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research paper

Fish assemblages associated with artificial reefs assessed using multiple gear types in the northwest Gulf of Mexico

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ABSTRACT.—Quantitative surveys of fishes associated with artificial reefs in the northwest Gulf of Mexico were conducted over a 4-yr period (2014–2017). Artificial reefs surveyed were comprised of three types: concrete structures, rig jackets, and decommissioned ships. All reefs were surveyed using vertical long line (VLL), fish traps, and Adaptive Resolution Imaging Sonar (ARIS 1800). Mean fish abundance did not significantly differ using VLL [1.7 ind set^{-1} (SD 2.2)] among the three reef types. However, relative abundance among all fishes collected was significantly highest on rig reefs using traps [6.2 ind soak^{-1} (SD 3.8)], while results from sonar surveys indicated that the mean relative fish density was highest on concrete reefs [15.3 fish frame^{-1} (SD 26.8)]. Red snapper (n = 792), followed by gray triggerfish (n = 130), pigfish (n = 70), tomtate (n = 69), and hardhead catfish (n = 57) were the most numerically abundant species using VLL and traps; red snapper comprised 90.7% of total catch using VLL and 43.9% using traps. Mean Brillouin's diversity (H_B) was highest on ships using VLL [0.41 (SD 0.14)] and highest on rigs using traps [0.87 (SD 0.58)] compared to the lowest diversity found on concrete [VLL 0.07 (SD 0.11); traps 0.36 (SD 0.32)]. Findings from this study can be used to inform the planning of future artificial reefs and their effect on the assemblages of reef-associated fishes. Additionally, these results highlight the value of using multiple gear types to survey reef fish assemblages associated with artificial reefs.

Fish assemblages associated with artificial reefs are affected by differences in reef structure, depth, and size of habitats (Rooker et al. 1997, Rilov and Benayahu 2000, Strelcheck et al. 2005, Perkol-Finkel et al. 2006). Factors affecting the assemblages of fishes on artificial reefs have been widely studied in the northern Gulf of Mexico (GOM) and found to have similar drivers to other reef systems, with vertical relief,

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rugosity, and available surface for colonizing organisms commonly identified as influential factors (Rooker et al. 1997, Boswell et al. 2010, Dance et al. 2011, Gregalis et al. 2012, Ajemian et al. 2015a). Large man-made structures used to create complex, high-relief artificial reefs (e.g., free standing oil and gas rig jackets, decommissioned ships) harbor diverse fish assemblages off the coast of Texas (Ajemian et al. 2015a) and throughout the GOM (Boswell et al. 2010, Reynolds et al. 2018). Meanwhile, smaller concrete structures with lower relief (quarry rocks, pyramids, reef balls) have been widely implemented to supplement natural hard structure across the region (Strelcheck et al. 2005, Dance et al. 2011) including the Texas coast (Arney et al. 2017, Streich et al. 2017a). Low-relief artificial reefs have also been shown to attract fishes similarly to high-relief reefs; however, these assemblages often differ in demography, composition, and reduced diversity (Rilov and Benayahu 2000).

Artificial reef programs have been widely implemented in coastal areas to supplement or replace benthic habitat (Baine 2001). In 1984, the United States implemented the National Fishing Enhancement Act (NFEA) calling for the responsible and effective use of artificial reefs to enhance recreational and commercial fisheries. Within the northern GOM, there are numerous artificial structures that serve as potential reef habitat including 1786 currently active oil platforms (as of December 2019; BOEM, www.data.boem.gov). To decommission current oil and gas marine structures and permit reefing sites, individual states must mediate between current oil and gas leasing stakeholders working with federal agencies such as the United States Army Corps of Engineers (USACE), United States Coast Guard (USCG), and Bureau of Ocean Energy Management (BOEM). In the NW GOM, the Texas Parks and Wildlife Department (TPWD) has helped to deploy artificial reefs on the inner continental shelf off the cost of Texas since 1990 (www.tpwd.Texas.gov) in response to NFEA. According to TPWD, three programs operating within the greater Artificial Reef Program have been developed to help with the design and implementation of these artificial reefs: Rigs-to-Reefs, Ships-to-Reefs, and Nearshore Reefing. TPWD's goal of creating these artificial reefs was to supplement the NW GOM's low density of natural reef structures and limited hard bottom habitat. Artificial reefs monitored by TPWD consist of structures including ships, rocks, prefabricated concrete, and rig jackets. These artificial reef structures differ with respect to vertical relief, water depth, and proximity to shore, and they are generally used to accomplish different goals, such as enhance local marine communities and tourism. This network of diverse reef structures along the Texas coast provides an ideal location for examining the effects of reef structure on associated reef fish assemblages.

Size or species-specific gear bias of single gear types may bias estimates of the relative abundance of fishes (Willis et al. 2000, Diaz et al. 2003). Fishery-independent surveys using trawls, longlines, entanglement nets, and traps can result in varied catch composition with each gear having size and species-specific biases (Wells et al. 2008). Historically, less invasive methods such as active acoustics were capable of providing biomass estimates but were expensive and did not have enough resolution to create accurate counts or identify species (Jolly and Hampton 1990). However, increased resolution with the innovation of multi-beam transducers has enhanced the ability to enumerate and, in ideal cases, identify fishes taxonomically (Holmes et al. 2006). Adaptive Resolution Imaging Sonar (ARIS) is one tool that allows users to obtain near video-quality imaging to enumerate and observe fishes in habitats that are turbid or low-light similar to other acoustic cameras (e.g., DIDSON; Moursund et al. 2003). The NW GOM can be a challenging location to conduct visual surveys due to a strong nepheloid layer (Shideler 1981, Rezak et al. 1985) and high levels of turbidity in coastal waters (Kennicutt 2017). The ARIS 1800 (Soundmetrics Corp.) uses a high-operating frequency, 1.1 MHz, at distances greater than 15 m and 1.8 MHz at distances less than 15m (www.soundmetrics.com) to track moving objects with higher resolution than traditional single transducer sonars. In comparison to other sampling techniques (e.g., hook and line fishing, netting), ARIS surveys are minimally invasive, and therefore less likely to disturb or influence observations of study species (Boswell et al. 2010). Relative abundance estimates using active acoustics, in conjunction with more traditional sampling techniques, can help to evaluate fish assemblage structure, especially when visual surveys are not an option.

Fishery-independent surveys are often used in an attempt to remove the bias of catch data normally associated with fishery-dependent sampling (Rotherham et al. 2007). Fishery-independent surveys using vertical longlines (VLLs) are commonly used to sample fishes associated with artificial reefs in the northern GOM (Gregalis et al. 2012), but have been shown to be strongly biased toward red snapper (*Lutjanus campechanus*; Scott-Denton et al. 2011). Fish traps are also effective tools for sampling fishes associated with complex structures (Newman and Williams 1995) as they appear to sample a broader range of size classes and species than VLLs (Streich et al. 2017a), but are size selective based upon the size of the trap, the trap opening, and mesh (Wells et al. 2008). Acoustic surveys are effective at estimating relative fish abundance and biomass (Boswell et al. 2010) and are highly effective in turbid environments (Mueller et al. 2006), but this gear provides limited taxonomic resolution (Mueller et al. 2010) and may not accurately quantify more sedentary benthic fishes (Able et al. 2014). Thus, a combination of these techniques can utilize their unique strengths and be used to better characterize fish assemblage structure.

Surveys of fish assemblages on nearshore artificial reefs along the northern Texas coast are limited, and most of the recent research to date has been performed along the mid to south Texas coastline (Rooker et al. 1997, Ajemian et al. 2015a, Arney et al. 2017, Streich et al. 2017a,b). With respect to two of the rigs which were turned into reefs and included in this study, Rig B and Rig C, both were included in a community level assessment in 1982, 32 yrs before this study was conducted (Gallaway and Lewbel 1982). Moreover, only two other studies have addressed the questions of gear selectivity on fishes associated with artificial reefs of Texas (Ajemian et al. 2015b, Streich et al. 2018) and another on the validity of acoustic techniques to assess fish biomass (Bollinger and Kline 2017). The objectives of this study were to use traditional fisheries survey methods and active acoustics to characterize temporal and spatial differences in fish assemblages across three different artificial reef structures (concrete structures, rig jackets, and decommissioned ships) along the upper coast of Texas. We hypothesized that due to the increased rugosity and water depth (due to the logistic constraints for the placement of these reefs), rig and ship sites would have higher diversity and relative abundances than low-relief concrete reefs in this region.

Materials and Methods

Surveys.—Fishery-independent surveys were conducted at eight artificial reef sites, representing three reef types including concrete structures (n = 3), rig jackets (n = 3), and decommissioned ships (n = 2; Table 1, Fig. 1). Sampling was conducted

Table 1. Description of each structure using estimated footprint size, relief, depth, and materials used in reef construction provided through the Texas Parks and Wildlife's Texas Artificial Reef Interactive Mapping Application.

Structure	Footprint (km^2)	Relief (m)	Depth (m)	Materials
Concrete A	1.6	4.0	13.1	Prefabricated reef pyramids, quarry rocks
Concrete B	1.3	4.0	13.1	Quarry rocks
Concrete C	3.3	4.9	15.2	Concrete anchors, MARAD, buoy pieces, quarry rocks
Rig A	1.7	4.9	20.1	Rig jacket pieces, quarry rocks, 2 m caisson
Rig B	2.9	6.1	21.3	Rig jacket pieces
Rig C	1.9	6.1	21.3	Rig jacket pieces
Ship A	59.5	6.7	16.8	Prefabricated reefs (including pyramids), concrete culverts, George Vancouver Liberty Ship
Ship B	14.6	12.8	31.1	Rig jacket pieces, coal ash blocks, SS V.A Fogg, William F. Allen Liberty Ship

over a 4-yr period (2014–2017) with each sampling event (fishing and ARIS surveys, respectively) occurring once per year per site (Table 2). Sampling occurred primarily from May through August for each year sampled, with two exceptions in 2014 where sampling events occurred in October and December, respectively. Reef sites varied in both depth (m) and distance from shore (km), while depth and reef were highly correlated due to the constraints of maintaining navigable waters above the reef structure. Concrete structures were reefs characterized as low relief (<4.9 m) that primarily consisted of quarry rock, MARAD buoy pieces, experimental reef pyramids, and large concrete anchors located in the northern region of the sampling area (hereafter concrete A, B, and C). Artificial reefs with higher relief (>4.9 m) that were primarily constructed from toppled and cutoff rig jackets as well as concrete blocks located in the central region of the sampling area were designated as rig reefs (hereafter rig A, B, and C). Reef structures that included decommissioned sunken ships along with rig jackets and concrete pyramids located further south than the other reef types were designated as ship reefs (hereafter ship A and B; Fig. 1). Abiotic parameters, including both salinity (PSU) and temperature (°C), were measured at the bottom via sonde at each site. Each site consisted of three replicate subsites sampled haphazardly within the reef footprint using both VLL and small fish traps. VLL fishing followed pre-established SEAMAP (2013) protocol and was accomplished using bandit reels spooled with 136 kg test mainline and outfitted with a 7.3 m backbone of 181 kg test monofilament containing 10 gangions (45 kg test, 45.7 cm). The terminal end of the backbone was weighted with a 3.6 kg mushroom weight and each gangion was baited with cut Atlantic mackerel (*Scomber scombrus*). VLL sampling of each

Table 2. Surveyed replicates for each structure type over the 4-yr (2014–2017) sampling period.

Structure	Years Sampled	2014	2015	2016	2017
VLL and Trap Surveys					
Concrete	3	0	12	6	6
Rig	4	9	9	9	6
Ship	3	6	6	3	0
ARIS Surveys					
Concrete	1	0	9	0	0
Rig	3	9	9	3	0
Ship	1	0	9	0	0

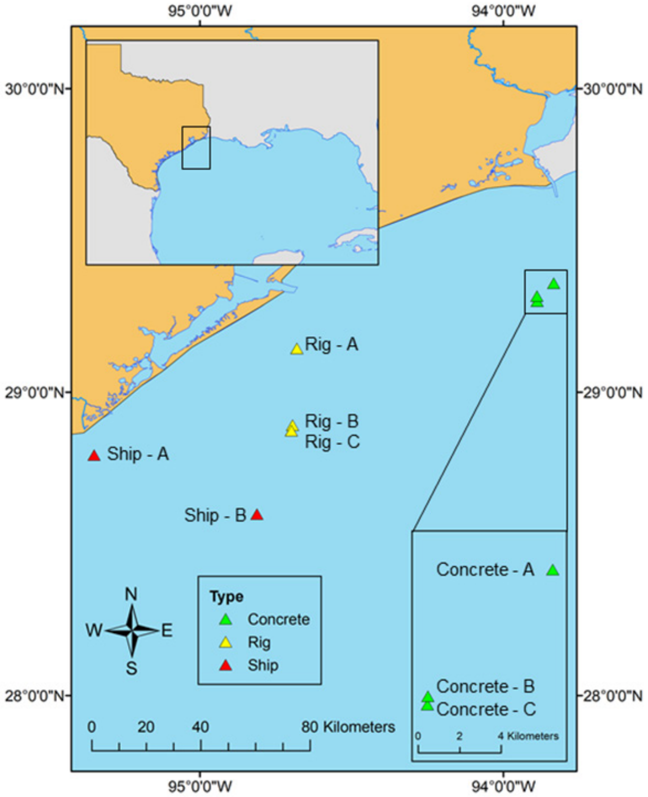

Figure 1. Map of the artificial reef sites along the northern Texas coast.

subsite replicate consisted of four separate soaks of a backbone outfitted with gangions of one hook size (2/0, 8/0, 11/0, and 15/0) which was fished in haphazard order on alternating sides of the boat for 5 min. In addition to VLL, two small fish traps—identical to those used in Brandt and Jackson (2013) and Streich et al. (2017a); 63.5 cm width × 96.5 cm length × 50.8 cm height, 1.6 cm mesh with two 15.2 cm × 13.7 cm openings—also baited with cut Atlantic mackerel, were concurrently deployed at each replicate and soaked for 1 hr. All fishes after being caught were identified, enumerated, and measured in length (mm fork length, FL).

On nonconcurrent sampling events, a 5-min drift transect survey using the ARIS 1800 sonar with attached spreader lens that maintained a 28° viewing field was conducted at each site with three replicates at each subsite to estimate the relative fish density (fish frame^{-1}). The ARIS unit was attached to a rotating arm (AR2, SoundMetrics) for axis control and to a hydrofoil prior to being submerged for each drift transect. Once submerged, the ARIS unit was pointed downwards (angle varied from 0° to 20°) toward the target structure and set to record (Fig. 2). Depth and distance to structure varied among sites with an average survey area of 27.5 m^2 (SD 8.7), which was corrected for by removing frames that did not include structure. Each site was visited in either 2014 or 2015, with an additional survey of rig B in 2016.

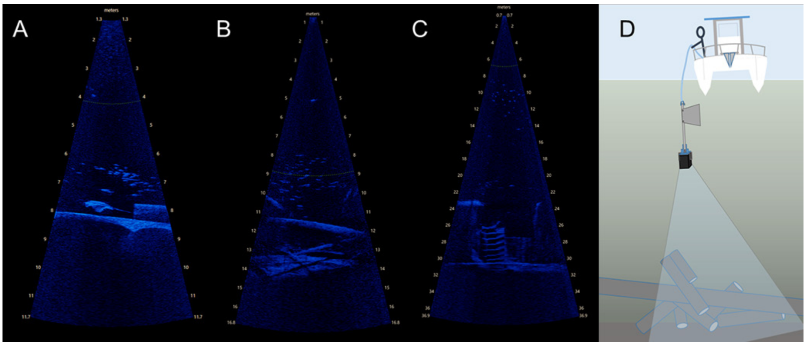

Figure 2. ARIS images representing each reef type: (A) concrete, (B) rig jackets, (C) ship, and (D) schematic of ARIS surveys.

Fishing Survey Analysis.—Catch from the VLL and trap surveys were converted into catch per unit effort (CPUE) to estimate the relative abundance of each species by gear type at each site (Table 1). Abiotic conditions, depth, and distance from shore were investigated using a Kruskal-Wallis test due to the lack of normality and heteroscedasticity. To determine if our sampling was adequate for describing the fish assemblages as it pertained to each gear type, we created species accumulation curves using the vegan package v2.5 in R v3.51 (Oksanen et al. 2007, R Core Team 2014). Species accumulation curves were created using random starts, permuted 999 times, and presented using a 95% confidence interval. Due to differences between the two gear types, (5-min VLL soak vs 1-hr trap soak), they were not compared statistically. Due to the violation of assumptions (nonnormal distribution), unequal variances (heteroscedasticity), and a high proportion of zeros, nonparametric statistical tests were used to determine differences in reef fish assemblages among factors. To identify trends in less abundant species, species-specific CPUE was 4^{th} root transformed, while total fish CPUE was left untransformed for separate analyses and used to create a Bray-Curtis similarity matrix. The similarity matrix was created using species-specific CPUE and analyzed using a two-factor permutational MANOVA (PERMANOVA) in PRIMER v7 (Clarke and Gorley 2015) to examine assemblage structure differences among factors of reef type and year. In addition to testing for differences in assemblages using the main factors of reef type and year, PERMANOVAs were also conducted nesting site within reef type and year within reef type to investigate differences in fish assemblages between the interaction of reef type and year. Approximate *P*-values for PERMANOVA were generated using Monte Carlo random draws from the asymptotic permutation distribution in nested analyses due to the limited number of possible permutations. SIMPER (similarity percentages) analysis was used to identify individual species driving the differences in fish assemblages among reef type and year factors. To determine if PERMANOVA results were due to dispersion over location effects, each PERMANOVA test was also analyzed using a betadisper analysis within the vegan package v2.5 in R v3.51.

In addition to identifying assemblage differences using relative abundance, biodiversity indices and size selectivity analyses were used to differentiate the effectiveness of each gear type. Fish assemblage diversity was estimated using the Brillouin's diversity index (H_B). Brillouin's (H_B) diversity is most useful when the randomness of a sample is not guaranteed and, in this case, when using passive gears that attract a specific number of species over another (Peet 1974). The Brillouin index is calculated as:

$$H_B = \frac{\ln N! - \sum_{i-1}^{s} \ln n_1!}{N}$$

where N is the total number of individuals in the sample, n_i is the number of individuals belonging to the i^{th} species, and S is the number of species (Brillouin 1962). Size selectivity between fishing gear types was investigated using a one-way Kruskal-Wallis test due to the nonnormality and heteroscedasticity of the species length data. Comparisons were done for species fork length between the two gear types (VLL or trap), for the three species that were caught at least two times by each gear type. The species used for size selectivity analyses between gear types were red snapper, gray triggerfish (*Balistes capriscus*), and hardhead catfish (*Ariopsis felis*). Additionally, differences in size were tested (Kruskal-Wallis and post-hoc Mann-Whitney tests) among all four hook sizes (2/0, 8/0, 11/0, and 15/0) used in VLL sampling. Size differences for fishes caught using VLLs were analyzed for the two species (red snapper and gray triggerfish) that were caught more than three times. Normality and heteroscedasticity were tested for each ANOVA prior to running pairwise post-hoc testing; post-hoc testing was performed using Shaffer's multiple comparison procedure using the multcomp package v 1.4 in R v 3.51 (Hothorn et al. 2008), and significance was determined at $P \leq 0.05$.

Acoustic Survey Analysis.—Acoustic transect data were imported using ARIScope v2.0 (SoundMetrics). Each ARIS transect was preprocessed using platform motion, beam pattern correction, and crosstalk reduction functions to maximize the clarity of each individual frame. Transects were then reviewed to determine if the structures and fishes could be qualitatively separated from background noise. Individual frames containing structure, to account for differences in bottom area surveyed, were then isolated using video software VLC (VideoLAN). One out of every 50 frames (2%) was analyzed sequentially from the original transect video and used for analysis, which is comparable to other nonautomated surveys using DIDSON (Makabe et al. 2012, Grote et al. 2014). All frames to be analyzed were then imported into Image-J (Abràmoff et al. 2004) for analysis. Frames were first set to appropriate scale using maximum depth as the distance on the y-axis, and then the color threshold was adjusted to maximum contrast to separate background from structure and fishes. The analyze particle function was then set to identify particles sized between 0.005 m^2 and 0.5 m^2 to avoid identifying objects that were too large (structure) or too small (noise). Particles isolated via the identify particles function were qualitatively reviewed to omit background noise or structure and counted. Analysis was conducted on the total fishes counted per frame (fish frame^{-1}). Fish frame^{-1} was compared across reef types, years, and sites using Kruskal-Wallis test due to lack of homogeneity of variance of fish counts and zero inflation. Individual tests were followed by a Mann-Whitney pairwise analysis and used to determine whether relative abundance was different among reef types.

Redundancy Analysis (RDA).—Further investigation of species abundance, gear type, and structure type was conducted using a redundancy analysis (RDA), which is

Table 3. Total abundance and mean size (SD) for fish caught during VLL surveys at artificial reef sites.

Species	Concrete	Rig	Ship	Mean size (mm)
Red snapper (*Lutjanus campechanus*)	277	200	51	373 (76)
Gray triggerfish (*Balistes capriscus*)	2	17	10	358 (49)
Hardhead catfish (*Ariopsis felis*)	3	0	0	347 (33)
Atlantic croaker (*Micropogonias undulatus*)	0	2	0	264 (24)
Sand seatrout (*Cynoscion arenarius*)	2	10	0	336 (36)
Pinfish (*Lagodon rhomboides*)	0	0	1	153 (0)
Atlantic spadefish (*Chaetodipterus faber*)	1	1	0	224 (0)
Black drum (*Pogonias cromis*)	0	0	1	538 (0)
Common remora (*Remora remora*)	0	1	0	672 (0)
Blue runner (*Caranx crysos*)	0	0	1	210 (0)
Atlantic sharpnose shark (*Rhizoprionodon terraenovae*)	0	0	1	690 (0)
Spinner shark (*Carcharhinus brevipinna*)	1	0	0	N/A

a method of direct gradient analysis using Canoco 5 (ter Braak and Smilauer 2012). Correlations between variables (gear type, reef type, salinity, temperature, depth, and distance from shore) and canonical axes were used to explain each variable's influence to the total abundance of fish species, and for those species that were caught more than once during sampling (Tables 3 and 4). Each axis used in the analysis was derived from an iterative reciprocal weighted averaging of species abundance among samples in addition to the ordination scores for the linear combinations of categorical variables.

Results

Species Comparisons.—A total of 72 fishing survey replicates across three varied reef types (concrete, n = 24; rig, n = 33; ship, n = 15) were analyzed along with 39 acoustic survey replicates (concrete, n = 9; rig, n = 21; ship, n = 9). Abiotic variables among reefs including salinity [concrete, 34.35 (SD 1.85); rig, 35.72 (SD 0.83); ship, 35.42 (SD 2.21)); $\chi^2_{2,24} = 5.582$, $P = 0.061$), temperature [concrete, 28.94 °C (SD 2.33);

Table 4. Total abundance and mean size (SD) for fish caught during trap surveys at artificial reef sites.

Species	Concrete	Rig	Ship	Mean size (mm)
Red snapper (*Lutjanus campechanus*)	80	150	34	251 (54)
Gray triggerfish (*Balistes capriscus*)	30	57	14	270 (74)
Pigfish (*Orthopristis chrysoptera*)	1	58	11	203 (20)
Tomtate (*Haemulon aurolineatum*)	0	54	14	228 (22)
Hardhead catfish (*Ariopsis felis*)	34	19	1	333 (24)
Lane snapper (*Lutjanus synagris*)	0	12	3	233 (33)
Atlantic croaker (*Micropogonias undulatus*)	3	7	0	228 (24)
Pinfish (*Lagodon rhomboides*)	0	8	0	185 (35)
Atlantic spadefish (*Chaetodipterus faber*)	4	0	1	117 (25)
Oyster toadfish (*Opsanus tau*)	0	4	0	311 (43)
Gray snapper (*Lutjanus griseus*)	0	1	0	331 (0)
Southern kingfish (*Menticirrhus americanus*)	0	1	0	298 (0)
Gag grouper (*Mycteroperca microlepis*)	0	1	0	283 (0)
Atlantic bumper (*Chloroscombrus chrysurus*)	0	1	0	182 (0)

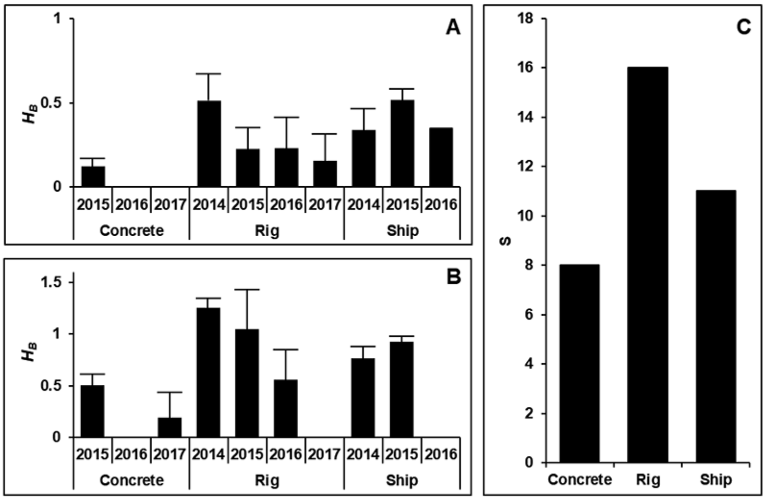

Figure 3. Mean (SE) for: (A) diversity (H_B) using VLL, (B) diversity (H_B) using fish traps, and (C) total species richness (*S*) for all fishes collected on concrete, rig, and ship artificial reefs in the NW GOM.

rig, 28.42 °C (SD 1.54); ship, 28.27 °C (SD 1.53); $\chi^2_{2,24} = 3.417$, $P = 0.181$), and distance from shore [concrete, 37.05 km (SD 3.69); rig, 38.89 km (SD 8.60); ship, 33.64 km (SD 23.55); $\chi^2_{2,24} = 4.444$, $P = 0.108$) did not significantly differ. However, depth was significantly different among the reef types ($\chi^2_{2,24} = 18.36$, $P = 0.001$) with concrete reefs shallower in depth [13.8 m (SD 1.2)] than both rig [20.9 m (SD 0.7)] and ship [23.9 m (SD 10.1)] reefs. A total of 1185 individuals were collected from VLL and trap surveys, and dominant species included red snapper (66.8%), gray triggerfish (11.0%), pigfish (*Orthopristis chrysoptera*; 5.9%), tomtate (*Haemulon aurolineatum*; 5.7%), hardhead catfish (4.8%), lane snapper (*Lutjanus synagris*; 1.3%), Atlantic croaker (*Micropogonias undulatus*; 1.0%), sand seatrout (*Cynoscion arenarius*; 1.0%), pinfish (*Lagodon rhomboides*; 0.74%), Atlantic spadefish (*Chaetodipterus faber*; 0.6%), and oyster toadfish (*Opsanus tau*; 0.3%). Rare species collected from our artificial reef surveys included black drum (*Pogonias cromis*), common remora (*Remora remora*), blue runner (*Caranx crysos*), Atlantic sharpnose shark (*Rhizoprionodon terraenovae*), gray snapper (*Lutjanus griseus*), southern kingfish (*Menticirrhus americanus*), gag grouper (*Mycteroperca microlepis*), Atlantic bumper (*Chloroscombrus chrysurus*), and spinner shark (*Carcharhinus brevipinna*; Table 3 and 4). Overall species richness (*S*) was highest on rig reefs with 16 species, followed by ship reefs with 11, and smallest on concrete reefs where only eight species were observed (Fig. 3). Individual reef species accumulation curves showed that for VLL sampling only ship reefs failed to reach asymptote, indicating further sampling was needed to estimate the reef fish assemblages. Using trap sampling, it appears that across all three reefs sampling reached asymptote, indicating that sampling was sufficient to describe the reef fish assemblages (Fig. 4).

Of the three species (red snapper, gray triggerfish, and hardhead catfish) analyzed for variation in mean size between gear types, both red snapper ($\chi^2_{1,724} = 330.9$, $P < 0.001$) and gray triggerfish ($\chi^2_{1,127} = 31.60$, $P < 0.001$) captured by VLL [373 mm (SD

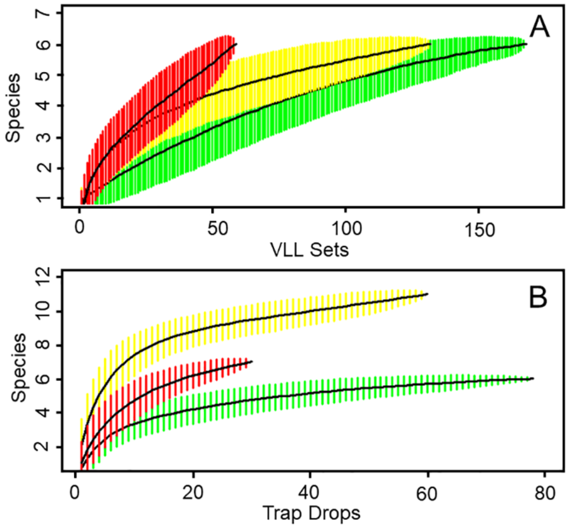

Figure 4. Species accumulation curves (95% CI) for all three reef types (concrete = green, rig = yellow, ships = red) using (A) VLL and (B) fish traps.

76) and 358 mm (SD 49), respectively] were significantly larger than those captured in fish traps [251 mm (SD 54) and 270 mm (SD 74), respectively] (Tables 3 and 4). For hardhead catfish, no difference was found in mean sizes collected between gear types ($\chi^2_{1,53} = 0.33$, $P = 0.565$). VLL hook size had a significant effect on red snapper size ($\chi^2_{3,433} = 49.12$, $P < 0.001$). Size 15/0 hooks caught the largest red snapper [410 mm (SD 73)] compared to 2/0 ($P < 0.001$), 8/0 ($P < 0.001$), and 11/0 hooks ($P < 0.001$). The hooks with the second largest median red snapper were 11/0 hooks [346 mm (SD 75)] when compared to 2/0 ($P = 0.040$) and 8/0 hooks ($P = 0.003$). Size 8/0 hooks and 2/0 hooks did not significantly differ ($P = 0.128$) in fish size [8/0, 321 mm (SD 55); 2/0, 326 mm (SD 62)]. Gray triggerfish size comparisons were not statistically comparable due to the fact that gray triggerfish were collected almost exclusively on size 2/0 hooks (n = 18), compared to 8/0 (n = 3), 11/0 (n = 1), and 15/0 (n = 1) hooks.

Fishing Surveys.—Total CPUE varied among reef types and years between the two gear types. VLL CPUE (individuals set^{-1}) did not differ between reef types (pseudo-$F_{2,187} = 0.815$, $P = 0.492$), years ($F_{3,187} = 1.557$, $P = 0.204$), or their interaction (pseudo-$F_{5,187} = 1.769$, $P = 0.158$). Trap CPUE [individuals (ind) soak^{-1}] did vary by both reef type (pseudo-$F_{2,101} = 5.635$, $P < 0.001$) and year (pseudo-$F_{3,160} = 6.380$, $P = 0.007$), and had a nonsignificant interaction (pseudo-$F_{5,160} = 0.186$, $P = 0.935$). CPUE for traps was highest on rigs [6.2 ind soak^{-1} (SD 6.8)] when compared to concrete [1.9 ind soak^{-1} (SD 3.0)] and ship reefs [2.6 ind soak^{-1} (SD 4.11)], while the latter reef types did not differ. Among years sampled, 2014 had significantly higher total CPUE for traps [7.0 ind soak^{-1} (SD 7.2)] compared to 2015 [3.3 ind soak^{-1} (SD 4.5)], 2016 [2.0 ind soak^{-1} (SD 4.2)], and 2017 [1.4 ind soak^{-1} (SD 1.8)], which did not differ respectively (Fig. 5).

Fish assemblages collected from VLL surveys significantly differed across reef type (pseudo-$F_{2,184} = 4.649$, $P = 0.008$), years (pseudo-$F_{3,184} = 2.737$, $P = 0.031$), and their interaction (pseudo-$F_{4,184} = 2.137$, $P = 0.046$). Due to the significant interaction

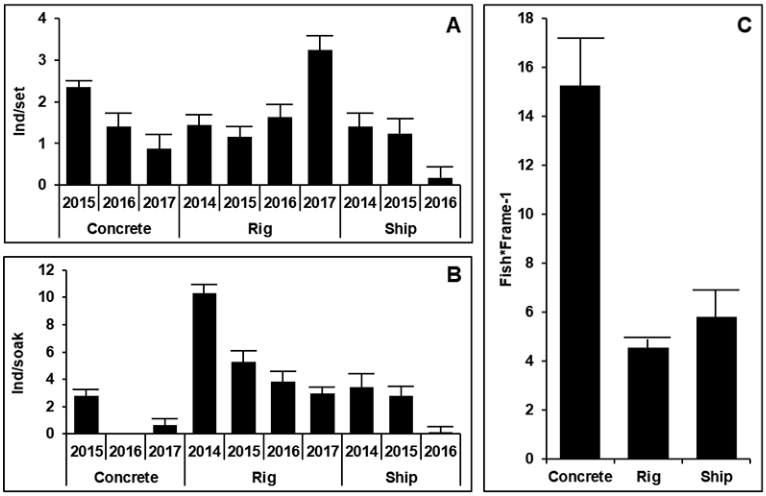

Figure 5. Mean (SE) for: (A) total fish CPUE (ind set^{-1}) using VLL, (B) total fish CPUE (ind soak^{-1}) using fish traps, and (C) Fish frame^{-1} using ARIS for concrete, rig, and ship artificial reefs in the NW GOM.

between reef type and years, reef types were compared nested within years (Table 5). Results from betadisper analysis showed that there were significant effects of dispersion within the PERMANOVA tests across reef types ($F_{2,184}$ = 8.907, P = 0.001) but not across years ($F_{3,184}$ = 1.814, P = 0.134). Further investigation of dispersion effects revealed that only reef types sampled within 2015 had significant dispersion effects ($F_{2,99}$ = 3.659, P = 0.029), which may have altered PERMANOVA results. There were significant differences in fish assemblages caught in VLL surveys on concrete reefs compared to rig and ship reefs in 2015 and also between concrete and ship reefs in 2016 (Table 5). No other significant differences existed in fish assemblages among the reef types using VLL. Year significantly differed when nested within reef type

Table 5. Results from PERMANOVA test nesting reef type within year. Significant differences ($\alpha < 0.05$) in fish assemblages between factors are in bold.

Reef Types	VLL Surveys			Trap Surveys		
	df	*t*	Approximate *P*	df	*t*	Approximate *P*
2014						
Rig × Ship	31	1.405	0.138	24	1.617	0.052
2015						
Rig × Ship	26	0.495	0.648	18	1.585	0.072
Rig × Concrete	91	3.163	**0.001**	46	2.060	**0.011**
Concrete × Ship	81	3.336	**0.002**	40	1.875	**0.040**
2016						
Rig × Ship	20	1.645	0.138	9	2.040	**0.029**
Rig × Concrete	29	1.067	0.305	- - -	- - -	- - -
Concrete × Ship	11	2.966	**0.035**	- - -	- - -	- - -
2017						
Rig × Concrete	25	1.115	0.268	8	4.142	**0.001**

Table 6. Results from PERMANOVA test nesting year within reef type. Significant differences ($\alpha < 0.05$) in fish assemblages between factors are in bold.

Years	VLL Surveys			Trap Surveys		
	df	*t*	Approximate *P*	df	*t*	Approximate *P*
Concrete						
2015 × 2016	83	0.532	0.826	- - -	- - -	- - -
2015 × 2017	81	0.816	0.553	37	1.857	**0.035**
2016 × 2017	18	1.435	0.171	- - -	- - -	- - -
Rig						
2014 × 2015	36	1.616	0.094	29	1.125	0.300
2014 × 2016	37	0.896	0.446	26	2.326	**0.003**
2014 × 2017	35	1.900	**0.033**	22	2.513	**0.002**
2015 × 2016	37	0.995	0.392	21	1.700	**0.036**
2015 × 2017	35	1.634	0.13	17	2.561	**0.003**
2016 × 2017	36	1.045	0.345	14	2.918	**0.003**
Ship						
2014 × 2015	21	0.629	0.565	13	0.596	0.530
2014 × 2016	14	1.450	0.135	7	1.916	0.023
2015 × 2016	9	1.545	0.128	6	2.745	0.005

only for rig reefs where the assemblages among all rig reefs in 2014 differed from all rig reefs in 2017 (Table 6). Fish assemblages also significantly differed among sites when nested within reef type (pseudo-$F_{6,194}$ = 2.987, approximate P = 0.005) from samples collected with VLL, with site-specific differences at the rig reefs across all years sampled.

Using SIMPER, red snapper and gray triggerfish were the two species that primarily drove the differences among reef types and years using VLL. Red snapper were more abundant on concrete reefs [1.64 ind set^{-1} (SD 2.00)] than rig reefs [1.52 ind set^{-1} (SD 2.12); average dissimilarity 37.10% (SD 1.48)] or ship reefs [0.88 ind set^{-1} (SD 1.45); average dissimilarity 29.97% (SD 1.40)]. Conversely, gray triggerfish were less abundant on concrete reefs [0.01 ind set^{-1} (SD 0.11)] than rig reefs [0.13 ind set^{-1} (SD 0.51); average dissimilarity 6.53% (SD 0.51)] or ship reefs [0.17 ind set^{-1} (SD 0.50); average dissimilarity 9.54% (SD 0.66); *see* Fig. 6]. The average relative abundance of red snapper across all reef types collected with VLL doubled from 2014 [0.92 ind set^{-1} (SD 1.39)] to 2017 [1.96 ind set^{-1} (SD 2.60); average dissimilarity 38.62% (SD 1.68)]. Mean species diversity increased with structure size and reef type, and was lowest on concrete reefs [0.07 H_B (SD 0.11)], followed by rigs [0.30 H_B (SD 0.26)], and highest on ships [0.41 H_B (SD 0.14)]. Mean species diversity (H_B) using VLL also decreased every year of sampling starting in 2014 [0.37 H_B (SD 0.28)] followed by 2015 [0.22 H_B (SD 0.20)] and 2016 [0.18 H_B (SD 0.23)], with the lowest diversity in 2017 [0.08 H_B (SD 0.16); Fig. 3].

Fish assemblages estimated from trap surveys also differed across reef types (pseudo-$F_{2,102}$ = 5.828, P = 0.001), year (pseudo-$F_{3,102}$ = 2.219, P = 0.010), and their interaction (pseudo-$F_{4,102}$ = 3.755, P = 0.001). However, betadisper did not detect any effect of dispersion on the results of individual PERMANOVA for reef type ($F_{2,102}$ = 1.125, P = 0.325) or year ($F_{3,102}$ = 0.395, P = 0.786). Again, similarly to VLL, due to the significant interaction between reef type and year factors, differences in fish assemblages among reefs collected using traps were compared nesting reef type within years (Table 5).

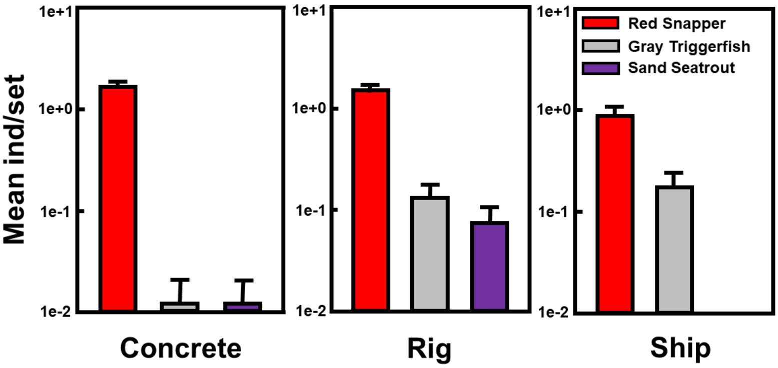

Figure 6. Mean (SE) for CPUE (ind set^{-1}) using VLL for the three species collected on at least two of the three reef types (concrete, rig, and ship).

There were significant differences in fish assemblages caught in trap surveys on concrete reefs compared to rig reefs and ships in 2015 and between concrete and rig reefs in 2017 (Table 5). The only other difference between fish assemblages collected in traps were between rig and ship reefs in 2016 (Table 5). Fish assemblages collected in traps significantly differed among years when nested within reef type only for concrete reefs between years 2015 and 2017 (Table 6). Fish assemblages in traps sampled from rig reefs differed for every year sampled except for 2014 when compared to 2015 (Table 6). Fish assemblages also significantly differed when site was nested within reef type (pseudo-$F_{6,102}$ = 4.243, P = 0.001), with differences namely occurring on concrete and rig reefs.

Fish assemblages in trap surveys were significantly more diverse than VLL surveys and differences in assemblages were driven by multiple species, highlighted using SIMPER. In contrast to results using VLL, fish assemblages in traps for concrete reefs [1.0 ind soak^{-1} (SD 2.3)] had lower abundances of red snapper than rig reefs [2.5 ind soak^{-1} (SD 3.8); average dissimilarity 26.18% (SD 1.14)] or ship reefs [1.1 ind soak^{-1} (SD 1.8); average dissimilarity 29.02% (SD 1.43)]. Differences in relative abundances of gray triggerfish also differed between rig and ship reefs using fish traps and was higher on rig reefs [0.95 ind soak^{-1} (SD 1.97)] than on ships [0.47 ind soak^{-1} (SD 1.12); average dissimilarity 15.74% (SD 0.83)]. Hardhead catfish were more abundant in fish traps fished on concrete reefs [0.42 ind soak^{-1} (SD 0.78)] compared to rig reefs [0.32 ind soak^{-1} (SD 0.79); average dissimilarity 6.12% (SD 0.54)] and ship reefs [0.03 ind soak^{-1} (SD 0.18); average dissimilarity 13.39% (SD 0.72); Fig. 7]. The number of red snapper collected in fish traps was the major difference among years. Red snapper relative abundance was highest in 2014 [2.84 ind soak^{-1} (SD 3.99)] relative to 2015 [1.44 ind soak^{-1} (SD 2.52); average dissimilarity 22.55% (SD 1.17)] on all reefs. Red snapper relative abundance declined both years after 2015, in 2016 [0.86 ind soak^{-1} (SD 3.07); average dissimilarity 23.28% (SD 1.01)], and 2017 [1.00 ind soak^{-1} (SD 1.78); average dissimilarity 32.71% (SD 1.33)]. The overall trends in declining species diversity were similar in both trap surveys and VLL surveys, as both decreased over the course of the study. Mean species diversity among reef types was higher on rig reefs [0.87 H_B (SD 0.58)], relative to ship reefs [0.68 H_B (SD 0.40)], and concrete reefs [0.36 H_B (0.32)]. Additionally, mean species diversity (H_B) was highest in 2014 [1.07 H_B (SD

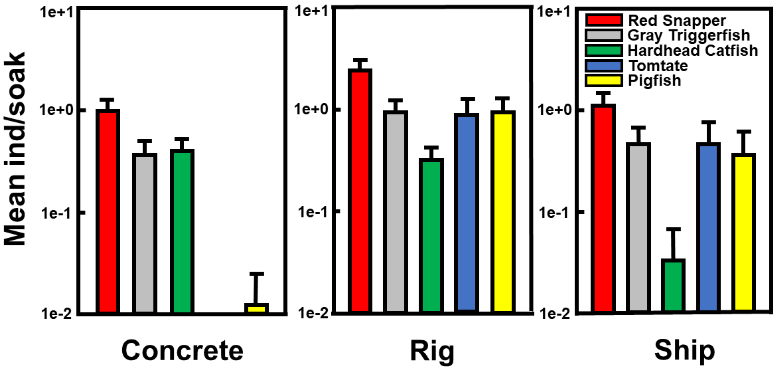

Figure 7. Mean (SE) CPUE (ind soak^{-1}) using fish traps for the five species collected on at least two of the three reef types (concrete, rig, and ship).

0.31)] followed by 2015 [0.72 H_B (SD 0.42)] and 2016 [0.28 H_B (SD 0.44)], with the lowest diversity in 2017 [0.15 H_B (SD 0.25); Fig. 3].

Acoustic Surveys.—Relative fish density (fish frame^{-1}) estimated using ARIS transects significantly differed among reef type ($\chi^2 = 55.53$, $P < 0.001$). Acoustic surveys conducted on concrete structures, rigs, and ships contained 56,796 frames, of which 90.1% (n = 51,173) met the criteria for analyses, leading to 1023 (2%) frames being analyzed. Pairwise analyses show significant differences exist among all reef types ($P < 0.001$). Fish relative density was highest on concrete structures [15.31 fish frame^{-1} (SD 26.83)], followed by ships [5.84 fish frame^{-1} (SD 13.55)], and then rigs [4.52 fish frame^{-1} (SD 10.59); Fig. 5]. Individual differences existed among sites within reef type ($\chi^2 = 97.1$, $P < 0.001$), with pairwise differences among concrete B and the other two concrete reef sites.

Redundancy Analysis.—Gear type, reef type, salinity, temperature, depth, and distance from shore were significantly correlated with canonical axes (pseudo-F = 15.9, $P = 0.002$). Eigenvalues for the first four multivariate axes were 0.090 (Axis 1), 0.021 (Axis 2), 0.013 (Axis 3), and 0.004 (Axis 4). Pseudo-canonical correlation coefficients for the four axes were 0.538, 0.298, 0.336, and 0.161, respectively. The total cumulative percentage of variance explained by the first four axes was 12.39%. Axes 1 and 2 accounted for 9.04% and 2.10% of the variation, respectively, and 99.12% of the cumulative variance modeled by the RDA (Fig. 8). Fishes were ordinated in multivariate space proportionally along the gradients that most affected their abundance. Gray triggerfish, tomtates, lane snapper, and pinfish had positive relationships with rig and ship reefs, increasing depth and salinity, as well as being primarily collected using fish traps. Several other species consisting of oyster toadfish, pigfish, Atlantic croaker, hardhead catfish, and Atlantic spadefish were also positively associated with trap collections with similar numbers between rig and concrete reefs. Red snapper were positively associated with concrete reefs and VLLs. Although not associated with specific reef types, sand seatrout were primarily collected using VLL.

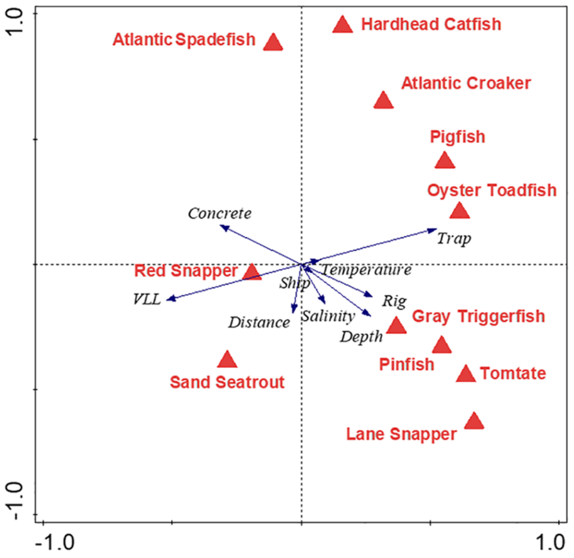

Figure 8. Plot of fish species and redundancy analysis (RDA) scores on the first two axes, with gear type (VLL or fish trap), reef type (concrete, rig, and ship), salinity, temperature, depth, and distance from shore as factors relative to species abundance.

Discussion

Results of this study highlight fish assemblages on artificial reefs in the NW GOM and how they differ according to structure type. Reefs constructed from decommissioned rig jackets and ships were shown to have higher CPUE for multiple species and higher diversity, but lower relative density than low-relief concrete structures. The size and relief of artificial structure has been hypothesized to have a positive effect on fish assemblage diversity by providing additional structure usable by pelagic fishes that are not normally associated with benthic habitats (Rilov and Benayahu 2000, Komyakova et al. 2019, Lemoine et al. 2019). Our findings also confirm a previously observed relationship for more diverse assemblages being found on larger, higher relief structures. Bohnsack et al. (1994), using visual surveys, observed that larger, higher-relief habitats have lower fish densities than low-relief and smaller habitats. Our ARIS surveys of relative fish density similarly showed that the highest densities existed on smaller, low-relief concrete habitats relative to larger, more rugose structures. This result, using ARIS, differed from both VLL and trap surveys which indicated that the relationships between habitat and relative fish abundance varied among years. This unique description of the fish assemblages on nearshore artificial reefs in the NW GOM illustrates the utility of using three complimentary techniques to describe the differences in assemblages driven by the structure of individual reefs.

Several estuarine-associated fish species were collected over artificial reefs during the course of this study. Estuarine-associated species included four sciaenids (Atlantic croaker, sand seatrout, black drum, and southern kingfish), oyster toadfish, pigfish, and pinfish. Benthic substrate on the inner continental shelf in the NW

GOM has limited hard structure (e.g., natural reef) and is comprised primarily of mud and sand with interspersed natural shell rubble banks (Rezak et al. 1990). A study by Wells et al. (2009) over a natural drowned barrier island in the NW GOM indicated that the most abundant species collected were predominantly estuarine associated fishes (e.g., sciaenids). Fish assemblages on naturally occurring unconsolidated sand or mud bottom in close proximity to artificial reefs indicate that there is substantial connectivity with surrounding habitat acting as a source of biodiversity for artificial reefs (Bohnsack 1989). Bohnsack et al. (1994) found that artificial reefs had fish assemblages comprised of species from both surrounding natural reefs and surrounding unconsolidated sand habitat. Additionally, more recent work has observed strong connectivity among nearshore habitats and artificial structure. Reeves et al. (2018) in a study on fish assemblages that utilize oil and gas platforms also noted high proportions of nearshore reef-associated fishes implying high levels of connectivity between natural oyster reef and artificial structure. Fish assemblages on artificial reefs have been described as having high levels of transience with “settled” fishes accounting for very little biomass (5.7%; Bohnsack et al. 1994). This connectivity is an important consideration for the implementation of artificial reefs and is often the goal of managers when weighing potential benefits to regional fisheries (Pickering and Whitmarsh 1997). This supplementation of nearshore artificial habitat in the NW GOM can potentially increase connectivity among fish assemblages that utilize a diversity of adjacent habitats including unconsolidated sand or mud bottom and reef.

Fish assemblages on artificial reefs observed in this study were similar to those observed in other regions of the GOM using similar gear. Species richness on artificial reefs has been reported to be high in visual surveys (e.g., ROV surveys) in the NE and NW GOM (Stanley and Wilson 2000, Dance et al. 2011, Ajemian et al. 2015a, Reeves et al. 2018). However, traditional fisheries sampling with entrapment or entanglement gear reduces diversity estimates due to the effect of gear bias (Connell et al. 1998), as was the case in this study. VLLs are an effective tool for collecting predatory marine fishes such as red snapper (Streich et al. 2018). Gregalis et al. (2012), using VLLs, found that the fish assemblages of artificial reefs off the coast of Alabama were dominated by red snapper, comprising 87% of total catch (with 19 other species comprising the remaining 13%). Similarly, in this study, red snapper comprised 90.7% of all fishes caught on VLLs, while 11 other species comprised the remainder of the total catch. Fish traps were effective at collecting more diverse fish assemblages than VLLs. Likewise, over the course of post-reefing surveys on prefabricated concrete pyramids off the coast of Texas, small fish traps had a species richness of 14 species, while VLLs only collected 7 species (Streich et al. 2017a). While small fish traps have been shown to collect diverse fish assemblages, albeit collecting smaller fishes than VLLs, red snapper was still by far the most dominant species in fish traps, comprising 43.9% of all fishes caught. High proportions of red snapper on artificial reef assemblages in the northern GOM have been well documented using alternative methods (e.g., ROV, SCUBA surveys; Wells and Cowan 2007, Redman and Szedlmayer 2009, Dance et al. 2011). Red snapper have also been shown to have higher densities on artificial structures relative to other structures (e.g., natural reefs and unconsolidated sand and mud bottom; Karnauskas et al. 2017, Streich et al. 2017b), and their high densities may play a role in shaping the fish assemblages on artificial reefs in the northern GOM.

Structures and/or materials used for artificial reefs are often a product of opportunity, utilizing available structure with the lowest cost of implementation that will produce the greatest desired effect (Baine 2001). Large artificial structures are costly to implement; however, this cost can be offset by using existing decommissioned marine structures (Dafforn et al. 2015). To maximize the desired effects of artificial reefs (e.g., marine community enhancement), specific design features need to be incorporated such as vertical relief, reef footprint, and rugosity. In this study, reefs with the highest diversity were large and had higher vertical relief, which is a trend that has been described in studies examining reef fish assemblages (Bohnsack et al. 1994, Rooker et al. 1997). A study by Rilov and Benayahu (2000) observed that an increase in vertical relief resulted in a higher proportion of pelagic fishes that utilized the artificial reef vs low-relief habitats which had smaller proportions of pelagic fishes. In Ajemian et al. (2015a), the authors observed differences in fish assemblages among reef types when looking at differences among ships and toppled, cutoff, and freestanding rig platforms. These differences were a function of the combination of water depth and vertical relief, with water depth being the primary factor in determining fish assemblages (Ajemian et al. 2015a). Water depth in this study varied from 13 to 32 m, with the mean depth of concrete reef sites [13.8 m (SD 1.2)] being significantly shallower than rig [20.9 m (SD 0.7)] or ship [23.9 m (SD 10.1)] reef sites, which may have affected the fish assemblages that were observed. This difference in depth may have affected the diversity of the fish assemblages. Additionally, both rigs and ships had higher diversity than concrete reefs across both fishing gear types and individual relationships among species to depth appeared to be species-specific, which may have also affected fish assemblage diversity (Fig. 8). Reef type was associated with depth, as it was necessary for high-relief structures to be situated in deeper water to prevent posing a hazard to maritime navigation (Peter et al. 2003). Water depth is highly correlated with distance from shore and both may affect fish assemblages, as previously shown. Therefore, we were not able to make strong conclusions that changes in fish assemblages were exclusively a result of structure type and not a result of location or depth, due to similar reefs being clustered in similar locations.

The relative fish density from ARIS surveys was almost three times higher on low-lying artificial reefs relative to ship or rig reefs, which may be due to differences in habitat rugosity and complexity. Habitat complexity has been linked to the ability to observe fishes on natural reefs due to the ability for fishes to seek refuge in more complex habitats. Reef habitats with a more rugose topography were strongly correlated with lower species richness estimates during visual diver surveys (Wilson et al. 2007). Furthermore, ARIS surveys produce mixed results when used in environments with potential obstructions to the field of view, evidenced by the density estimates on low-lying concrete reefs relative to rig jackets and ships. Demersal and sedentary fishes, or species that closely associate with structure, were unlikely to be counted using acoustic surveys in estuaries (Able et al. 2014), which may also be true for highly complex benthic structures that are in offshore marine environments. The ARIS sonar has a narrow-fixed beam angle which can cover an entire quarry block or reef pyramid at a time, while the high-relief habitats surveyed (rig jackets and decommissioned ships) have a much larger footprint and do not fit within a single frame. In this study, using ARIS, low-lying concrete reefs had higher observed relative fish abundances compared to both rig and ship reefs, which was counter to the results of fishing surveys where the highest relative fish abundances were found on rig reefs. This result may be due to high fish densities on smaller, less complex concrete reefs, or sampling bias due to the narrow-fixed width of the ARIS field of view. In future studies, caveats such as structure size and complexity need to be taken into consideration when interpreting the relative fish density on artificial reefs.

The varied results in artificial reef fish assemblages among different gear types provide additional evidence for the need to use multiple gears when examining fish abundance. Gear bias is a common issue throughout fisheries surveys when estimating the relative abundance and size of multiple species (Jackson and Harvey 1997). Wells et al. (2008) found that across four gear types, there was distinct size selectivity and bias involved in each; demersal trawls were the most effective tool for collecting high numbers of juvenile red snapper, but were less effective at collecting larger individuals which were more successfully sampled using chevron traps and underwater camera arrays. With exclusively passive gear types, the effectiveness of the gear in our study was based on our ability to be proximate to structure, baiting fishes. However, the act of baiting can specifically attract carnivorous or omnivorous fishes (Løkkeborg 1990) and does less to attract planktivorous or herbivorous fishes, which have been shown to comprise a large portion of fish assemblages in studies that utilize visual surveys on artificial reefs (Rooker et al. 1997, Dance et al. 2011). For example, studies estimating reef fish assemblages on reefs in Louisiana and Texas noted that the reef fish assemblages were dominated by Atlantic spadefish (which were relatively rare within our estimation of assemblages), which was most likely an effect of gear type (Gallaway and Lewbel 1982, Stanley and Wilson 2000, Reeves et al. 2018). Additionally, aside from a few limited instances, planktivores and herbivores were not encountered in VLL or trap sampling, which was likely an artifact of using baited gear in this study. Combining baited gear types, which may be size- and species-specific, with acoustic surveys, that can provide taxonomic resolution, is necessary to fully describe the fish assemblages on artificial reef habitats.

Temporal and regional differences in fish assemblages are common and hypothesized to be driven by a combination of biotic and abiotic factors (Attrill and Power 2002). These changes in fish assemblages can be much stronger in coastal areas that have increased seasonal variability in salinity and temperature (Feyrer et al. 2015). Sites representing reef types were spatially distinct with concrete sites being located in the northern portion, ships in the southern portion, and rigs in the central portion of the sampling area. Neves dos Santos et al. (2005) conducted fish assemblage surveys on low-relief artificial reefs along the southern coast of Portugal, examining the differences between two reef locations. Each location was spatially distinct with varied differences in abiotic variables (salinity and temperature) and proximity to source habitat (estuaries and natural reefs) which was shown to alter fish assemblages, with higher biomass and species richness on reefs closer to the estuary (Neves dos Santos et al. 2005). For the reefs used in this study, neither salinity nor temperature significantly differed among sites. The sites sampled in this study were spatially distinct varying in distance to the nearest estuary as well as being surveyed exclusively in the summer (which may have ignored seasonal assemblage changes common in coastal waters). Estimating reef fish assemblages is challenging when both temporal and spatial factors vary within the sampling design. Future studies should attempt to include these factors in investigations into artificial reef fish assemblages in coastal ecosystems.

This study highlights the importance of using multiple gear types to sample fish assemblages associated with artificial reefs due to size selectivity and biases involved with certain gear types. Reef fish assemblages in shallow coastal environments of the NW GOM are similar to adjacent regions and are comprised of both reef-associated fishes and transient fishes that utilize surrounding unconsolidated sand and mud bottom habitats. Artificial structures that had both high relief and rugosity promoted increased assemblage diversity, while low-relief habitats supported higher relative fish density based on acoustic surveys. However, low-relief artificial reefs had lower assemblage diversity than higher-relief habitats. While these results indicate that structure type plays a significant role in reef fish assemblage and density, our project design was potentially confounded by lack of depth and structure location among our reef sites. Findings from this study can be used to inform the planning of future artificial reefs in the northern Gulf of Mexico and elsewhere. The information disseminated here should help planning organizations prioritize habitat structure based on their needs while understanding their effect on the assemblages of reef-associated fishes. As depth and structure height are often a concern for managers, we also suggest that reef structure and size might be as well. It is unclear how higher densities on smaller artificial reefs may affect individual fishes; however, our work may indicate that smaller, more dispersed artificial reef habitats may increase abundances of certain fish species (e.g., red snapper) compared to larger, centralized artificial reefs, which may have a greater effect on overall species diversity. Future work examining fish assemblages associated with artificial reefs should focus on developing methods to balance shortcomings of individual techniques and include other visual surveying techniques to better estimate the relative abundance of reef fishes.

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Literature Cited

Able KW, Grothues TM, Rackovan JL, Buderman FE. 2014. Application of mobile dual-frequency identification sonar (DIDSON) to fish in estuarine habitats. Northeast Nat (Steuben). 21(2):192–209. https://doi.org/10.1656/045.021.0207

Abràmoff MD, Magalhães PJ, Ram SJ. 2004. Image processing with ImageJ. Biophoton Int. 11(7):36–42.

Ajemian MJ, Wetz JJ, Shipley-Lozano B, Shively JD, Stunz GW. 2015a. An analysis of artificial reef fish community structure along the northwestern Gulf of Mexico shelf: potential impacts of "rigs-to-reefs" programs. PLoS One. 10(5):e0126354. https://doi.org/10.1371/journal.pone.0126354

Ajemian MJ, Wetz JJ, Shipley-Lozano B, Stunz GW. 2015b. Rapid assessment of fish communities on submerged oil and gas platform reefs using remotely operated vehicles. Fish Res. 167:143–155. https://doi.org/10.1016/j.fishres.2015.02.011

Arney RN, Froehlich CYM, Kline RJ. 2017. Recruitment patterns of juvenile fish at an artificial reef area in the Gulf of Mexico. Mar Coast Fish. 9(1):79–92. https://doi.org/10.1080/19425120.2016.1265031

Attrill MJ, Power M. 2002. Climatic influence on a marine fish assemblage. Nature. 417:275–278. https://doi.org/10.1038/417275a

Baine M. 2001. Artificial reefs: A review of their design, application, management and performance. Ocean Coast Manage. 44(3–4):241–259. https://doi.org/10.1016/S0964-5691(01)00048-5

Bohnsack JA. 1989. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference. Bull Mar Sci. 44(2):631–645.

Bohnsack JA, Harper DE, McClellan DB, Hulsbeck M. 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, U.S.A. Bull Mar Sci. 55(2–3):796–823.

Bollinger MA, Kline RJ. 2017. Validating sidescan sonar as a fish survey tool over artificial reefs. J Coast Res. 33(6):1397–1407. https://doi.org/10.2112/JCOASTRES-D-16-00174.1

Boswell KM, Wells RJD, Cowan JH, Wilson CA. 2010. Biomass, density, and size distributions of fishes associated with a large-scale artificial reef complex in the Gulf of Mexico. Bull Mar Sci. 86(4):879–889. https://doi.org/10.5343/bms.2010.1026

Brandt JR, Jackson DC. 2013. Influences of artificial reefs on juvenile red snapper along the Mississippi gulf coast. Mar Coast Fish. 5(1):1–10. https://doi.org/10.1080/19425120.2012.736445

Brillouin L. 1962. Science and information theory. Oxford, England: Dover Publications.

Clarke KR, Gorley RN. 2015. Primer v.7: user manual and tutorial. Plymouth, UK: PRIMER-E.

Connell SD, Samoilys MA, Lincoln Smith MP, Leqata J. 1998. Comparisons of abundance of coral-reef fish: catch and effort surveys vs visual census. Austral Ecol. 23(6):579–586. https://doi.org/10.1111/j.1442-9993.1998.tb00768.x

Dafforn KA, Glasby TM, Airoldi L, Rivero NK, Mayer-Pinto M, Johnston EL. 2015. Marine urbanization: an ecological framework for designing multifunctional artificial structures. Front Ecol Environ. 13(2):82–90. https://doi.org/10.1890/140050

Dance MA, Patterson WF, Addis DT. 2011. Fish community and trophic structure at artificial reef sites in the northeastern Gulf of Mexico. Bull Mar Sci. 87(3):301–324. https://doi.org/10.5343/bms.2010.1040

Diaz RJ, Cutter GR, Able KW. 2003. The importance of physical and biogenic structure to juvenile fishes on the shallow inner continental shelf. Estuaries. 26(1):12–20. https://doi.org/10.1007/BF02691689

Feyrer F, Cloern JE, Brown LR, Fish MA, Hieb KA, Baxter RD. 2015. Estuarine fish communities respond to climate variability over both river and ocean basins. Glob Change Biol. 21(10):3608–3619. https://doi.org/10.1111/gcb.12969

Gallaway BJ, Lewbel GS. 1982. The ecology of petroleum platforms in the northwestern Gulf of Mexico: a community profile. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, D.C. FWS/OBS-82/27. Bureau of Land Management, Gulf of Mexico OCS Regional Office. Open-File Report 82–03.

Gregalis KC, Schlenker LS, Drymon JM, Mareska JF, Powers SP. 2012. Evaluating the performance of vertical longlines to survey reef fish populations in the northern Gulf of Mexico. Trans Am Fish Soc. 141(6):1453–1464. https://doi.org/10.1080/00028487.2012.703154

Grote AB, Bailey MM, Zydlewski JD, Hightower JE. 2014. Multibeam sonar (DIDSON) assessment of American shad (*Alosa sapidissima*) approaching a hydroelectric dam. Can J Fish Aquat Sci. 71(4):545–558. https://doi.org/10.1139/cjfas-2013-0308

Holmes JA, Cronkite GMW, Enzenhofer HJ, Mulligan TJ. 2006. Accuracy and precision of fish-count data from a "dual-frequency identification sonar" (DIDSON) imaging system. ICES J Mar Sci. 63(3):543–555. https://doi.org/10.1016/j.icesjms.2005.08.015

Hothorn T, Bretz F, Westfall P. 2008. Simultaneous inference in general parametric models. Biom J. 50(3):346–363. https://doi.org/10.1002/bimj.200810425

Jackson DA, Harvey HH. 1997. Qualitative and quantitative sampling of lake fish communities. Can J Fish Aquat Sci. 54(12):2807–2813. https://doi.org/10.1139/f97-182

Jolly GM, Hampton I. 1990. A stratified random transect design for acoustic surveys of fish stocks. Can J Fish Aquat Sci. 47(7):1282–1291. https://doi.org/10.1139/f90-147

Karnauskas M, Walter JF, Campbell MD, Pollack AG, Drymon JM, Powers S. 2017. Red snapper distribution on natural habitats and artificial structures in the northern Gulf of Mexico. Mar Coast Fish. 9(1):50–67. https://doi.org/10.1080/19425120.2016.1255684

Kennicutt MC. 2017. Water quality of the Gulf of Mexico. *In:* Ward CH, editor. Habitats and biota of the Gulf of Mexico: Before the Deepwater Horizon oil spill. New York, NY: Springer New York. p. 55–164.

Komyakova V, Chamberlain D, Jones GP, Swearer SE. 2019. Assessing the performance of artificial reefs as substitute habitat for temperate reef fishes: implications for reef design and placement. Sci Total Environ. 668:139–152. https://doi.org/10.1016/j.scitotenv.2019.02.357

Lemoine HR, Paxton AB, Anisfeld SC, Rosemond RC, Peterson CH. 2019. Selecting the optimal artificial reefs to achieve fish habitat enhancement goals. Biol Conserv. 238:108200. https://doi.org/10.1016/j.biocon.2019.108200

Løkkeborg S. 1990. Rate of release of potential feeding attractants from natural and artificial bait. Fish Res. 8(3):253–261. https://doi.org/10.1016/0165-7836(90)90026-R

Makabe R, Kurihara T, Uye SI. 2012. Spatio-temporal distribution and seasonal population dynamics of the jellyfish *Aurelia aurita* s.l. studied with dual-frequency identification sonar (DIDSON). J Plankton Res. 34(11):936–950. https://doi.org/10.1093/plankt/fbs057

Moursund RA, Carlson TJ, Peters RD. 2003. A fisheries application of a Dual-Frequency Identification Sonar acoustic camera. ICES J Mar Sci. 60(3):678–683. https://doi.org/10.1016/S1054-3139(03)00036-5

Mueller A-M, Burwen DL, Boswell KM, Mulligan T. 2010. Tail-beat patterns in dual-frequency identification sonar echograms and their potential use for species identification and bioenergetics studies. Trans Am Fish Soc. 139(3):900–910. https://doi.org/10.1577/T09-089.1

Mueller RP, Brown RS, Hop H, Moulton L. 2006. Video and acoustic camera techniques for studying fish under ice: a review and comparison. Rev Fish Biol Fish. 16(2):213–226. https://doi.org/10.1007/s11160-006-9011-0

Neves dos Santos M, Monteiro CC, Lasserre G. 2005. Observations and trends on the intra-annual variation of the fish assemblages on two artificial reefs in Algarve coastal waters (southern Portugal). Sci Mar. 69(3):415–426. https://doi.org/10.3989/scimar.2005.69n3415

Newman SJ, Williams DM. 1995. Mesh size selection and diel variability in catch of fish traps on the central Great Barrier Reef, Australia: a preliminary investigation. Fish Res. 23(3–4):237–253. https://doi.org/10.1016/0165-7836(94)00353-X

Oksanen J, Kindt R, Legendre P, O’Hara B, Stevens MHH, Oksanen MJ, Suggests M. 2007. The vegan package. Community ecology package. 10:631–637.

Peet RK. 1974. The measurement of species diversity. Annu Rev Ecol Syst. 5(1):285–307. https://doi.org/10.1146/annurev.es.05.110174.001441

Perkol-Finkel S, Shashar N, Benayahu Y. 2006. Can artificial reefs mimic natural reef communities? The roles of structural features and age. Mar Environ Res. 61(2):121–135. https://doi.org/10.1016/j.marenvres.2005.08.001

Peter DD, Culbertson JC, Shively JD, Hammerschmidt PC, Embesi JA. 2003. Texas artificial reef program: over 50 years of marine habitat enhancement in the Gulf of Mexico. Austin, Texas: Texas Parks and Wildlife Coastal Fisheries Division. 45 p.

Pickering H, Whitmarsh D. 1997. Artificial reefs and fisheries exploitation: a review of the ‘attraction versus production’ debate, the influence of design and its significance for policy. Fish Res. 31(1–2):39–59. https://doi.org/10.1016/S0165-7836(97)00019-2

R Core Team. 2014. R: a language and environment for statistical computing. R foundation for statistical computing, Vienna, Austria.

Redman RA, Szedlmayer ST. 2009. The effects of epibenthic communities on reef fishes in the northern Gulf of Mexico. Fish Manag Ecol. 16(5):360–367. https://doi.org/10.1111/j.1365-2400.2009.00684.x

Reeves DB, Chesney EJ, Munnelly RT, Baltz DM, Marx BD. 2018. Abundance and distribution of reef-associated fishes around small oil and gas platforms in the northern Gulf of Mexico's hypoxic zone. Estuaries Coasts. 41(7):1835–1847. https://doi.org/10.1007/s12237-017-0349-4

Reynolds EM, Cowan JH, Lewis KA, Simonsen KA. 2018. Method for estimating relative abundance and species composition around oil and gas platforms in the northern Gulf of Mexico, U.S.A. Fish Res. 201:44–55. https://doi.org/10.1016/j.fishres.2018.01.002

Rezak R, Bright TJ, McGrail DW. 1985. Reefs and banks of the northwestern Gulf of Mexico: their geological, biological, and physical dynamics. New York: Wiley.

Rezak R, Gittings SR, Bright TJ. 1990. Biotic assemblages and ecological controls on reefs and banks of the northwest Gulf of Mexico. Am Zool. 30(1):23–35. https://doi.org/10.1093/icb/30.1.23

Rilov G, Benayahu Y. 2000. Fish assemblage on natural versus vertical artificial reefs: the rehabilitation perspective. Mar Biol. 136(5):931–942. https://doi.org/10.1007/s002279900250

Rooker JR, Dokken QR, Pattengill CV, Holt GJ. 1997. Fish assemblages on artificial and natural reefs in the Flower Garden Banks National Marine Sanctuary, USA. Coral Reefs. 16(2):83–92. https://doi.org/10.1007/s003380050062

Rotherham D, Underwood AJ, Chapman MG, Gray CA. 2007. A strategy for developing scientific sampling tools for fishery-independent surveys of estuarine fish in New South Wales, Australia. ICES J Mar Sci. 64(8):1512–1516. https://doi.org/10.1093/icesjms/fsm096

Scott-Denton E, Cryer PF, Gocke JP, Harrelson MR, Kinsella DL, Pulver JR, Smith RC, Williams JA. 2011. Descriptions of the US Gulf of Mexico reef fish bottom longline and vertical line fisheries based on observer data. Mar Fish Rev. 73(2):1–26.

SEAMAP (Southeast Area Monitoring and Assessment Program). 2013. SEAMAP Vertical Line Survey Protocol. Ocean Springs, Mississippi: Gulf States Marine Fisheries Commission.

Shideler GL. 1981. Development of the benthic nepheloid layer on the south Texas continental shelf, western Gulf of Mexico. Mar Geol. 41(1–2):37–61. https://doi.org/10.1016/0025-3227(81)90103-1

Stanley DR, Wilson CA. 2000. Variation in the density and species composition of fishes associated with three petroleum platforms using dual beam hydroacoustics. Fish Res. 47(2–3):161–172. https://doi.org/10.1016/S0165-7836(00)00167-3

Streich MK, Ajemian MJ, Wetz JJ, Shively JD, Shipley JB, Stunz GW. 2017a. Effects of a new artificial reef complex on red snapper and the associated fish community: an evaluation using a before-after control-impact approach. Mar Coast Fish. 9(1):404–418. https://doi.org/10.1080/19425120.2017.1347116

Streich MK, Ajemian MJ, Wetz JJ, Stunz GW. 2017b. A comparison of fish community structure at mesophotic artificial reefs and natural banks in the western Gulf of Mexico. Mar Coast Fish. 9(1):170–189. https://doi.org/10.1080/19425120.2017.1282897

Streich MK, Ajemian MJ, Wetz JJ, Stunz GW. 2018. Habitat-specific performance of vertical line gear in the western Gulf of Mexico: a comparison between artificial and natural habitats using a paired video approach. Fish Res. 204:16–25. https://doi.org/10.1016/j.fishres.2018.01.018

Strelcheck AJ, Cowan JH, Shah A. 2005. Influence of reef location on artificial-reef fish assemblages in the northcentral Gulf of Mexico. Bull Mar Sci. 77(3):425–440.

ter Braak CJF, Smilauer P. 2012. Canoco reference manual and canodraw for windows user's guide: software for canonical community ordination (version 5.0). Ithaca NY: Microcomputer Power.

Wells RJD, Boswell KA, Cowan JH, Patterson WF. 2008. Size selectivity of sampling gears targeting red snapper in the northern Gulf of Mexico. Fish Res. 89(3):294–299. https://doi.org/10.1016/j.fishres.2007.10.010

Wells RJD, Cowan JH Jr. 2007. Video estimates of red snapper and associated fish assemblages on sand, shell, and natural reef habitats in the north-central Gulf of Mexico. Pages 39–57 *In* W. F. Patterson, III, J. H. Cowan, Jr., G. R. Fitzhugh, D. L. Nieland, editors. Red Snapper

Ecology and Fisheries in the U.S. Gulf of Mexico. American Fisheries Society, Symposium 60, Bethesda, Maryland.

Wells RJD, Harper JO, Rooker JR, Landry AM, Dellapenna TM. 2009. Fish assemblage structure on a drowned barrier island in the northwestern Gulf of Mexico. Hydrobiologia. 625(1):207–221. https://doi.org/10.1007/s10750-009-9709-9

Willis TJ, Millar RB, Babcock RC. 2000. Detection of spatial variability in relative density of fishes: comparison of visual census, angling, and baited underwater video. Mar Ecol Prog Ser. 198:249–260. https://doi.org/10.3354/meps198249

Wilson SK, Graham NAJ, Polunin NVC. 2007. Appraisal of visual assessments of habitat complexity and benthic composition on coral reefs. Mar Biol. 151(3):1069–1076. https://doi.org/10.1007/s00227-006-0538-3