avg.       | St. Dev |
| Standing | Blue runner        | <i>Caranx crysos</i>            | 28.4       | 65.8    |
|          | Rainbow runner     | <i>Elagatis bipinnulata</i>     | 22.3       | 91.6    |
|          | Horse-eye jack     | <i>Caranx latus</i>             | 19.3       | 51.2    |
|          | Red snapper        | <i>Lutjanus campechanus</i>     | 15.5       | 20.8    |
|          | Little tunny       | <i>Euthynnus alletteratus</i>   | 12.5       | 49.3    |
|          | Bermuda chub       | <i>Kyphosus sectarix</i>        | 5.6        | 14.6    |
|          | Crevalle jack      | <i>Caranx hippos</i>            | 4.9        | 9.7     |
|          | Greater amberjack  | <i>Seriola dumerili</i>         | 4.5        | 4.2     |
|          | Vermillion snapper | <i>Rhomboplites aurorubens</i>  | 2.1        | 6.5     |
|          | Almaco jack        | <i>Seriola rivoliana</i>        | 1.8        | 3.5     |
|          | Bar jack           | <i>Caranx ruber</i>             | 1.4        | 7.0     |
|          | Gray snapper       | <i>Lutjanus griseus</i>         | 0.5        | 1.6     |
|          | Gray triggerfish   | <i>Balistes capriscus</i>       | 0.5        | 1.0     |
|          | Barracuda          | <i>Sphyraena barracuda</i>      | 0.4        | 0.7     |
|          | Yellow jack        | <i>Carangoides bartholomaei</i> | 0.3        | 1.3     |
|          | Silky shark        | <i>Carcharhinus falciformus</i> | 0.2        | 0.8     |
|          | Scamp              | <i>Mycteroperca phenax</i>      | 0.09       | 0.3     |
|          | Dusky shark        | <i>Carcharhinus obscurus</i>    | 0.03       | .2      |
| Toppled  | Little tunny       | <i>Euthynnus alletteratus</i>   | 17.4       | 40.0    |
|          | Red snapper        | <i>Lutjanus campechanus</i>     | 14.0       | 13.8    |
|          | Greater amberjack  | <i>Seriola dumerili</i>         | 9.5        | 21.9    |
|          | Rainbow runner     | <i>Elagatis bipinnulata</i>     | 1.0        | 4.2     |
|          | Almaco jack        | <i>Seriola rivoliana</i>        | 0.4        | 0.8     |
|          | Grey triggerfish   | <i>Balistes capriscus</i>       | 0.4        | 0.8     |
|          | Vermillion snapper | <i>Rhomboplites aurorubens</i>  | 0.4        | 1.6     |
|          | Horse-eye jack     | <i>Caranx latus</i>             | 0.1        | 0.5     |
|          | Bermuda chub       | <i>Kyphosis sectarix</i>        | 0.1        | 0.3     |
|          | Silky shark        | <i>Carcharhinus falciformus</i> | 0.1        | 0.3     |
|          | Gray snapper       | <i>Lutjanus griseus</i>         | 0.05       | 0.2     |



## **Discussion**

### **Video Data**

This study aimed to determine the differences in the fish community structure between standing and toppled oil and gas platforms in the northern GOM in relation to variables such as depth of the water column, type of site, and season. Historically, video-based surveys have included drop cameras, carousels, manned diver cameras, and remotely operated vehicles (ROVs) and provide an opportunity to sample a variety of species and sizes of fishes (Somerton and Gledhill 2004, Cappo et al. 2006, Pacunski et al. 2008, Murphy and Jenkins 2010, Bryan et al. 2013, Ajemian et al. 2015). Large artificial reef structures like standing and toppled oil and gas platforms are less sampled than their smaller counterparts like reef balls or pyramids because they have additional challenges associated with sampling efforts. Depth constraints and physical complexities have limited non-SCUBA video-based surveys that focus on the community structure of fishes associated with artificial reefs (Rooker et al. 1997, Stanley C.A. 2000, Wilson et al. 2003, Patterson III et al. 2009, 2013, Pradella et al. 2014, Ajemian et al. 2015). Sampling large artificial reefs such as standing and toppled platforms using BRUVs and BRUSVs allowed for efficient sampling and rapid data collection.

This study found significant differences between fish community structure at standing and toppled platforms in relation to the depth layer, and season. Even though each variable was statistically significant on its own, the only significant interaction effect was layer by season. These results are similar to Rooker et al. (1997) and Wilson et al. (2000, 2003), which concluded that the associated fish community structure at artificial reefs differed, based on the depth of the water column and season.

Diversity indices of species richness ( $S$ ), Pielou's Evenness ( $J'$ ), and Shannon Diversity ( $H'$ ) were used to describe the community structure in relation to the type of structure, layer in the water column, and season. All three indices were higher at standing platforms compared to toppled platforms, though averages of these indices were much lower compared to those observed from previous studies at other artificial reefs (Rooker et al. 1997, Wilson et al. 2003, Langland 2015). Lower species richness and diversity observed during winter could be due to the colder temperatures and well-mixed layers. Migratory fishes, which could be a prey-source for larger fishes, may leave the area or be less active during winter, also leading to lower species richness and diversity. A recent study observed significantly lower LSmeans for richness and diversity at an artificial reef when compared to natural reefs (Langland 2015). The average species richness and diversity in my study were similar to Langland's (2015) results for artificial reefs using the same measures, suggesting that lower species diversity and richness at artificial reefs is not uncommon. These patterns indicate that there may be greater habitat complexity and greater area available on the natural reefs, subsequently leading to higher species richness and diversity.

Fish communities differed between layers in the water column at both structure types. This finding is consistent with a study by Rooker et al. (1997) who determined that abundance and diversity of reef fishes were lowest in the upper water column and increased with depth. Patterson et al. (2013) determined significant differences in the fish communities between types of habitat in addition to the depth of the water column by using ROV surveys at artificial and natural reefs. Red snapper and greater amberjack drove the majority of the similarities and differences between these fish communities,

which is similar to results reported by Wilson et al. (2003) and Patterson et al. (2013). These studies observed high densities and abundances of red snapper and greater amberjack at various artificial reefs. These structures have significant fishing value, as commercial and recreational fishers target the standing and toppled platforms where these fishes aggregate, which causes concern for these fish populations (Cowan et al. 2011). Because I also observed high amounts of red snapper at these structures, I believe that in a perfect world, setting aside a specific amount of these structures as “no-take” zones while decreasing the total allowable catch from the other areas may help protect biomass of fishes. These structures have changed the distribution of fishes and have most likely made them more vulnerable, therefore contributing to overfishing.

Cold temperatures during winter may explain the differences in fish communities in different depth layers observed. There was a large difference in the average abundance of fishes during winter compared to fall and summer, in addition to less individual species observed. It is well known that during colder weather, fishes metabolism tends to decrease, which leads to less movement and feeding (Adams and Breck 1990). Fewer fishes may have been attracted to the bait on the camera arrays due to this reason, resulting in less fishes observed during the winter months; the fishes might have been there but I did not see them. Moreover, the colder temperatures could have caused smaller fishes to move to warmer waters, resulting in less prey-resource availability for larger fishes such as red snapper.

Using BRUVs and BRUSVs for this study allowed for accurate data to be collected from each survey. Baited systems can be helpful to attract the fishes around the sampling area, however, it has also been shown to cause differences in the fish

assemblage structure and composition when compared to non-baited systems (Dorman et al. 2012). Inherent bias that is associated with video arrays (e.g. larger fishes were observed while cryptic fishes most likely missed, effects of bait on fish behavior and possible following of camera) may have affected my results. However, sampling methods and post-processing of the data were conducted while keeping these biases in mind to minimize errors. For example, the first 30 seconds of video data were excluded from analysis due to the immediate attraction of fishes to the bait and to avoid re-counting fishes that followed the bait. A consistent survey method in addition to a large sample size (25 camera surveys, three depth stops during each survey) helped to minimize other possible sources of sampling bias such as a small sampling size, the number of species of fish that can be effectively counted simultaneously, and the method of counting itself (Harvey et al. 2002).

Future surveys using BRUV and BRUSV arrays at additional standing and toppled oil and gas platforms would allow for a better understanding of the fish community structure in addition to the effects on these fishes on a larger temporal and spatial scale. In addition, our sampling only occurred at one side of the standing platform (downcurrent) or one area of the toppled platform per survey based on the currents and the feasibility of keeping the R/V steady. For a more complete survey to determine the community structure around other parts of the structures, multiple BRUV/BRUSV deployments on different sides of the structure would be necessary to gain a better idea of the entire community and account for bias that may have occurred due to sampling only one side of the structure. With the continued rapid assessment of fish communities around these platforms, detail is sacrificed, but sample size increases. Ecologically and

economically important species tend to aggregate around these structures, making it important to understand how fishes are using these types of structure. The LARP and fisheries management in the GOM should attempt to better understand how fishes are using these structures due to the affinity for structure exhibited by numerous species.

### **Biomass Apportioning**

Coupling non-destructive acoustic and video surveys can be rapidly performed and together can determine what species are contributing to the biomass determined from the acoustic surveys. A number of studies have explored fish biomass, density, abundance and community structure at artificial reefs, including oil and gas platforms, by combining visual and acoustic survey methods (Gledhill et al. 1996, Stanley and Wilson 1996, 1997, 1998, 2000, 2003, Rooker et al. 1997, Patterson et al. 2009). While useful information regarding fish community structure and biomass has been determined, some previous studies have lacked to address the comparison of standing and toppled oil and gas platforms using video and acoustic survey methods. I was able to calculate relative biomass contributions for individual species from the video data that ultimately allowed me to determine the relative contribution of an individual species to the acoustic biomass.

Red snapper are a prevalent species at both standing and toppled platforms which was determined from the video data analysis and I concluded that red snapper also had a high relative MaxN and biomass. Previous studies have also characterized red snapper as a dominant species around artificial reefs in the northern GOM (Rooker et al. 1997, Wilson et al. 2003, Patterson et al. 2013). Although this species is commonly observed around these structures, studies have evidence that they do not feed or reproduce around these structures, rather they move away to do so (Simonsen et al. 2014). It has also been

established that despite structure removal in the GOM over the last decade, the spawning stock biomass of red snapper has not declined (Cowan, Jr., J.H.,<sup>7</sup> personal comm.). Because of this, I question the role of these structures as important habitat for this particular species.

While red snapper contributed greatly to relative MaxN and biomass, when using average count as a metric, pelagic planktivores such as blue runner and little tunny were the most abundant species at standing and toppled platforms, respectively, which is consistent with results from Wilson et al. (2003). Numerically, these fishes are plentiful when observed, schooling by the hundreds. However, these schools are patchy and when measured, they were much smaller (approximately 30 cm FL) than red snapper and greater amberjack. Therefore, the average abundance of this species does not necessarily reflect what one may expect to see on any given camera drop. Additionally, because blue runner were not observed at toppled platforms during any of our surveys, it leads me to believe that the additional vertical relief of the standing platforms may be providing an additional source of food, such as plankton, or refuge (structure) that the toppled structure is lacking.

Seven species were observed (and measured) at standing platforms that were not observed at toppled platforms, six of which are considered high trophic level fishes (Table 3.12). Simonsen et al. (2014) found that red snapper diet at standing platforms consisted of primarily fishes, squid, and shrimp, while at toppled platforms greater amounts of crabs, shrimps, and other crustaceans were consumed. These results lead me to believe that higher trophic level fishes occupy the area around standing platforms more

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<sup>7</sup> Cowan, Jr., J.H. 2015. Louisiana State University, Department of Oceanography and Coastal Sciences.

often than toppled due to the greater amounts of prey fishes associated with these structures. However, even though species composition differs between these standing and toppled platforms, it is unlikely that site-specific prey alone is the main contributor to these differences (Simonsen et al. 2014). Furthermore, the only true structure-related species in this group is scamp, which leads me to believe that these structures may be redistributing fishes that would normally be observed in the open-ocean.

The relative biomass results from toppled platforms should be interpreted with caution. This study used a BRUV array to sample toppled platforms, which decreased the risk of equipment loss, but also meant that lengths of fishes could not be determined from the surveys. As a result, fish lengths at toppled sites were based on fishes from the standing platforms. However, the toppled platforms were relatively close (2.5-9 km) to the standing platforms, so assuming lengths were comparable was my best estimate. Future surveys using this array should include two lasers by each camera set at a known distance from each other, to obtain relative fishes measurements from the toppled structures.

Relating the biomass results derived from the video data to the acoustic biomass allowed me to better understand what fishes were most likely the main contributors to the acoustic biomass observed. It is clear that even though the acoustic biomass remains consistent throughout all seasons, different species comprise the acoustic biomass observed depending on season and depth layer. Because of these results, I believe that integrating the knowledge from these two studies is important, as it provides a further understanding of the community structure. Even though the methods used to ground-truth the acoustic data with the video data have inherent bias (e.g. calculation errors,

assumptions in lengths for toppled platform species, limited sampling sites), the results from these surveys allow for a better understanding of what species are contributing to the biomass around the structures.

It is obvious that combining the data provides more information than it would when considered separately. This study demonstrated that a rapid, accurate, non-destructive assessment of fish biomass and community structure at standing and toppled platforms is possible. Continuing rapid assessments of these structures throughout the GOM would help determine any patterns in fish communities and biomass distributions that may be similar with varying physical locations including further west, east, offshore, and inshore. By using non-destructive sampling methods around oil and gas platforms to continuously monitor fish populations, these artificial reefs can be better understood in relation to biomass and population sizes.

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## CHAPTER 4: HYDROACOUSTIC AND VIDEO SURVEYS: IMPLICATIONS AND USES FOR NON-DESTRUCTIVE FISHERIES MONITORING

### Introduction

The worldwide demand for marine resources is constantly increasing which has led to various ecological and economic pressures. Overfishing has become one of the major threats altering the structure, functioning, and health of marine ecosystems resources (Jennings and Kaiser 1998, Pauly et al. 1998, Jackson et al. 2001, Banaru et al. 2010, Cresson et al. 2014). Therefore, effective monitoring methods in addition to communication and policies are needed to limit fishing pressure on marine ecosystems and to ensure the future sustainability of marine resources.

Fisheries and fish communities in the Gulf of Mexico (GOM) vary greatly, as there are many diverse species that occupy a variety of habitats including natural hard bottom reefs, soft-bottom substrates, and artificial reefs (such as standing or toppled oil and gas platforms). Habitat quality is important to fishes and is considered important if it provides prey resources that subsequently allow increases in available energy for reproduction, growth, and foraging (Mittelbach 2002, Schwartzkopf 2014). Specifically, red snapper (*Lutjanus campechanus*) is an important species in the GOM, as it is ecologically dominant in addition to having an economic importance for fisheries. Schwartzkopf (2014) determined that the natural reefs on the Louisiana continental shelf provided higher quality habitat for red snapper than artificial reefs.

To better manage ecologically and economically important species such as red snapper, it is necessary to understand their life history and where they aggregate and spawn. Many species, in addition to red snapper, have been observed congregating around artificial reefs, as these structures provide hard substrate where little exists



naturally (Gallaway et al. 1981, Bohnsack and Sutherland 1985, Pickering and Whitmarsh 1997, Stanley and Wilson 1997, 2000, Boswell et al. 2010). Scientists have hypothesized that artificial reefs have the potential to diversify habitat, modify the communities of organisms in the surrounding areas, concentrate existing resources, and increase fishery resources (Bohnsack et al. 1991, Seaman and Sprague 1991, Scarborough-Bull et al. 2008).

Standing and toppled oil and gas platforms serve as artificial reefs in the GOM, especially in the waters off the coast of Louisiana. These structures provide habitat for economically important fishes and programs are in place to manage these structures due to their ability to attract and aggregate pelagic and demersal fishes. The Louisiana Artificial Reef Program (LARP), established in 1986, helps to manage the standing and toppled oil and gas platforms off the coast of Louisiana and monitor the fish communities associated with them.

Non-destructive survey techniques such as hydroacoustic (acoustic) and video surveys allow for rapid, accurate assessments of biomass, density, community structure, and size of fishes associated with various habitats such as natural and artificial reefs, deep and shallow sea marine environments, marine protected areas, temperate, and tropical habitats (Ellis and DeMartini 1994, Rooker et al. 1997, Priede and Bagley 2001, Jones et al. 2003, Willis et al. 2003, Wilson et al. 2003, Cappelletti et al. 2004, Harvey et al. 2007, Wells 2007, Boswell et al. 2010).

This chapter provides overviews of both acoustic and video survey methods in addition to discussing the practicality of using these surveys for future non-destructive fisheries monitoring, specifically in the GOM. The feasibility of using these survey

methods to gain accurate data that can be used for stock and habitat assessments is important to determine. Policy and management groups that regulate fish stocks and habitats (i.e., standing/toppled oil and gas platforms) they are associated with should be using data from reliable and accurate sources. It is imperative that the science and policy communication gap is bridged, and that reliable data is used to accurately influence regulations and management of ecologically and economically valuable species in the GOM.

### **Hydroacoustic Surveys**

Acoustics are a well-established and standard methodology to collect data for commercial fisheries resource assessment and management (Thorne 1983). Acoustic surveys have quickly developed into a common non-lethal procedure for estimating quantities, species, spatial patterns, abundance, and biomass of economically and ecologically important fish stocks in lakes, rivers, and oceans worldwide. Recognized as a prospective tool in fishery operations and research before World War II, acoustic technology has continued to evolve with the growing demands of research (Koslow 2009). During the 1990s, acoustic sampling for east coast fisheries had been limited to a smaller number of stocks (Brown et al. 1992). However, the new and improved calibrated instruments are now considered a standard qualitative and quantitative tool for fishery research and stock assessment (Koslow 2009). The northwest coastal fisheries in the U.S. have utilized acoustics for data collection since 1997, when surveys were initially conducted by the Alaska Fisheries Science Center (AFSC) triennially along the U.S. and Canadian Pacific Coast. The Northeast Fisheries Science Center (NEFSC) initiated a fisheries acoustic survey program shortly after in 1998 to provide fisheries-

independent estimates of absolute and relative abundances of pelagic fish stocks (Northeast Fisheries Science Center National Oceanic and Atmosphere Administration 2013).

Historically, the southeast U.S. fisheries have focused intensively on both shrimp (*Penaeus* spp.) and menhaden (*Brevoortia* spp.). These species have short life spans and are heavily fished throughout their respective seasons (Brown et al. 1992). Typically, trawl surveys are used to estimate shrimp recruitment, abundance, and size, which are necessary to manage the stocks and regulate the shrimp season. Trawl sampling became the primary method for shrimp stock data collection because of its relative easy and low cost. Menhaden were primarily sampled with air surveys, thus avoiding the need for development and use of acoustic surveys. The northeastern U.S. has traditionally used trawl sampling as well, due to the mixed-size and species of catches. Consequently, development of acoustic survey techniques was given low priority in other regions, such as the GOM region, due to the success of using trawl surveys to collect data. (Brown et al. 1992).

When the use of acoustic methods was discussed as a viable option for assessment of stocks such as bluefin tuna (*Thunnus thynnus*), the general consensus based on reviews from the Southeast Fisheries Science Center (SEFSC), was that acoustic sampling would be of little use due to the thought at the time that this method would work best for sampling reef-loving species that congregate around structure (Brown et al. 1992).

As acoustic technology advanced and the application of this tool was found to have utility, the northwest U.S. fisheries research groups adopted acoustic surveys to use as a management tool. The University of Washington Marine Acoustics Program has utilized

acoustic surveys and applied these results to fishery management for many years. For instance, members of this program collectively participated in over 1,000 surveys between 1969 and 1980 (Thorne 1983). This program paved the way for agencies such as Washington State Department of Fisheries, Alaska Department of Fish and Game, International Pacific Salmon Fisheries Commission, National Marine Fisheries Service Northwest and AFSC to continue the routine application of acoustic techniques in fisheries within their respective management areas (Thorne 1983). Since the late 1990s, acoustics surveys and methods have been frequently integrated into data collection methodologies and have proven to be a valuable technique for fisheries managers (Kracker 2007).

There are several advantages and disadvantages associated with using acoustics for research and monitoring of fish distributions and habitats. These specific advantages and unique limitations to acoustic techniques should be taken into consideration when applying acoustics to fisheries management. Some advantages of using acoustics compared to other tools include, but are not limited to, independence from fishery catch statistics, short time scale, low operational costs, low levels of variability, and capability for absolute population estimates (Thorne 1983). For example, acoustic surveys are used to provide harvest quotas for herring stocks in the Gulf of Georgia of Washington and Alaska. Surveys are conducted on a weekly basis during the open period of the fishery. An acoustic survey is conducted the first night of the week, the results are analyzed the next day, a quota is set for the week, and the fishery opens the next day (Trumble 1978, Thorne 1983). Costs are kept low due to the high sampling power and efficiency of the acoustic equipment. The initial costs of equipment are high, but are still less costly than

constant exploratory fishing costs (Thorne 1983). Additionally, low levels of variability due to the high sampling capabilities of acoustic surveys ensure accurate measurements, leading to reliable fisheries stock management decisions.

Although there are many advantages to using acoustic surveys for fisheries management and stock assessment, there are several disadvantages. These limitations include insufficient species identification, little to no sampling capabilities near surface or bottom, relative high complexity (i.e. processing and analysis), large initial investment, lack of biological samples (when using acoustics alone), and potential bias associated with target-strength calculations (Thorne 1983, Kracker 2007).

Additionally, the presence or absence and shape of a swim bladder can affect the backscatter strength return that is then used to make biomass estimates (Kracker 2007). Even though it is difficult to definitively identify fish species and secondary information is needed to confirm the species and size distribution of fishes detected in the water column, acoustics are still a useful tool for estimating biomass, number of organisms in the water column, and providing relative information on size distribution of targets (Kracker 2007). However, these small influences on backscattering strength return can have large effects on the inferences drawn about the species composition and size distribution of the acoustic targets (De Robertis et al. 2010). Consequently, acoustic measurements at multiple frequencies have been developed to account for the characteristics of various target species (De Robertis et al. 2010).

Acoustics has been the basis of numerous fishery research studies and subsequently used in ecosystem-based fishery management (EBFM). According to Koslow (2009), EBFM is centered on a “need to assess broader fishery effects on the ecosystem, i.e. on

the predators, competitors, and prey of the exploited species, as well as on by catch species and the essential habitat.” Based on this idea, the function of exploited species is best understood when assessed in relation to the ecosystem where they live as a whole (Koslow 2009). Environmental conditions are constantly changing and therefore are important to consider when examining fish stocks, yet oceanic environmental conditions are complex and difficult to express in a stock assessment (Koslow 2009). Dynamic models of ecosystem interactions are best modeled with spatial and temporal data that can relate to the species present and to environmental factors. Acoustic methods are useful because of several of the advantages acoustics provide including minimal avoidance reactions, high spatial resolution, and the ability to collect large quantities of data over a relatively short period of time (Koslow 2009).

Models that incorporate acoustic data have been proven to be helpful for fisheries management and stock assessment purposes. For example, an Ecopath model (Butler et al. 2002) integrated the results from various diet studies, then calculated biomass estimates of the orange-roughy population from both acoustic and egg count surveys. This study lead to the conclusion that acoustics are an important tool in modeling pelagic fish communities. The acoustic data provided additional information that trawl catch information simply could not supply. It also allowed for managers to solve a conflict between fishing-industry and environmental stakeholders in the creation of one of the world’s first deep-water marine reserves south of Tasmania (Koslow 2009). Case studies such as this one exhibit various uses of acoustic data to influence decisions in management and future studies.

Using acoustic surveys, this study was able to accurately estimate acoustic biomass of fishes around standing and toppled platforms, measured as the mean-volume backscattering strength (MVBS, dB), a proxy for biomass. Acoustic biomass was highest close to the structure and declined greatly as distance from the structure increased, leveling off to a background-noise level at 100 m distance from the structure. Overall, type of structure (standing/toppled), distance from structure, and depth layers of the water column all had a significant influence on the amount of acoustic biomass around the structure.

Based on the data and conclusions that were drawn from this study, I believe acoustics to be a useful tool for sampling fish populations in various habitats, specifically in the GOM. This non-destructive survey method can collect large quantities of data in a short period of time, which can then be used to guide management. However, the large sample size (i.e., 98,203) of data collected during acoustic surveys can be nearly impossible to understand in a biological context. Future acoustic surveys should aim to account for this bias by either modeling a random subsample of data collected, or using other statistical approach's to account for a large sample size.

Even though the data may have limited biological implications due to large sample sizes, which causes even the smallest differences to be statistically significant, I do believe that these surveys can be conducted across the GOM in conjunction with artificial reef programs. Acoustic surveys can be a useful tool to gain information about fish biomass, distributions, and densities in the GOM and worldwide. It is likely that future acoustic surveys if carefully planned, could contribute greatly to the ecologically and economically important assessments and management of fish stocks of the GOM (Thorne

et al. 1989, Luecke and Wurtsbaugh 1993, Gledhill et al. 1996, Stanley and Wilson 1996, 1998, 2003, Soldal et al. 2002, Boswell et al. 2010).

### **Video Surveys**

Visual survey techniques are commonly used both to determine the abundance of marine organisms and to characterize marine fish communities (Gledhill et al. 1996, Harvey et al. 2004, Wells 2007). Since visual surveys were first used in the 1950s to assess the abundance of reef fishes, the technology and methods have improved and have a wide range of applications (Odum and Odum 1955, Harvey et al. 2004). They are a non-lethal, robust sampling tool useful for assessing species identification, community composition, distribution, relative abundances, and sizes of marine fishes (Ellis and DeMartini 1994, Harvey et al. 2004, Wells 2007).

Historically, visual or video surveys included a variety of methods such as drop cameras, carousels, manned diver cameras, and remote operated vehicles (ROVs) (Somerton and Gledhill 2004, Cappelletti et al. 2006, Pacunski et al. 2008, Murphy and Jenkins 2010, Bryan et al. 2013, Ajemian et al. 2015). Recently, baited-remote underwater videos (BRUV) arrays are a commonly used technology to survey fish communities over a variety of different habitats. These systems are capable of being used to accurately assess fish communities, are non-destructive, less disturbing to habitats and organisms, and provide more accurate data compared to SCUBA diver surveys (Harvey et al. 2004). Furthermore, BRUVs have been used extensively within the GOM as part of SEAMAP video surveys conducted by the National Marine fisheries Service (NMFS) since 1991 (Gledhill et al. 1996).



Many studies conduct video surveys to use as either supplementary data or as the main source of data. The data collected from video surveys in conjunction with appropriate statistical analyses of the inferred data, has been used to estimate the abundance of species and their age distributions (Mellody 2014). The National Academy of Sciences reported that the National Oceanographic Atmospheric Administration (NOAA) Fisheries is exploring an automatic means of fish stock assessments, such as video data collection in addition to traditional methods such as trawl surveys (Mellody 2014). Currently, inadequate data collection tools and techniques largely limit accuracy and efficiency of fisheries stock assessments. Video collection methods offer the potential to improve efficiency and reduce sources of variability that are associated with other data collection methods (Mellody 2014).

Recently, ROV surveys were used to quantify benthic organisms and fishes at the Flower Garden Banks National Marine Sanctuary (FGBNMS), a large area along the continental shelf in the northwest GOM home to dozens of reefs and banks. The reefs and banks in this area are well studied due to its inclusion in a national marine sanctuary, and video surveys are commonly used to gather further information about the fish communities. Video surveys are appropriate for estimating fish abundances when factors such as physical complexities of structures or habitat exist in addition to depth constraints (Bortone et al. 1986, Greene and Alevizon 1989, Wells 2007). By using video surveys at the FGBNMS, information about fish densities, community structure, and habitat type and coverage can be reliably determined. This information provides a biological baseline that can be used to further manage both fish stocks and the national marine sanctuary.

There are many advantages and disadvantages associated with using video surveys for research and monitoring of fish distributions and habitats. Some advantages include, but are not limited to, independence from fishery catch statistics, low costs, precise measurements, and non-destructive measurements of fish lengths (Harvey et al. 2003, 2007). For example, accurate information on the length frequency or biomass of wild and cultured fish populations is essential to the management of harvest (Harvey et al. 2002b). Stereo-video camera systems can supply accurate length measurements, which can be used to better determine fishes condition, size variations and growth rates, consequently contributing positively to management decisions for these fish populations (Petrell et al. 1997, Harvey et al. 2002b).

Although there are many advantages to using video surveys for fisheries data collection and management, there are some disadvantages. These limitations include inherent errors from calibration, light limitations and water visibility, dynamic physical complexities of the habitat, difficulty in species identification, fish movement that results in double counting, and avoidance/under-representation of smaller, cryptic species (Sale and Douglas 1981, Bohnsack and Bannerot 1986, Harvey et al. 2004, Wells and Cowan 2007). Despite the limitations of video surveys, they are still more accurate than data collected with other visual survey techniques such as SCUBA diver surveys, due to their less disruptive nature and their ability to be deployed and retrieved rapidly from depth (Harvey et al. 2004, Wells and Cowan 2007).

Video surveys have become a common research method in recent years; this is reflected in the use of video camera arrays used by the NOAA SEFSC. To conduct reef surveys, equipment such as stationary video cameras and stereo cameras are deployed to

determine relative abundance of fishes, fish lengths, and percent cover of habitat. The stationary camera array has reduced error associated with data collected due to the lack of lights and sound, in addition to little to no movement associated with it. Using this array is advantageous compared to ROVs, which can be noisy and include movements that could affect fish behavior. However, this array is more spatially restricted compared to an ROV sampling method due to its static positioning (National Marine Fisheries Service 2015). The data collected from video surveys can provide spatially explicit demographic and biological information, which is linked to habitat parameters for ecologically and economically valuable reef fishes. Subsequently, the information gathered can enhance fishery ecosystem model development and management (National Marine Fisheries Service 2015).

By using video surveys in my study, I was able to both accurately characterize the fish communities associated with these structures and determine relative biomass estimates of individual species. The standing structures hosted a more diverse variety of species compared to the toppled platforms, but red snapper was a dominant species around both types of structures. Overall, type of structure (standing/toppled), depth layers of the water column, and season all significantly influenced the fish communities around the structures.

Based on the data and conclusions that were drawn from this study, I believe that video surveys are a useful tool for sampling fish populations in various habitats, specifically in the GOM. This non-invasive survey method can collect accurate, non-invasive data on fish communities and lengths, which can then be used to determine additional amount of information helpful for guiding management. Because this tool is

valuable when sampling artificial reefs such as standing and toppled platforms, it seems likely that these surveys can be efficiently conducted across the GOM. Video surveys can be a useful tool to gain information about fish species composition, densities, and abundances in the GOM and worldwide. It is likely that video surveys will contribute to ecologically and economically important fisheries stock and habitat assessments and management in the GOM (Ellis and DeMartini 1994, Rooker et al. 1997, Harvey et al. 2002a, Wilson et al. 2003, Cappo et al. 2006, Wells 2007, Wells and Cowan 2007, Merritt et al. 2011, Misa et al. 2013, Whitmarsh et al. 2014, Ajemian et al. 2015).

Finally, this project utilized video methods to determine which species of fishes contributed to the biomass observed in the acoustic surveys, thus encompassing a more comprehensive view of fish communities associated with these structures. By using video data to ground truth the acoustic data, I was able to identify the species likely to be contributing to the biomass observed from the acoustic surveys based on length-weight relationships that allowed for calculations of relative biomass of individual species. Additionally, the video data provided additional information that the acoustic data did not supply: even though the biomass around both standing and toppled structures is consistent throughout the year, the fish species that are contributing to that biomass changes depending on the season and layer in the water column. This is important information to agencies when interested in the management of both the fishes around the structures and the structures themselves.

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## GENERAL SUMMARY AND CONCLUSIONS

The objective of my research was to determine the spatial biomass distribution and community structure of fishes around standing and toppled oil and gas platforms located in the Eugene Island Oil and Gas Block Lease approximately 130 km off the coast of Louisiana in the northern Gulf of Mexico (GOM). The Louisiana Artificial Reef Program (LARP) is interested in using rapid assessments to estimate the biomass and community structure of fishes associated with standing and toppled platforms in order to better understand the efficacy of these structures as viable habitat for ecologically and economically important species such as red snapper in the GOM. This research also aimed to determine whether rapid assessment surveys could be successfully conducted around these structures using non-destructive hydroacoustic (acoustic) and video survey techniques. My goal was to use each of these methods individually then combine the results to better determine which species based on relative biomass (%) from the video were the main contributors to the acoustic biomass.

Chapter 2 described the spatial biomass distribution of nekton and fishes around three standing and two toppled oil and gas platforms using mobile acoustic surveys. Spatial biomass distribution was measured using mean-volume backscattering strength (MVBS), a proxy for biomass. The fish MVBS was significantly different in relation to distance from structure, depth (layers), and structure type (standing/toppled). Fish MVBS was highest close to the structure and declined greatly as distance from structure increased, leveling off to a constant level after 100 m from structure. The observed decline of fish MVBS with increased distance from the structure is consistent with previous studies by Gerlotto et al. 1989, Stanley and Wilson 1996, 1997, 1998, 2003,

Boswell et al. 2007, 2010, Shipley and Cowan 2010. In my opinion, these areas are important habitat for fishes because they provide some type of service, whether that is habitat and/or prey resources, and therefore should be considered as a management tool.

Acoustic biomass was highest in depth layer 3 (>60 m) compared to layers 1 (0-30 m) and 2 (30-60 m), which are the opposite results of a previous study by Simonsen (2013) conducted at the same study sites in 2010. The location of fishes in the water column are likely a result of physical changes, such as an unusually cold winter (2013), that were not efficiently monitored due to the limits of sampling. Future work may benefit from sampling more frequently and calculating other variables such as density and target strength (TS). Since these structures attract economically and ecologically important species, they should be monitored to the best of our ability using non-destructive techniques in order to have a minimal impact on the current populations.

Chapter 3 examined the fish community structure using a baited remote underwater stereo-video (BRUSV) array and a baited remote underwater video (BRUV) array at standing and toppled oil and gas platforms, respectively. Both types of video arrays allowed me to accurately characterize the fish communities associated with these structures and BRUVs allowed me to further determine relative biomass estimates of the fish species using length-weight relationships. The standing structures hosted a more diverse variety of species compared to the toppled structures, but red snapper and greater amberjack were dominant species around both types of structures, which is consistent with previous studies (Wilson et al. 2003, Patterson et al. 2013). Overall, type of structure, depth layers, and season all significantly influenced the fish communities around the structures. These structures have significant fishing value, as commercial and

recreational fishers target the standing and toppled platforms where these fishes aggregate, which causes concern for these fish populations (Cowan et al. 2011).

A previous study that sampled four of the same five sites focused solely on collecting acoustic data and diet analyses (Simonsen 2013). There was no video data, therefore this study lacked the opportunity to ground-truth their acoustic results. Therefore, a major goal of this study was to combine the acoustic and video survey data to determine a more comprehensive idea of how many and what species are distributed around these structures.

The most notable finding was that by combining the results from these two methodologies, it was easy to observe that even though acoustic biomass remained consistent throughout all seasons, different species comprised the acoustic biomass depending on the season and depth layers. Whether these patterns are specific to these sites or widespread throughout the GOM is of interest to me, and I believe can be answered using these rapid assessment surveys.

The standing and toppled oil and gas platforms sampled in this study are only a miniscule part of the largest *de facto* artificial reef complex in the world (Dauterive 2000). The artificial structures in the GOM serve as important and complex habitat for many reef and reef-associated fishes in an otherwise primarily flat area with natural hard and soft bottom areas (Bright 1977, Parker et al. 1983, Wells 2007). Ecologically and economically valuable fishes utilize both the natural and artificial habitats, making them equally important to manage. The information provided by this study about when and what fishes are aggregating around these structures can be of use to the LARP when they make future management decisions about the placement of structures. These artificial

reef structures may be providing habitats to fishes and organisms, which would make these artificial reef structures a valuable resource to manage. However, if these structures are simply attracting reef-associated fishes from nearby habitats, they could make these fishes more vulnerable to exploitation (Cowan et al. 2011).

It is clear from this study that non-destructive survey methods such as acoustic and video techniques can be efficiently used for rapid assessments of fish community structure and biomass around standing and toppled oil and gas platforms in the GOM. When rapidly assessing these structures, detail is sacrificed; however, the benefit of a large sample size over a short period of time may outweigh the loss of fine-scale detail. This study's methodology may be the future continual monitoring method for these structures and used by artificial reef programs. Based on the data and conclusions that were drawn from this study, I believe that video and acoustic surveys are a useful tool for sampling fish populations in various habitats, specifically in the GOM.

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## APPENDIX A: CHAPTER 2 SUPPLEMENTARY DATA

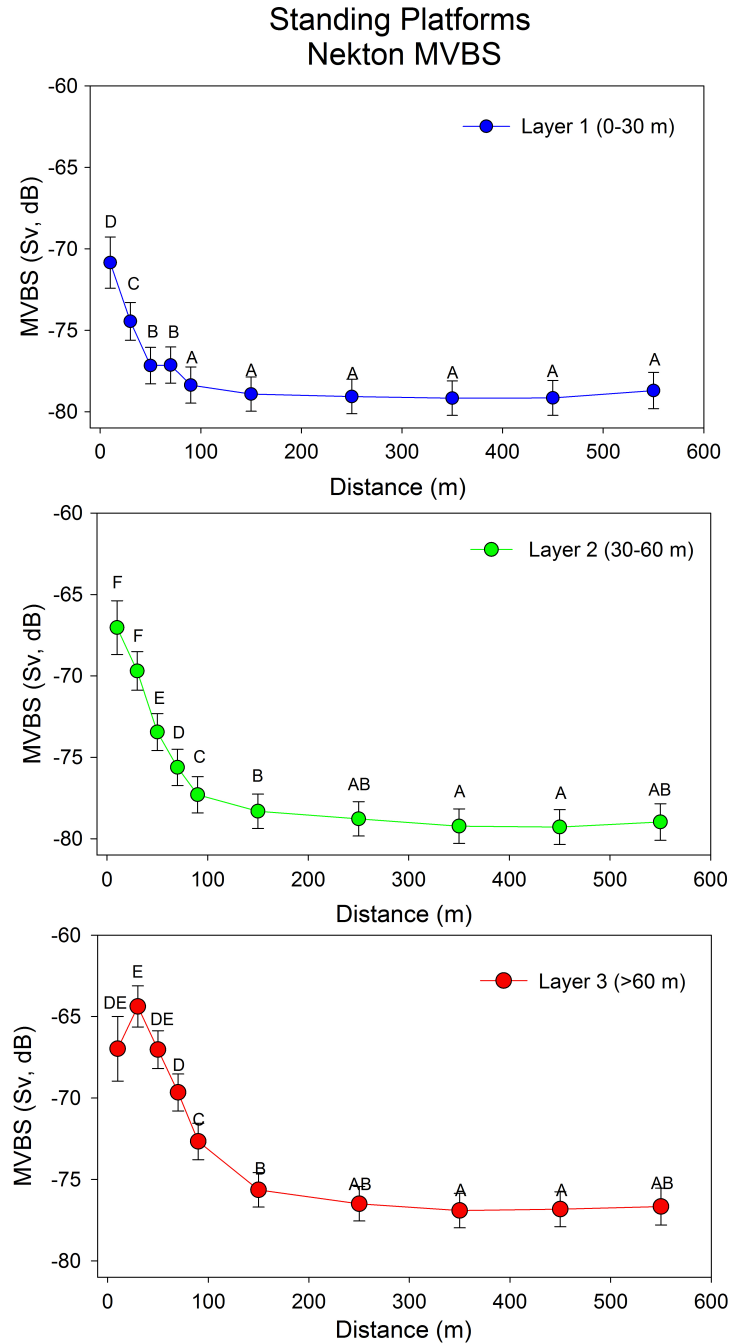


Figure A.1. LSmean-volume backscatter (MVBS, Sv) of nekton data set versus distance from the structure at standing platforms in the northern Gulf of Mexico. Each layer in the water column is graphed separately; depth layer 1 (0-30 m) (blue), depth layer 2 (30-60 m) (green), depth layer 3 (>60 m) (red). Groups that have different letters are significantly different from each other.

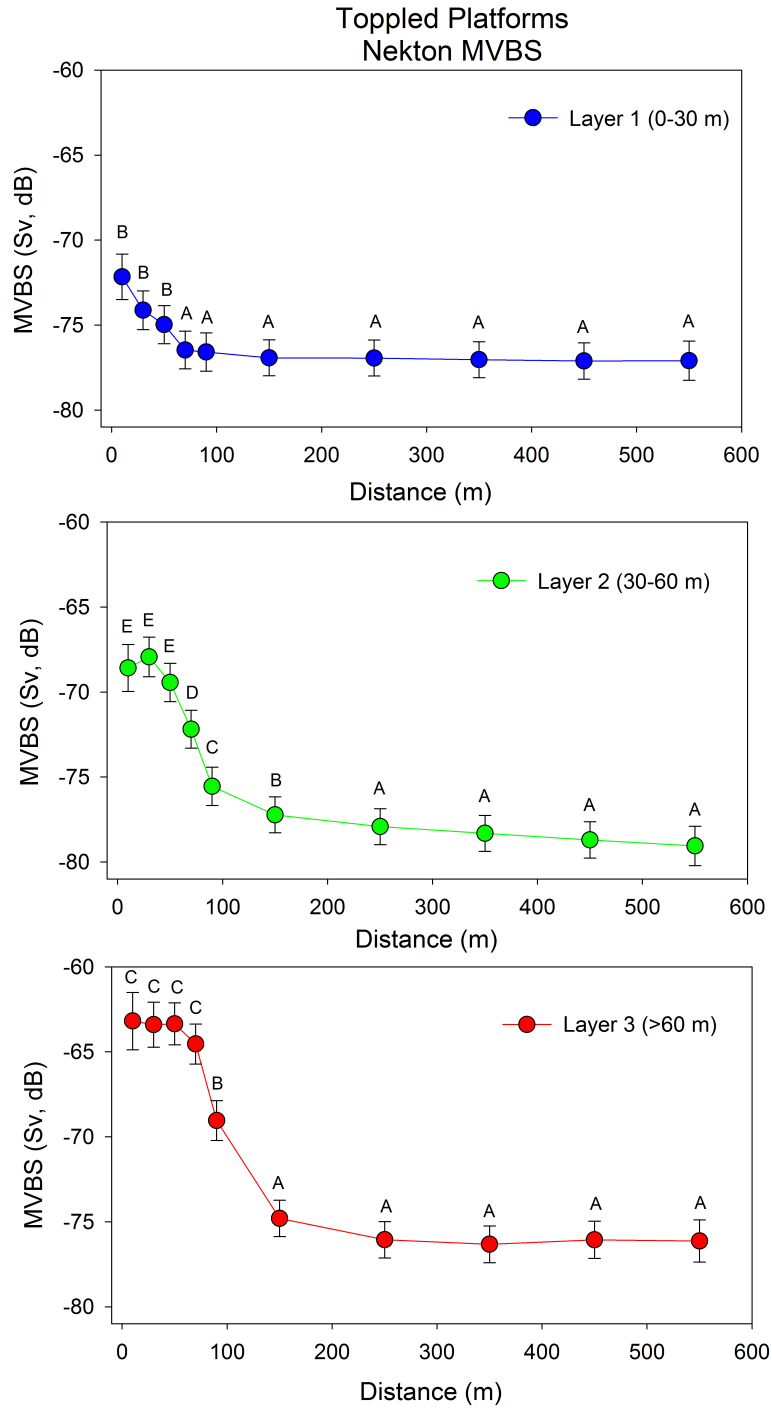


Figure A.2. LSmean-volume backscatter (MVBS, Sv) of nekton data set versus distance from the structure at toppled platforms in the northern Gulf of Mexico. Each layer in the water column is graphed separately; depth layer 1 (0-30 m) (blue), depth layer 2 (30-60 m) (green), depth layer 3 (>60 m) (red). Groups that have different letters are significantly different from each other.

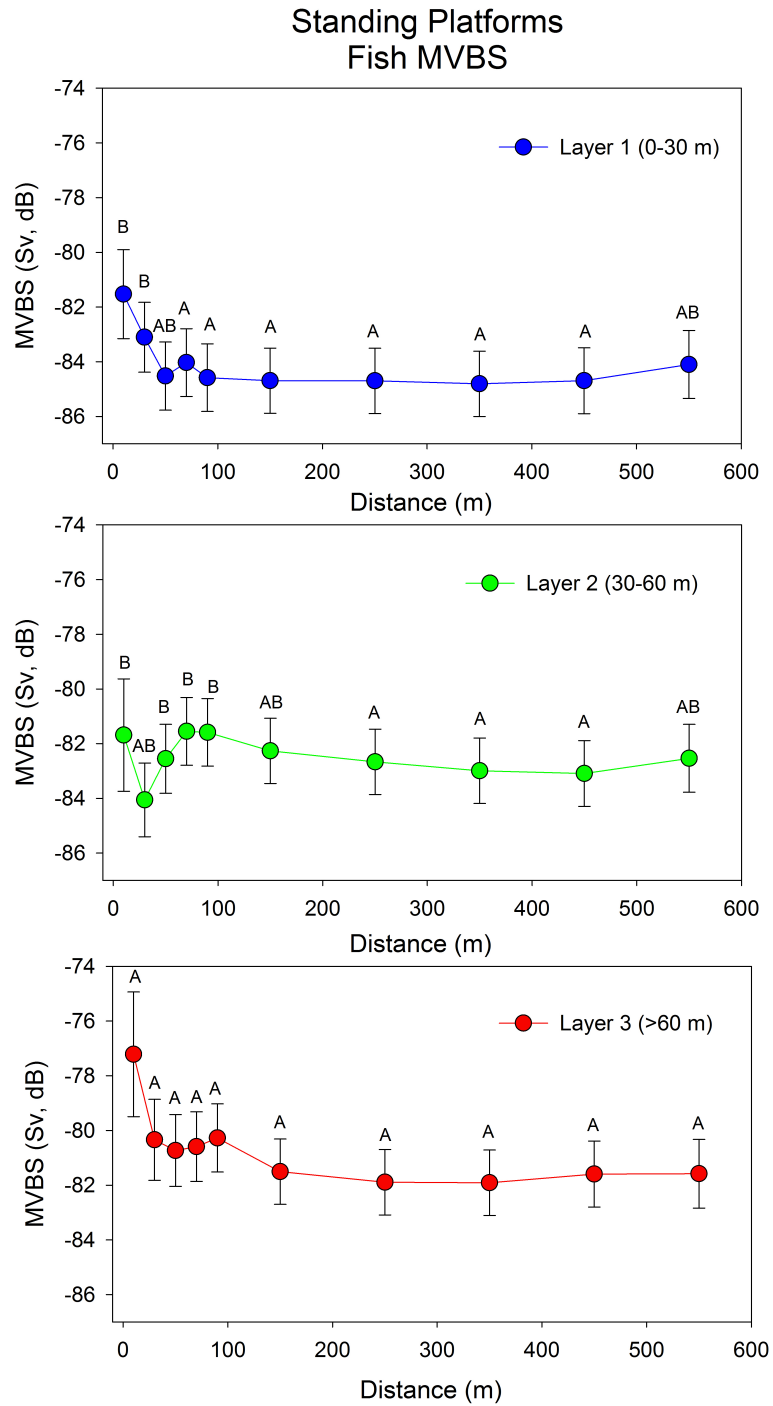


Figure A.3. LSmean-volume backscatter (MVBS, Sv) of fish data set versus distance from the structure at standing platforms in the northern Gulf of Mexico. Each layer in the water column is graphed separately; depth layer 1 (0-30 m) (blue), depth layer 2 (30-60 m) (green), depth layer 3 (>60 m) (red). Groups that have different letters are significantly different from each other.

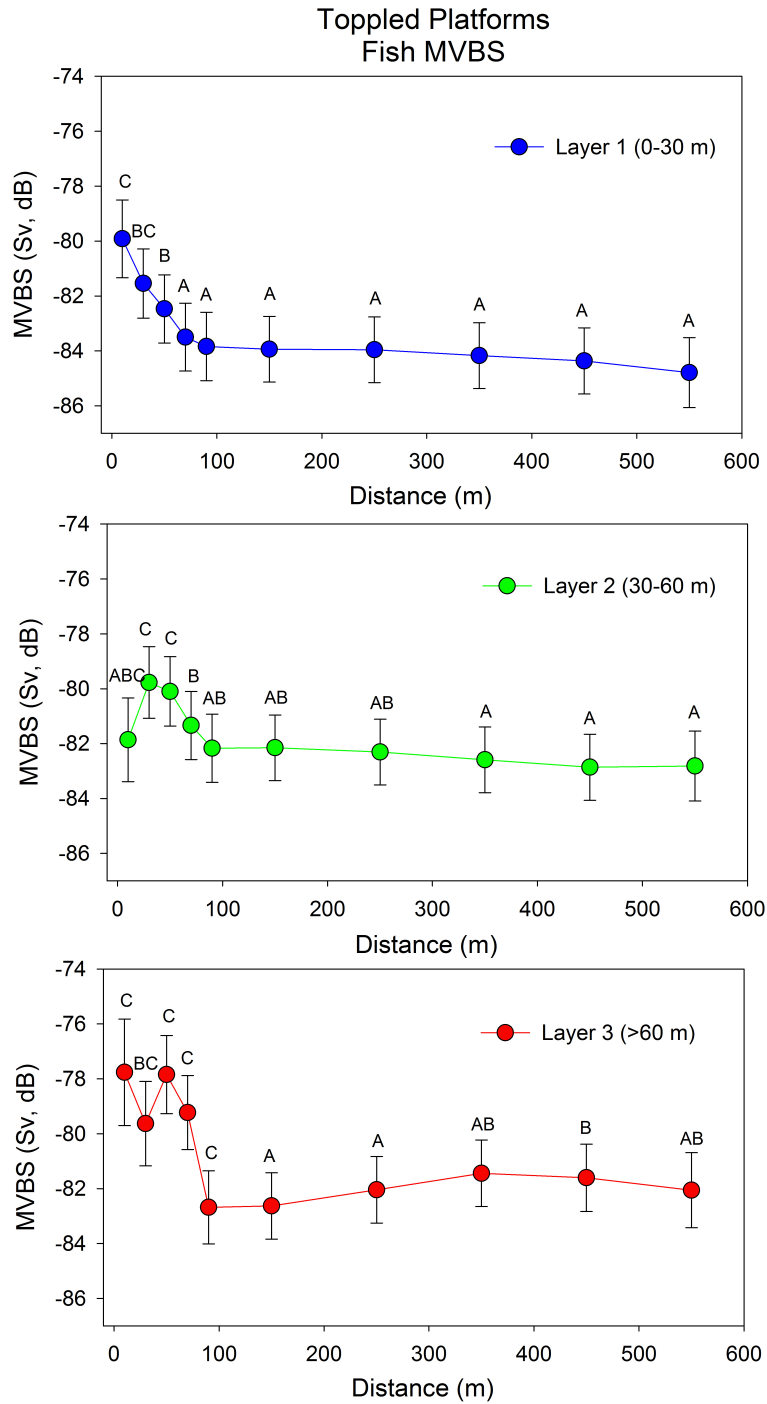


Figure A.4. LSmean-volume backscatter (MVBS, Sv) of fish data set versus distance from the structure at toppled platforms in the northern Gulf of Mexico. Each layer in the water column is graphed separately; depth layer 1 (0-30 m) (blue), depth layer 2 (30-60 m) (green), depth layer 3 (>60 m) (red). Groups that have different letters are significantly different from each other.

## APPENDIX B: CHAPTER 3 SUPPLEMENTARY DATA

Table B.1. Table of coefficients and their respective sources used to calculate individual species length-weight relationships. Coefficients are unitless, length used in calculation is Fork Length in cm and weight is in grams.

| Species           | Scientific name                 | Coefficients |       | Source                            |
|-------------------|---------------------------------|--------------|-------|-----------------------------------|
|                   |                                 | a            | b     |                                   |
| Almaco jack       | <i>Seriola rivoliana</i>        | 0.0146       | 3.055 | Bohnsack and Harper (1998)*       |
| Bar jack          | <i>Caranx ruber</i>             | 0.00738      | 3.237 | Bohnsack and Harper (1998)*       |
| Barracuda         | <i>Sphyraena barracuda</i>      | 0.00497      | 3.038 | Bohnsack and Harper (1998)*       |
| Bermuda chub      | <i>Kyphosus sectarix</i>        | 0.0174       | 3.079 | Bohnsack and Harper (1998)*       |
| Blue runner       | <i>Caranx crysos</i>            | 0.0525       | 2.69  | Bohnsack and Harper (1998)*       |
| Crevalle jack     | <i>Caranx hippos</i>            | 0.0517       | 2.734 | Bohnsack and Harper (1998)*       |
| Dusky shark       | <i>Carcharhinus obscurus</i>    | 0.0324       | 2.786 | Kohler et al. (1995) <sup>§</sup> |
| Gray snapper      | <i>Lutjanus griseus</i>         | 0.0232       | 2.881 | Bohnsack and Harper (1998)*       |
| Gray triggerfish  | <i>Balistes capriscus</i>       | 0.0251       | 2.935 | Bohnsack and Harper (1998)*       |
| Greater amberjack | <i>Seriola dumerili</i>         | 0.0363       | 2.771 | Frota et al. (2004) <sup>°</sup>  |
| Horse-eye jack    | <i>Caranx latus</i>             | 0.0674       | 2.668 | Frota et al. (2004) <sup>°</sup>  |
| Yellow jack       | <i>Carangoides bartholomaei</i> | 0.0259       | 2.908 | Bohnsack and Harper (1998)*       |
| Little tunny      | <i>Euthynnus alletteratus</i>   | 0.0138       | 3.035 | Diouf, T. (1980) <sup>#</sup>     |

(Table B.1 continued)

| Species            | Scientific name                 | Coefficients          |       | Source                                       |
|--------------------|---------------------------------|-----------------------|-------|--|
|                    |                                 | a                     | b     |  |
| Rainbow runner     | <i>Elagatis bipinnulata</i>     | 0.0135                | 2.92  | Kulbicki et al. (1993) <sup>¶</sup>          |
| Red snapper        | <i>Lutjanus campechanus</i>     | 1.17x10 <sup>-8</sup> | 3.04  | Wilson and Nieland (2001)                    |
| Scamp              | <i>Mycteroperca phenax</i>      | 0.01435               | 3.00  | Matheson et al. (1986)                       |
| Silky shark        | <i>Carcharhinus falciformis</i> | 0.0154                | 2.922 | Kohler et al. (1995) <sup>§</sup>            |
| Vermillion snapper | <i>Rhomboplites aurorubens</i>  | 0.0172                | 2.996 | Claro and Garcia-Arteaga (1994) <sup>^</sup> |

\*Bohnsack and Harper. 1998. Length-weight relationships of selected marine fishes from the southeastern United States and the Caribbean. NOAA Tech. Mem. NMFS-SEFEC-215:31 p.

<sup>^</sup>Claro, R. and J.P. García-Arteaga. 1994. Crecimiento. p.321-402. In R.Claro (ed.). Ecología de los peces marinos de Cuba. Instituto de Oceanología Academia de Ciencias de Cuba and Centro de Investigaciones de Quintana Roo (CIQRO), México.

<sup>#</sup>Diouf, T. 1980. Pêche et biologie de trois Scombridae exploités au Sénégal: *Euthymus alletteratus*, *Sarda sarda* et *Scomberomorus tritor*. These de Doctorat 3ème cycle. Université de Bretagne Occidentale, France. 159 p.

<sup>°</sup>Frota, L.O., P.A.S., Costa and A.C. Braga. 2004. Length-weight relationships of marine fishes from the central Brazilian coast. NAGA WorldFish Center Q. 27 (1&2): 20-26.

<sup>§</sup>Kohler, N.E., J.G. Casey, and P.A. Turner. 1995. Length-weight relationships for 13 species of sharks from the western North Atlantic. Fish Bull. 93: 412-418.

<sup>¶</sup>Kulbicki, M., G. Mout Tham, P. Thollot and L. Wantiez. 1993. Length-Weight relationships of fish from the lagoon of New Caledonia. Naga ICLARM Q. 16(2-3): 26-29.

Table B.2. List of species observed during video surveys; For each fish: family, species, number of fish observed over all surveys combined (n), percentage of total number of fishes seen, number of surveys within which the species was observed, and the percentage of all surveys during which each species was observed; each survey includes all three depth layers sampled.

| Family                | Species                         | n   | Percent Total | Surveys | Percent Total Surveys |
|-----------------------|---------------------------------|-----|---------------|---------|-----------------------|
| <i>Acanthuridae</i>   | <i>Acanthurus coeruleus</i>     | 3   | 0.06          | 1       | 4                     |
| <i>Balistidae</i>     | <i>Balistes capriscus</i>       | 23  | 0.47          | 9       | 36                    |
| <i>Carangidae</i>     | <i>Caranx crysos</i>            | 937 | 19.2          | 8       | 32                    |
|                       | <i>Caranx hippos</i>            | 219 | 4.5           | 10      | 40                    |
|                       | <i>Caranx latus</i>             | 639 | 13.1          | 10      | 40                    |
|                       | <i>Caranx ruber</i>             | 46  | 0.95          | 2       | 8                     |
|                       | <i>Carangoides bartholomaei</i> | 9   | 0.18          | 2       | 8                     |
|                       | <i>Elagatis bipinnulata</i>     | 755 | 15.4          | 7       | 28                    |
|                       | <i>Selene vomer</i>             | 12  | 0.24          | 1       | 4                     |
|                       | <i>Seriola dumerili</i>         | 320 | 6.55          | 20      | 80                    |
|                       | <i>Seriola rivoliana</i>        | 70  | 1.43          | 14      | 56                    |
| <i>Carcharhinidae</i> | <i>Carcharhinus falciformis</i> | 9   | 0.18          | 4       | 16                    |
|                       | <i>Carcharhinus brevipinna</i>  | 18  | 0.37          | 3       | 12                    |
|                       | <i>Carcharhinus obscurus</i>    | 2   | 0.04          | 2       | 8                     |
|                       | <i>Carcharhinus plumbeus</i>    | 2   | 0.04          | 2       | 8                     |
| <i>Chaetodontidae</i> | <i>Chaetodon ocellatus</i>      | 2   | 0.04          | 1       | 4                     |
| <i>Echeneidae</i>     | <i>Remora remora</i>            | 11  | 0.22          | 4       | 16                    |
| <i>Ephippidae</i>     | <i>Chaetodipterus faber</i>     | 1   | 0.02          | 1       | 4                     |
| <i>Kyphosidae</i>     | <i>Kyphosus sectatrix</i>       | 186 | 3.81          | 7       | 28                    |
| <i>Labridae</i>       | <i>Bodianus puchellus</i>       | 22  | 0.45          | 3       | 12                    |
|                       | <i>Bodianus rufus</i>           | 2   | 0.04          | 1       | 4                     |
| <i>Lutjanidae</i>     | <i>Lutjanus campechanus</i>     | 756 | 15.5          | 22      | 88                    |
|                       | <i>Lutjanus cyanopterus</i>     | 1   | 0.02          | 1       | 4                     |
|                       | <i>Lutjanus griseus</i>         | 19  | 0.39          | 5       | 20                    |
|                       | <i>Rhomboplites aurorubens</i>  | 75  | 1.53          | 5       | 20                    |
| <i>Pomacanthidae</i>  | <i>Pomacanthus paru</i>         | 2   | 0.04          | 1       | 4                     |
| <i>Rachycentridae</i> | <i>Rachycentron canadum</i>     | 1   | 0.02          | 1       | 4                     |
| <i>Scombridae</i>     | <i>Euthynnus alletteratus</i>   | 726 | 14.9          | 11      | 44                    |
| <i>Serranidae</i>     | <i>Mycteroperca phenax</i>      | 3   | 0.06          | 3       | 12                    |
|                       | <i>Ephinephelus nigritus</i>    | 1   | 0.02          | 1       | 4                     |
| <i>Sphyraenidae</i>   | <i>Sphyraena barracuda</i>      | 13  | 0.27          | 7       | 28                    |
| Totals                | Total numbers (n=4885)          |     |               |         |                       |
|                       | Total number of surveys=25      |     |               |         |                       |

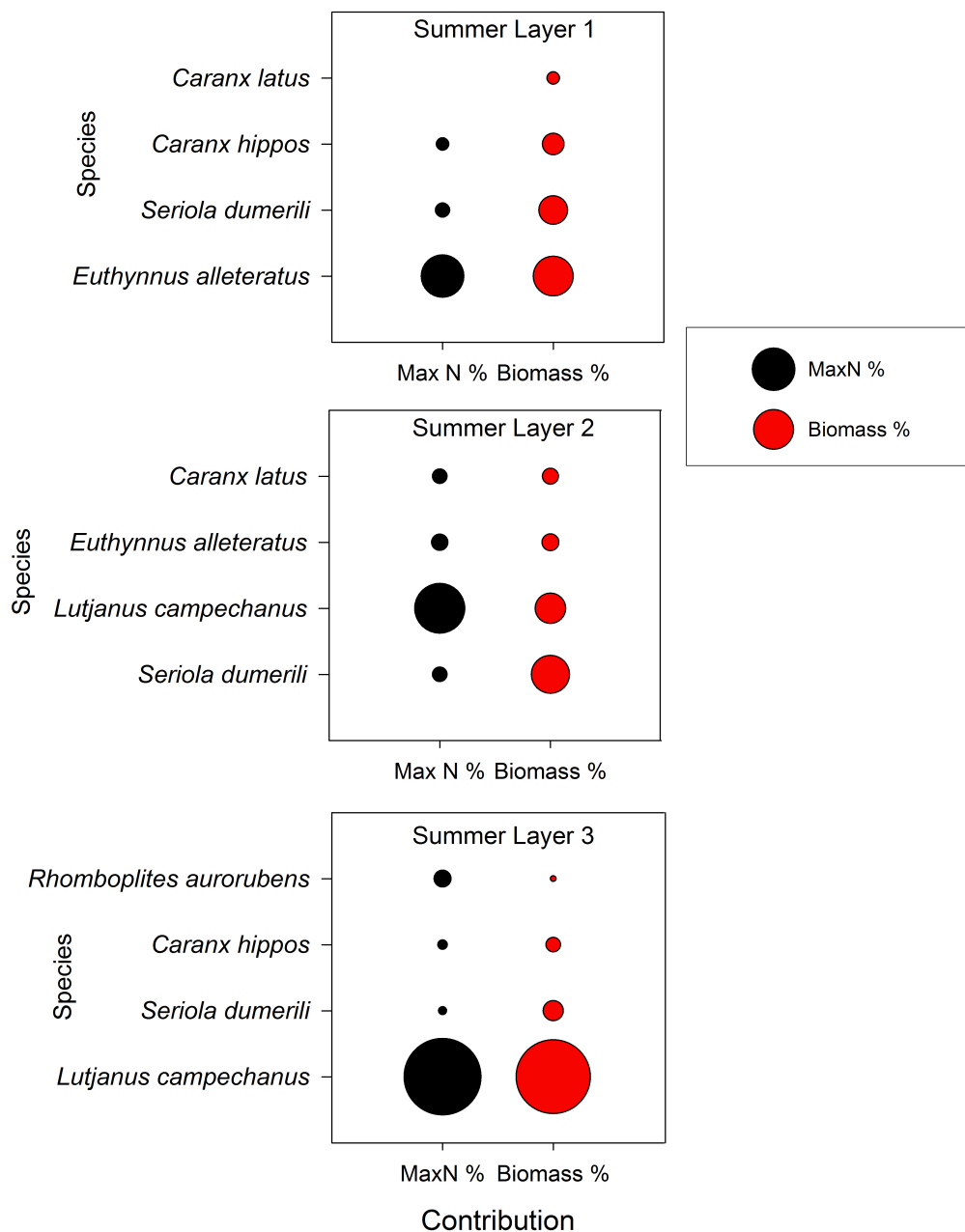


Figure B.1. Bubble plot of the four most dominant species contributing to relative MaxN (black) and relative Biomass (red) within layers during summer; Summer= June, July, August. Fall= September, October, November. Winter= December, January, February. Layer 1= 0-30 m. Layer 2= 30-60 m. Layer 3= > 60 m.



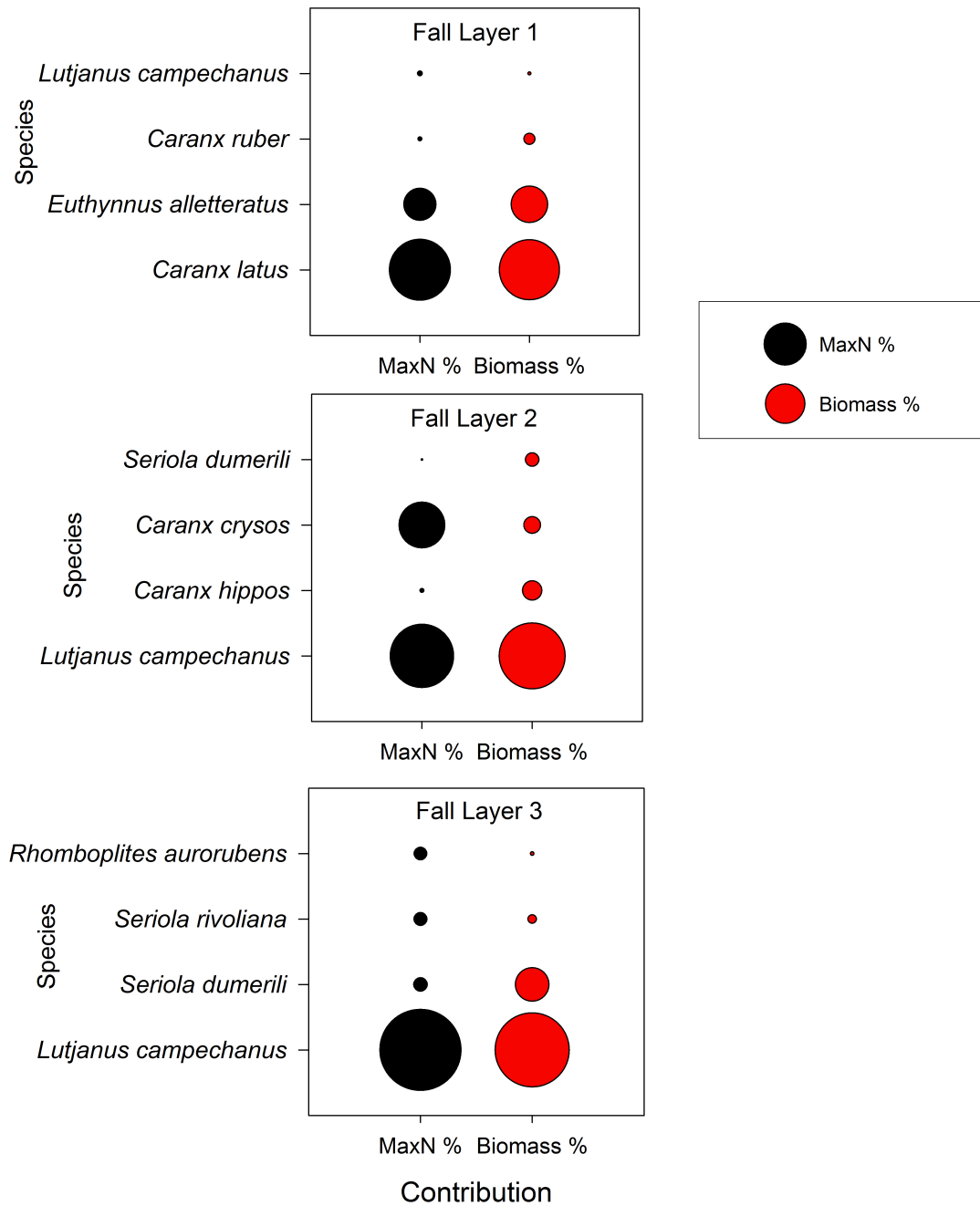


Figure B.2. Bubble plot of the four most dominant species contributing to relative MaxN (black) and relative Biomass (red) within layers during fall; Summer= June, July, August. Fall= September, October, November. Winter= December, January, February. Layer 1= 0-30 m. Layer 2= 30-60 m. Layer 3= > 60 m.

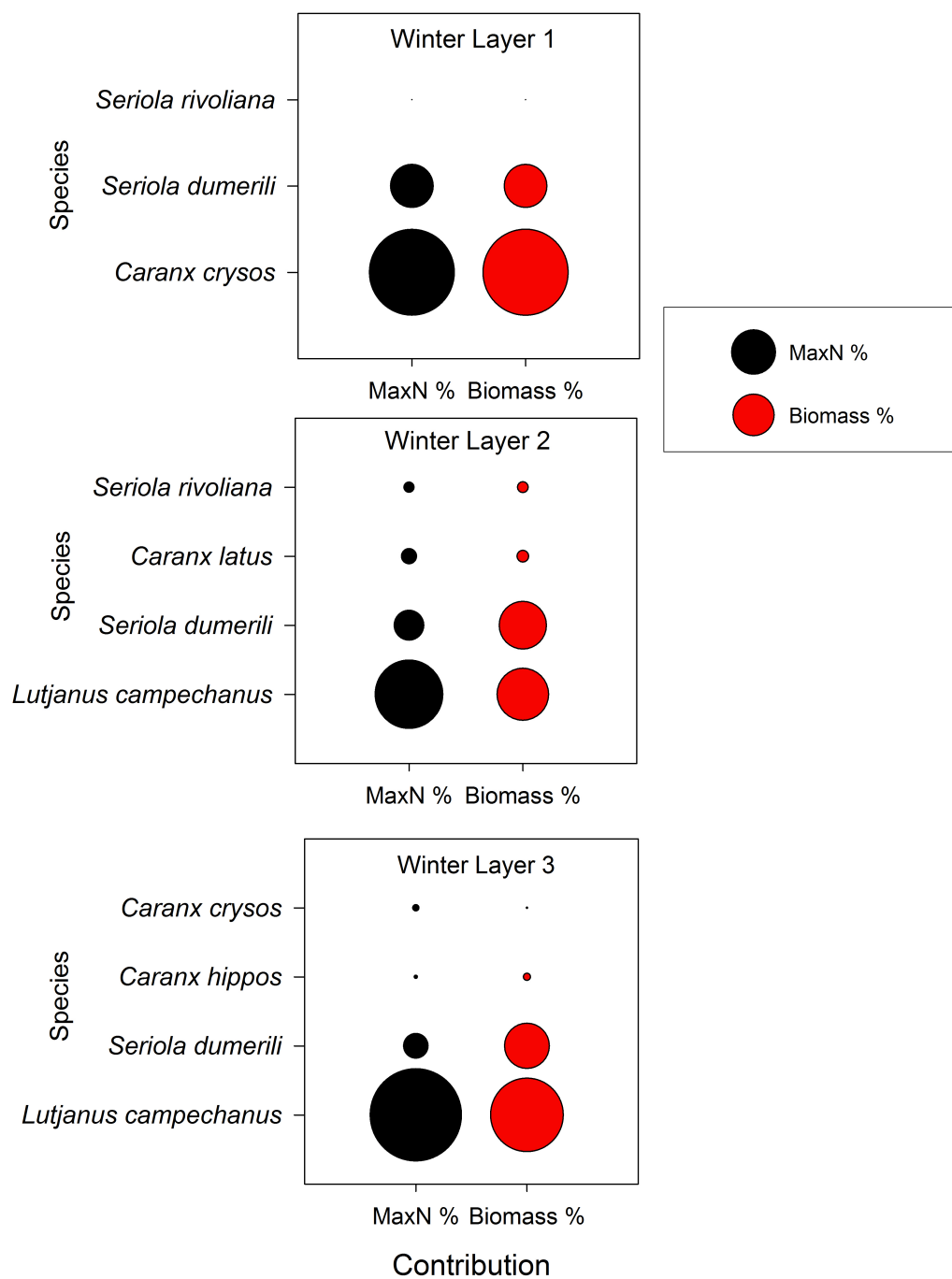


Figure B.3. Bubble plot of the four most dominant species contributing to relative MaxN (black) and relative Biomass (red) within layers during winter; Summer= June, July, August. Fall= September, October, November. Winter= December, January, February. Layer 1= 0-30 m. Layer 2= 30-60 m. Layer 3= > 60 m.

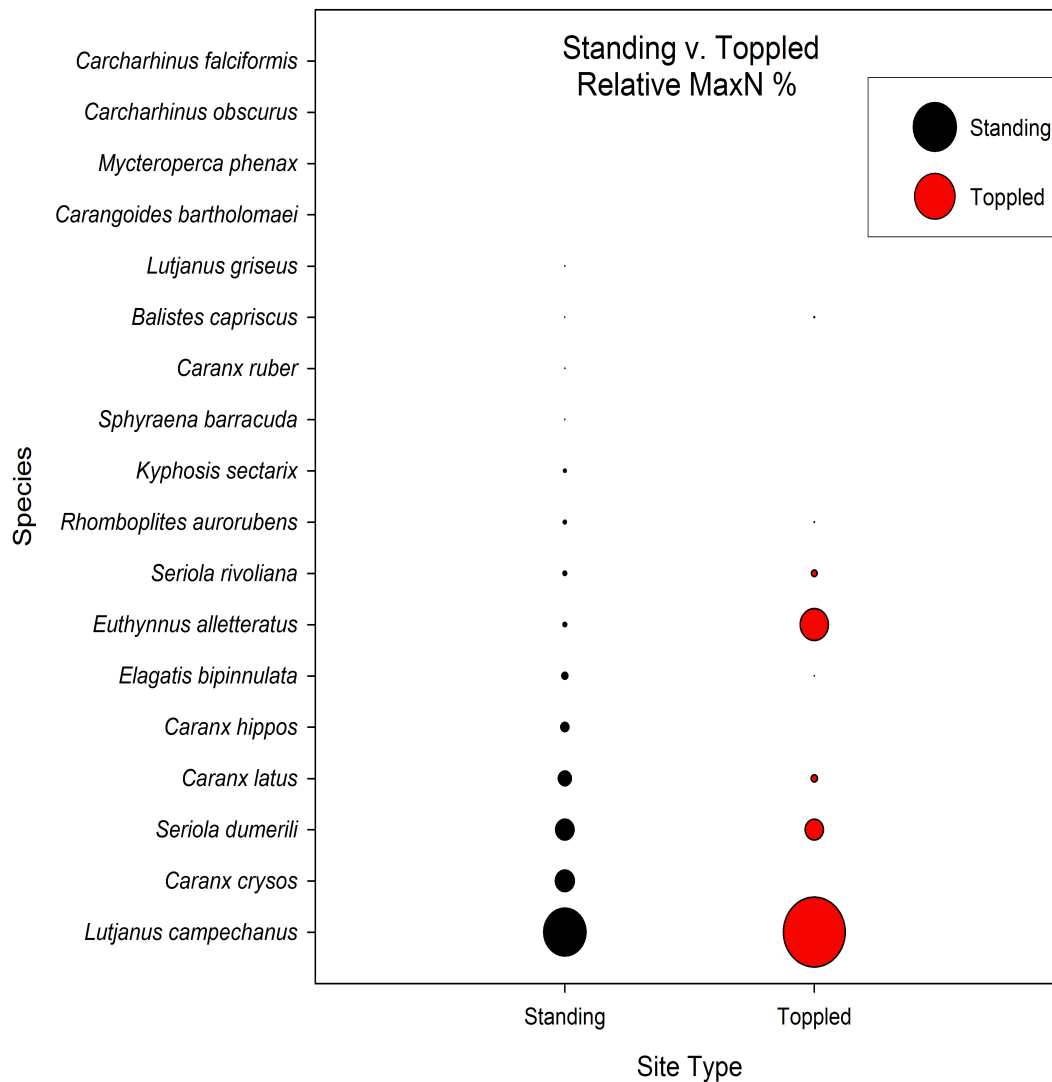


Figure B.4. Bubble plot of species contributing to relative MaxN at standing (black) and toppled (red) oil and gas platforms in the northern Gulf of Mexico; Species shown are the species present at both structure types; Note that some species had values that were too small to see on this graph.

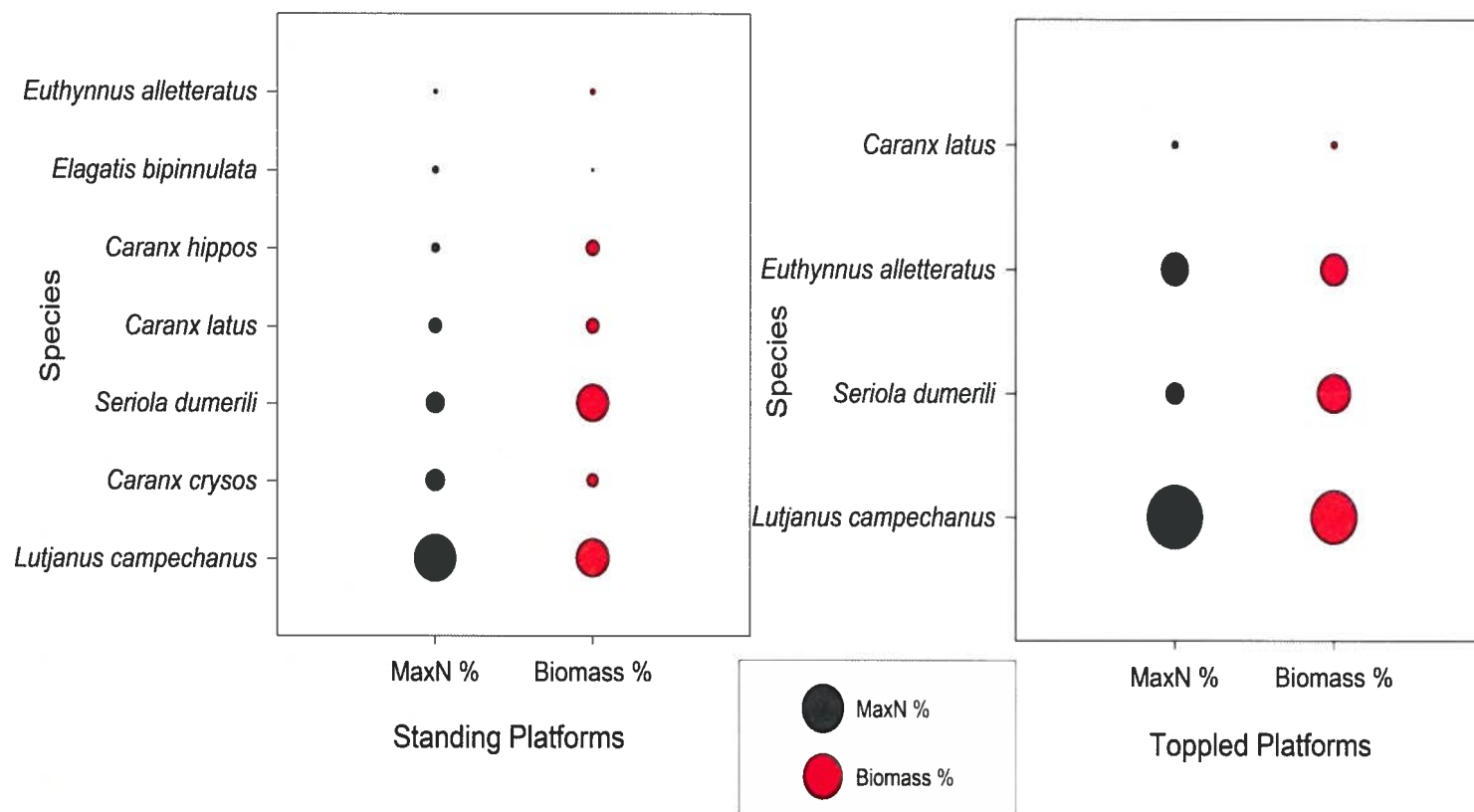


Figure B.5. Bubble plot of species contributing to 90% of the fish observed at standing and toppled platforms by relative MaxN (black); relative Biomass (red) is shown strictly for comparison.

## VITA

Emily Marissa Reynolds was born in Washington, D.C. in August of 1990. She grew up an only child in Silver Spring, Maryland where she became an avid swimmer at the age of 5. She immediately found a love for the water, including every animal in the water and on land. She graduated from James Hubert Blake High School in 2008 and moved to Atlanta, Georgia to attend Emory University where she earned a Bachelor of Science in Environmental Studies with a minor in Sociology. She swam competitively on Emory's NCAA Division III Swimming team all four years, becoming an Academic All-American. She studied abroad in the Turks and Caicos Islands, where her passion for marine science grew. It was during these three months that she decided she wanted to pursue fisheries research and continued chasing this dream by becoming an intern at the National Aquarium and later spending a summer as an aquaculture intern at Harbor Branch Oceanographic Institute in Ft. Pierce, Florida. At a National Marine Fisheries Service undergraduate workshop during the spring of 2012, she learned of a great graduate school program, and in 2012 she entered the masters program in the Department of Oceanography and Coastal Sciences at Louisiana State University (LSU), under the supervision of Dr. James H. Cowan, Jr. and Dr. Joseph Powers. At LSU she served two years on the Education and Outreach Committee (chair for one year) for the Coast and Environment Graduate Organization (CEGO). She participated in multiple conferences during her time at LSU and served as Chief Scientist for the research project. She is currently a candidate for the degree of Master of Science in the Department of Oceanography and Coastal Sciences, which will be awarded in August 2015.