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ARTICLE

Depth and Artificial Reef Type Effects on Size and Distribution of Red Snapper in the Northern Gulf of Mexico

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Abstract
The Red Snapper *Lutjanus campechanus* is a highly exploited commercial and recreational species that dominates the artificial reef systems in the northern Gulf of Mexico. Off coastal Alabama there are few natural reefs, but in the last 50 years government programs and private fishers have placed numerous artificial reefs in the northeast Gulf of Mexico with the goal of increasing fisheries production. We examined the effects of artificial reef depth and reef type on Red Snapper densities and size distributions. We used hook-and-line, fish trap, and scuba visual surveys to estimate Red Snapper densities on four types of artificial reefs: army tanks, pyramids, small reefs (e.g., metal cages, pipelines), and large reefs (e.g., ships, dry docks, oil platforms). Small Red Snapper (<33 cm TL) were significantly more abundant at shallower depths (<35 m) and on small artificial reefs. Army tanks showed significantly more large fish (>33 cm TL) at shallow sites (<35 m) than at deeper sites (>35 m); in contrast, pyramids showed significantly more large fish at deeper sites. Ontogenetic habitat shift and fishing mortality were the most likely factors that would explain these Red Snapper distributions. Worldwide, artificial reefs are being used to enhance fishery resources and for habitat restoration. We document the importance of reef design and deployment location for Red Snapper, as well as the need for future artificial reef deployments to consider the relevant variables that affect the species that managers are attempting to enhance.

The use of artificial reefs to enhance catch rates has been used for more than 50 years (Dupont 2008). Artificial reefs are purposely submerged for a variety of functions including mitigation, prevention of trawling, enhancement of fish and invertebrate production, and to increase tourist opportunities (Baine 2001; Shipp and Bortone 2009). In the USA, the deployment of reefs has increased exponentially (McGurrin et al. 1989; Bohnsack et al. 1994). The increase is of particular importance in the northern Gulf of Mexico where many large species associated with reefs are exploited, e.g., Red Snapper *Lutjanus campechanus*, Gag *Mycteroperca microlepis*, and Gray Triggerfish *Balistes capriscus* (Lingo and Szedlmayer 2006; Dance et al. 2011). The natural bottom of the northern Gulf of Mexico consists primarily of sand or mud substrate, but artificial reefs have created an important component of structured habitat for reef fishes (Gallaway et al. 2009). The state of Alabama built its first reefs in 1953 by sinking multiple cars to provide habitat for reef-associated fish. In 1974, the artificial reef building program expanded by sinking several liberty ships followed by U.S. military army tanks in 1994. Since then, four artificial reef zones were established, and more than 15,000 reefs have been built (Minton and Heath 1998). Despite the prolific increase of artificial reefs, relatively little is known concerning the effects of reef type and placement location on common marine fish species.

The Red Snapper is one of the most important commercial and sport fish species in the northern Gulf of Mexico (SEDAR 2013). These snapper are geographically distributed throughout the Gulf of Mexico and along the Atlantic coast up to Cape Hatteras, North Carolina (Rivas 1966). Historically, Red Snapper fisheries originated off the Florida panhandle, shifting westward after the deployment of gas and oil platforms and permitted artificial reef zones in the 1950s (Camber 1955; Shipp and Bortone 2009). Fishery management models

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showed a substantial decline in Red Snapper in recent years, and this decline has been attributed to overfishing, shrimp trawl bycatch of juveniles, advancement in fishing technology, and change in habitats (SEDAR 2013). In the last decade, federal regulations have reduced total allowable catch, restricted bag and size limits, and shortened the recreational season in an effort to increase the Red Snapper stocks (Hood et al. 2007).

Red Snapper are the most abundant reef associated species in the northern Gulf of Mexico and use artificial reefs for shelter and prey resources (Ouzts and Szedlmayer 2003; Strelcheck et al. 2005; Lingo and Szedlmayer 2006; Piko and Szedlmayer 2007; Gallaway et al. 2009; Dance et al. 2011). Reef-specific factors, such as distance to nearest reefs, proximity to shore, reef design, and predation and competition pressures may affect fish response to artificial reefs, but their effects on Red Snapper are relatively unknown (Strelcheck et al. 2005; Lindberg et al. 2006; Piko and Szedlmayer 2007; Mudrak and Szedlmayer 2012).

Off the coast of Alabama, the continental shelf gradually increases in depth to approximately 50 m then drops sharply to over 1,000 m about 100 km from shore. Red Snapper may shift habitat as they age, older and larger fish being more abundant in deeper shelf areas, but such patterns are not well documented (Gallaway et al. 2009). Patterns of apparent habitat shift may also result from increased fishing pressure closer to shore due to increased time and cost of reaching distant locations (Gordon 1993). For example, many smaller boats are restricted by sea conditions and likely stay closer to shore (Kanamoto 1996; Womack 2003).

Reef complexity and reef size can also affect fish demographics (Bohnsack et al. 1994; Gratwicke and Speight 2005; Lindberg et al. 2006; Lingo and Szedlmayer 2006). For example, Lindberg et al. (2006) found larger reefs had greater densities and abundance of Gag than smaller reefs due to greater available shelter, but these greater densities resulted in slower growth. In addition, reef type has been shown to significantly affect Red Snapper demographics in the northern Gulf of Mexico, but these studies were limited in that they only examined small reefs <4.1 m^3 (Strelcheck et al. 2005; Lingo and Szedlmayer 2006; Dance et al. 2011). Studies have documented large numbers of Red Snapper on oil and gas platforms (Stanley and Wilson 1991), but again there is a lack of comparisons of Red Snapper distributions among large and small artificial reefs. Considering the size range of reef types (small metal cages to oil–gas platforms), it might be expected that reef type would affect resident Red Snapper.

Both the State of Alabama and private individuals continue to build artificial reefs of all shapes and sizes in Alabama's 3,100 km^2 permit areas (MRD 2006), but we suspect that not all artificial reef types have the same effect on Red Snapper. Thus, the objective of our study was to determine the effects of reef type, size and location (distance from shore and depth) on Red Snapper densities and size distributions. Information gained from our study may then be applied to future reef building projects to develop optimum designs for increased efficiency and maximum benefit for Red Snapper stocks.

METHODS

Study site.—The study area was located 20–50 km south of Mobile Bay, Alabama. Artificial reef types surveyed included pyramids, tanks, concrete rubble, metal reefs, barges, oil–gas platforms, ships, metal cages, and a few other miscellaneous structures. All reefs were located in three depth zones: inshore (18–26 m), midshore (26–34 m), and offshore (34–41 m). We defined public reefs as reefs with published locations made available to fishers by the Alabama Department of Conservation and Natural Resources (mostly pyramids, tanks, liberty ships, and oil–gas platforms). All of the approximately 500 published reefs were deployed at depths <41 m and were separated into three categories: tanks (volume = 51 m^3), pyramids (4 m^3), and big reefs (25–3,800 m^3; e.g., ships, oil platforms). Unpublished reefs were defined as reefs with locations usually known only to the builder. We located 241 unpublished reefs off the coast of Alabama with 19 side-scan sonar (Edgetech 4125-Dual 400/900hz) tows covering an area of 136 km^2. Unpublished reefs consisted of a wide variety of structures (volume = 4–15 m^3) from concrete culverts to metal cages. Between December 2011 and October 2013, Red Snapper were collected using stratified random sampling among three depth zones and four artificial reef types (pyramids, tanks, big reefs and small unpublished reefs) for a total of 48 public and unpublished artificial reefs from the known pool of approximately 750 reefs off the coast of Alabama (Figure 1). Sampling was limited to calm sea conditions (wave height, <1 m).

Fish surveys.—Red Snapper were collected following methods of Syc and Szedlmayer (2012). We used standardized hook-and-line fishing (30 min with 2 fishers) and fish trap (4 replicate 15-min trap sets) for each reef. Fishing gear included double 7/0 J hooks, 27-kg test monofilament line, 45-kg test monofilament leader, and whole Gulf Menhaden *Brevoortia patronus* as bait. Hook-and-line fishing gear was selected to target fish above the minimum retention size limit (33 cm total length). After completion of hook-and-line sampling, additional fish were collected with a fish trap (1.2 × 1.5 × 0.6 m; Collins 1990), again using Gulf Menhaden as bait. After collections of Red Snapper reached about 50 individuals per reef, additional fish caught were counted and released. All Red Snapper retained were immediately packed on ice and returned to the laboratory for size measurements. Between 1000 and 1600 hours, divers (scuba) used stationary point-count methods to estimate remaining Red Snapper after hook-and-line and trap methods. Divers also estimated Red Snapper total length to the nearest 2.5 cm. In addition, divers recorded the material and size of each artificial reef. At four sites, diver visual surveys were completed 8–19 d after Red Snapper collections due to shark sightings during collections. A YSI 6920

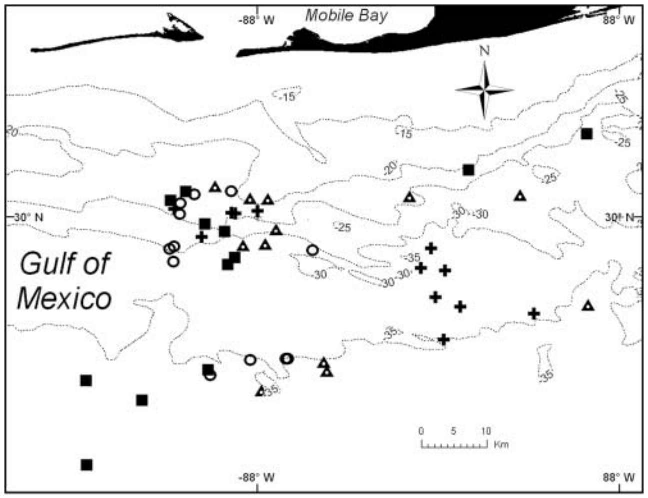

FIGURE 1. Artificial reefs in the northern Gulf of Mexico sampled for Red Snapper, including big reefs (squares), pyramids (pyramids), tanks (crosses), and small unpublished reefs (circles). Dotted lines indicate depth contours at 5-m intervals.

environmental meter was used to measure temperature (°C), salinity (‰), and dissolved oxygen (mg/L) at depth near each reef site. Proximity of the study site to other reefs was measured using ARCGIS by measuring the distance to the nearest published or unpublished reef.

Variation in visibility can affect Red Snapper diver visual counts; therefore, we standardized counts by the maximum distance that divers could see a Secchi disk (m) as the radius of our stationary point-count circle (up to 20 m radius). Fish densities (counts/100 m^2 of reef area) were estimated from area surveyed, diver counts, hook-and-line catch, and trap catch for each reef. Red Snapper were also separated into two size groups based on minimum retention size-limit of 33 cm TL for commercial fisheries, so prerecruits were <33 cm and postrecruits were >33 cm. Six sites with visibility <3 m were removed from the analysis. Total abundances (in contrast to densities) were only estimated on reefs <9 m^3 due to the inability of divers to completely survey larger reefs.

Data analyses.—The effects of depth, time of year (month, year), reef type, and reef proximity (meters to nearest reef) on fish size (TL), catch per unit effort (hook-and-line CPUE = number caught/30 min), and densities were examined with a generalized linear model (GLM). Reef type was considered a fixed factor while depth, time of year, and reef proximity were considered covariates. Each covariate was tested for two-way interactions, and stepwise simplification was performed using the corrected Akaike information criterion (AIC_c; Burnham and Anderson 2002). If a significant interaction with reef type was detected, we used regression analysis to show specific relations for each reef type.

We compared fish trap CPUE (number caught/15 min set) with a generalized linear mixed model (GLMM), and reef site (with the four replicate trap sets) was treated as a random factor in order to eliminate spatial pseudo-replication (Littell et al. 2002; Zar 2010). Chi-square analyses were used to compare size frequency distribution among depth zones and reef types. Differences were considered significant at $\alpha = 0.05$.

For CPUE and visual surveys, we used a negative binomial error distribution because our data consisted of individuals and were overdispersed (Zeileis et al. 2008). A normal error distribution was used for average total length of fish caught with hook and line and fish traps. Assumptions of the best-fit model were verified by visual examination of probability plots of residuals and QQ plots (defined as a plot of the percentiles of a standard chosen distribution versus the corresponding percentiles of the observed data; Zar 2010).

RESULTS

We collected 1,800 Red Snapper, 1,307 by fish trap, and 493 by hook and line. Due to the live release of some trap-collected fish, 1,434 fish were measured in the laboratory. Mean ± SE Red Snapper hook-and-line CPUE was 11 ± 1, and size was 598 ± 16 mm TL. Mean ± SE fish trap CPUE was 6 ± 1 and size was 359 ± 17 mm TL. Fish trap catch varied widely ranging from 0 to 68 Red Snapper/15 min trap set. Hook-and-line caught significantly larger Red Snapper (ANOVA: $F_{1,\ 73} = 106.08$, $P < 0.001$) and showed greater (but not significant) CPUE (GLM: df = 1, $\chi^2 = 2.92$, $P = 0.08$) than did the fish trap.

Hook and Line

There was a significant interaction (GLM) between depth and reef type for both hook-and-line CPUE (df = 3, $\chi^2 = 11.43$, $P = 0.01$) and Red Snapper size ($F_{3,\ 32} = 3.69$, $P = 0.02$). Hook-and-line CPUE was significantly lower (df = 3, $\chi^2 = 14.32$, $P = 0.003$) on pyramids (mean = 4.0 [SE = 1.55]) compared with small unpublished reefs (11.17 [2.54]), big reefs (11.75 [1.63]), and tanks (14.17 [1.55]). Within reef types, CPUE from pyramids increased significantly as depth increased ($R^2 = 0.43$, $P = 0.02$; Figure 2b). In contrast, CPUE from tanks significantly decreased as depth increased ($R^2 = 0.63$, $P = 0.003$; Figure 2a). Although fewer Red Snapper were caught on pyramids, they were significantly larger (mean = 689 mm TL, [SE = 57]) than all other reef types ($F_{3,\ 32} = 4.85$, $P < 0.007$; tanks: 585 mm TL [16]), small unpublished reefs (557 mm TL, [37]), big reefs (595 mm TL, [21]). For the larger reef types (big reefs and tanks), mean size of Red Snapper from hook-and-line catch significantly increased as depth increased for both tanks ($R^2 = 0.56$, $P = 0.008$) and big reefs ($R^2 = 0.55$, $P = 0.006$; Figure 3).

Fish Trap

Fish trap CPUE was significantly greater (3 times) on the small unpublished reefs compared with big reefs and pyramids

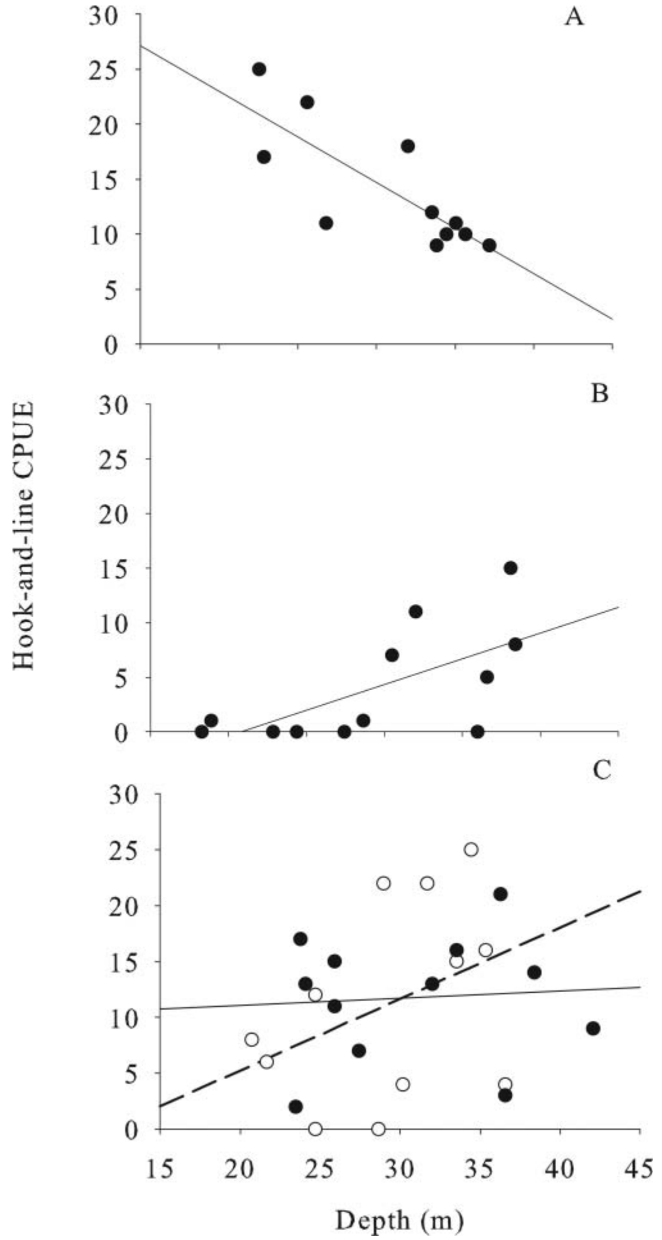

FIGURE 2. Comparison of mean Red Snapper hook-and-line CPUE (catch/30 min) by depth among northern Gulf of Mexico reefs types. The line is a linear regression slope that was significant for **(A)** tanks ($R^2 = 0.63$, $P = 0.003$) and **(B)** pyramids ($R^2 = 0.43$, $P = 0.02$) but was not significant for **(C)** big reefs (closed circles, solid line; $R^2 = 0.0052$, $P = 0.82$) or small unpublished reefs (open circles, dashed line; $R^2 = 0.15$, $P = 0.21$).

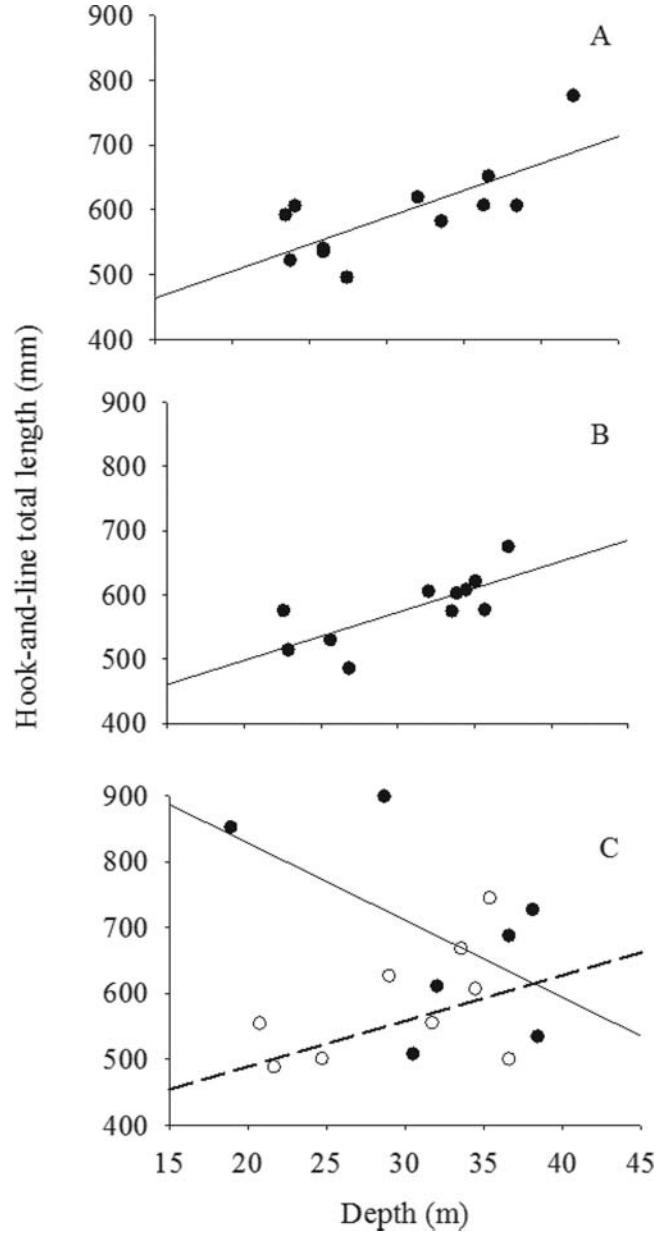

FIGURE 3. Comparison of mean total lengths of Red Snapper (hook-and-line catch/30 min) by depth among northern Gulf of Mexico reefs types. Line is a linear regression slope that was significant for **(A)** big reefs ($R^2 = 0.55$, $P = 0.006$) and **(B)** tanks ($R^2 = 0.56$, $P = 0.008$) but was not significant for **(C)** pyramids (closed circles, solid line; $R^2 = 0.28$, $P = 0.21$) or small unpublished reefs (open circles, dashed line; $R^2 = 0.11$, $P = 0.34$).

(GLMM: df = 3, $\chi^2 = 3.02$, $P = 0.03$, Figure 4). A significant interaction between depth and reef type was detected for fish trap CPUE (df = 3, $\chi^2 = 3.04$, $P = 0.03$), CPUE decreasing significantly on tanks with depth ($F_{1,\ 34} = 4.96$, $P = 0.032$). Fish size showed a marginally significant interaction effect between depth and reef type for fish trap collections (GLM: $F_{1,26} = 2.75$, $P = 0.063$). Red snapper mean size significantly decreased as depth increased in trap collections on tanks ($R^2 = 0.54$, $P = 0.02$; Figure 5a), but mean size significantly increased as depth increased on pyramids ($R^2 = 0.66$, $P = 0.03$; Figure 5b).

Fish Density

Red Snapper were present on all diver visual surveys. Divers counted 3,624 Red Snapper from 42 surveys. The most

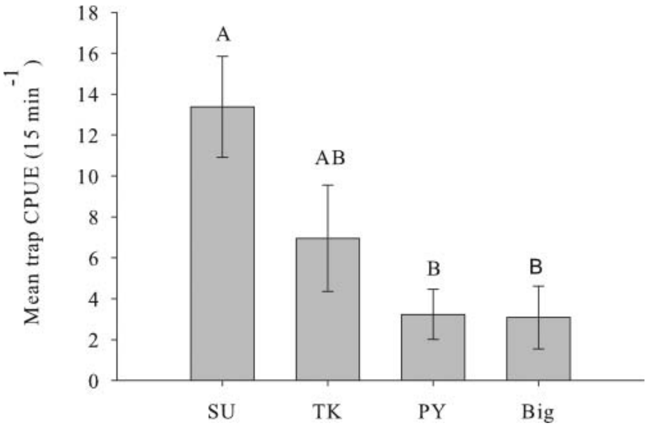

FIGURE 4. Comparison of mean (±SE) fish trap CPUE by artificial reef types: tanks (TK), pyramids (PY), small unpublished reefs (SU), and big reefs (Big). Significant differences among different reefs types are indicated by different letters.

abundant size-classes of Red Snapper were 0–25, 26–35, and 36–45 cm TL, each contributing 18–28% of the total. The 46–55-cm size-class comprised 16% of the total, and the fewest Red Snapper (≤10%) were found in the largest (56–65 and >65 cm) size-classes. When diver counts were added to captured fish, mean total density (Red Snapper/100 m^2) was 117.2 (SE, 32.4).

Total densities of Red Snapper did not show a significant relation with either reef type (GLM: df = 3, χ^2 = 1.95, P = 0.58), or depth (df = 1, χ^2 = 1.35, P = 0.25). When densities of Red Snapper were divided into size categories, prerecruit densities showed no significant type or depth-type interaction effects, but a significant depth effect was detected (GLM: df = 1, χ^2 = 4.67, P = 0.03). Prerecruit densities significantly decreased as depth increased (R^2 = 0.20, P = 0.003).

Red Snapper postrecruit densities showed a significant interaction between depth and reef type (GLM: df = 3, χ^2 = 12.49, P = 0.006). Within reef types, Red Snapper postrecruit densities significantly increased on pyramids as depth increased (R^2 = 0.44, P = 0.04; Figure 6b), significantly decreased on tanks as depth increased (R^2 = 0.88, P < 0.001; Figure 6a), and showed a marginally significant decrease on big reefs with depth (R^2 = 0.32, P = 0.06; Figure 6c). Although highest densities of postrecruits were found on tanks, we did not find significant differences among reef types (Table 1).

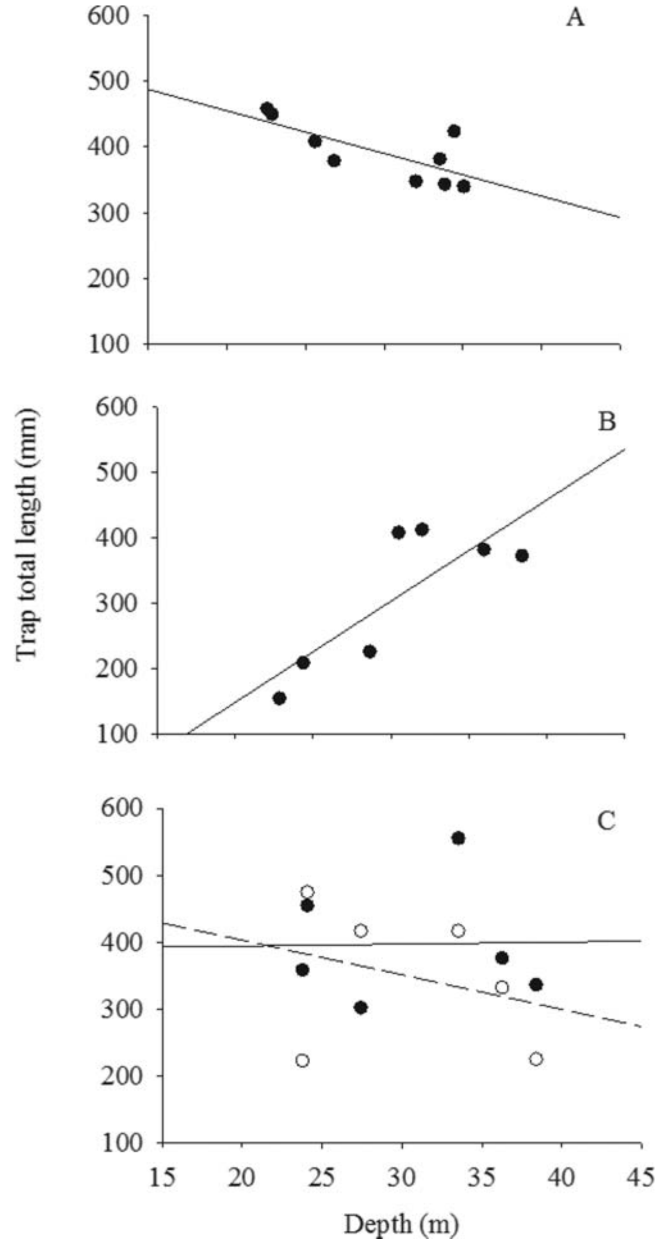

FIGURE 5. Comparison of mean total lengths (mm) of Red Snapper caught with traps (15 min/set) versus depth (m) by reefs types. The lines are a linear regression slopes that were significant for **(A)** tanks (R^2 = 0.54, P = 0.02) and **(B)** pyramids (R^2 = 0.66, P = 0.03), but was not significant for **(C)** big reefs (closed circles, solid line; R^2 < 0.001, P = 0.97) or small unpublished reefs (open circles, dashed line; R^2 = 0.19, P = 0.15).

Diver Estimates of Size Frequency

Diver estimates showed that fish size significantly increased as depth increased (df = 10, χ^2 = 43.95, P < 0.001; Figure 7). Size distributions were not significantly different between inshore (depth, 18–26 m) and midshore (26–34 m) reefs (df = 5, χ^2 = 6.73, P = 0.24); however, both were significantly different from offshore reefs (inshore–offshore: df = 5, χ^2 = 35.72, P < 0.001; midshore–offshore: df = 5, χ^2 = 30.87, P < 0.001). Inshore reefs and mid-shore reefs were dominated (31–33%) by Red Snapper in the <25-cm size-class, while offshore reefs only showed 4% of this smaller size-class. The opposite trend was shown for the largest size-category of Red Snapper (>65 cm), inshore reefs showing 4%, midshore 8%, and offshore 16% of these larger fish.

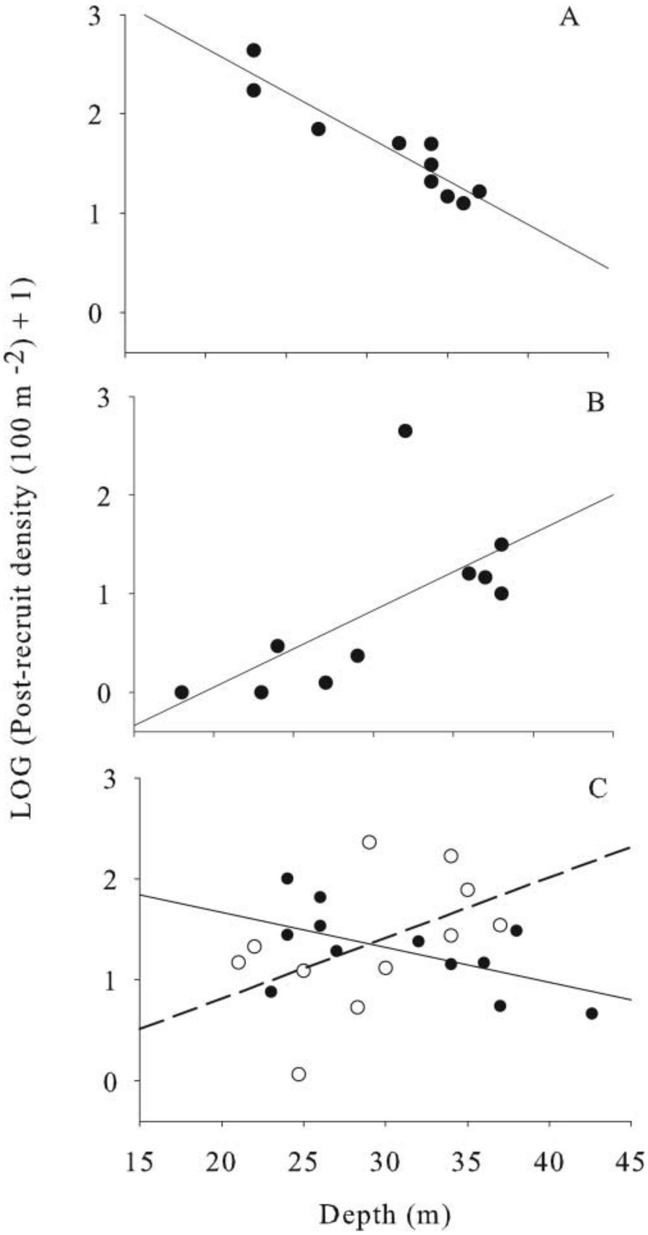

FIGURE 6. Comparison of density ($\log_{10} x + 1$) of postrecruit Red Snapper, where density is the number/100 m^2 of reef area, versus reef depth and type. The lines are a linear regression slopes that were significant for **(A)** tanks ($R^2 = 0.87$, $P < 0.001$) and **(B)** pyramids ($R^2 = 0.44$, $P = 0.04$), but was not significant for **(C)** big reefs (closed circles, solid line; $R^2 = 0.32$, $P = 0.06$) or small unpublished reefs (open circles, dashed line; $R^2 = 0.25$, $P = 0.12$).

There was a significant difference in Red Snapper size distributions among reefs types (df = 15, $\chi^2 = 101.97$, $P < 0.001$; Figure 8). Red Snapper size distributions were significantly skewed towards larger fish (>45 cm TL) on big reefs compared with pyramids (df = 5, $\chi^2 = 31.80$, $P < 0.001$) and small unpublished reefs (df = 5, $\chi^2 = 52.70$, $P < 0.001$). Significantly larger size distributions of Red Snapper were also shown on tanks than pyramids (df = 5, $\chi^2 = 27.77$, $P < 0.001$) and small unpublished reefs (df = 5, $\chi^2 = 66.78$, $P < 0.001$). Both big reefs and tanks showed normal size distributions, the highest percentage of fish being in the medium size-class (25–55 cm). In contrast, both small unpublished reefs and pyramids were dominated by small fish <25 cm. However, small unpublished reefs had less (4.5 times) Red Snapper in the 46–55 cm and higher percentage (16 times) in the >65-cm size-class compared with pyramids (df = 5, $\chi^2 = 20.01$, $P < 0.001$).

Environmental Effects

Dissolved oxygen and salinity showed little variation across sample periods or reef sites (Table 2). In addition, time of year (which is correlated with temperature) did not significantly affect Red Snapper CPUE (GLM for hook-and-line CPUE: df = 1, $\chi^2 = 0.54$, $P = 0.46$; GLMM for trap CPUE, df = 1, $\chi^2 = 0.20$, $P = 0.66$) or densities (GLM: df = 1, $\chi^2 = 1.88$, $P = 0.16$). Proximity to other reefs (known public reefs and unpublished reefs located by side-scan surveys) also did not significantly affect CPUE (GLM for hook-and-line CPUE: df = 1, $\chi^2 = 0.95$, $P = 0.33$; GLMM for trap CPUE: df = 1, $\chi^2 = 1.23$, $P = 0.27$) or densities (GLM: df = 1, $\chi^2 = 2.14$, $P = 0.14$). Both pyramids and big reefs have different known ages since deployment, but we failed to detect significant reef age effects on Red Snapper densities (GLM for pyramid: df = 1, $\chi^2 = 0.08$ $P = 0.78$; for big reef: df = 1, $\chi^2 = 3.21$, $P = 0.07$), fish trap CPUE (GLMM for pyramid: df = 1, $\chi^2 = 0.80$, $P = 0.38$; for big reef df = 1, $\chi^2 = 0.98$, $P = 0.33$), or hook-and-line CPUE (GLM for pyramid: df = 1, $\chi^2 = 2.19$ $P = 0.14$; for big reef: df = 1, $\chi^2 = 0.00$, $P = 0.96$).

DISCUSSION

We showed depth and reef type affected Red Snapper CPUE, densities of prerecruits and postrecruits, and mean size. Similar to previous studies, our visual surveys showed Red Snapper on all reef sites despite reef type differences (Dance et al. 2011; Syc and Szedlmayer 2012). Importantly, the effects of changing depths differed among reef types. Other factors are also known to affect Red Snapper densities and size, e.g., reef age (Syc and Szedlmayer 2012) and abiotic factors (Gallaway et al. 1999; Stanley and Wilson 2004), but our study failed to detect these factors as significant. For example, temperature varied seasonally, but we did not observe seasonal patterns of Red Snapper densities, similar to previous studies (Redman and Szedlmayer 2009; Dance et al. 2011). No significant salinity and dissolved oxygen effects were detected as these measures showed little variation during this study.

Red Snapper densities observed in our study (117 fish/100 m^2) appeared to be substantially greater than estimates of 5–75 fish/100 m^2 on pyramid and dome-shaped reefs off

TABLE 1. Densities of Red Snapper by fishery management sizes (i.e., prerecruit [<33 cm TL] and postrecruit [>33 cm TL]), as well as visual densities, which were of all sizes observed on artificial reefs located in the northern Gulf of Mexico. Significant differences (ANOVA; α = 0.05) were not detected.

	Mean ± SE density (number/100 m^2 of area)		
Reef type	Prerecruit	Postrecruit	Visual densities
Tank	13.0 ± 4.6	86.4 ± 41.6	99.4 ± 44.3
Pyramid	35.7 ± 20.5	51.9 ± 44.0	87.6 ± 48.3
Small unpublished	89.7 ± 43.0	54.5 ± 22.9	144.2 ± 44.0
Big	10.2 ± 2.9	28.2 ± 8.15	38.4 ± 10.2

northwest Florida (Dance et al. 2011). When estimated on a per reef volume, Red Snapper densities on small reefs (pyramids and metal cages mean: 13.6 fish/m^3) were similar to past estimates on metal cages (15.7/m^3; Syc and Szedlmayer 2012).

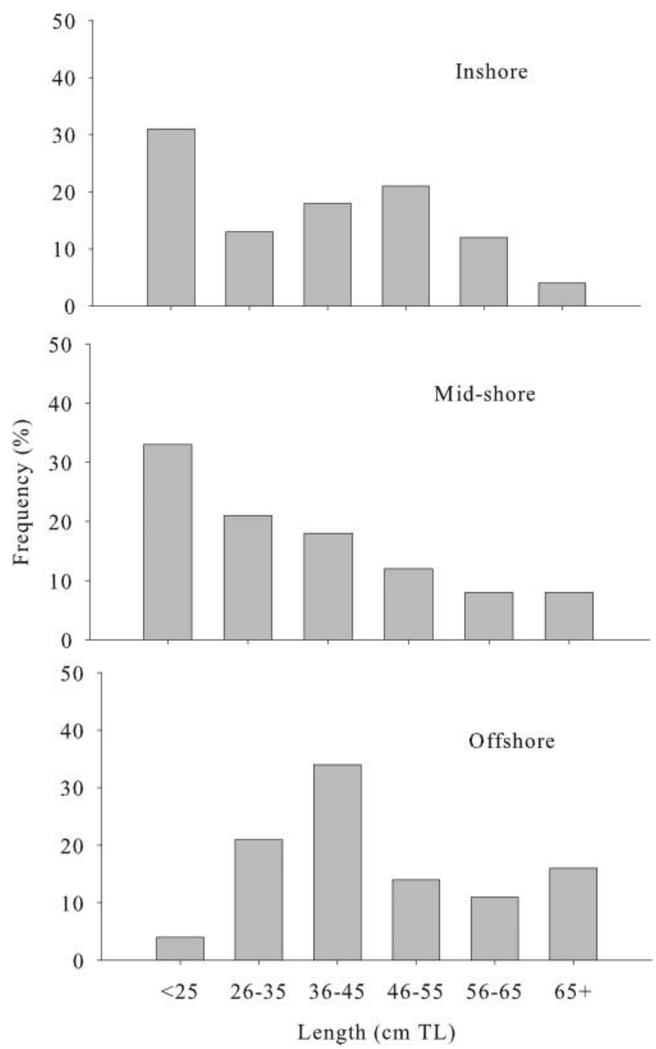

FIGURE 7. Length frequency of Red Snapper (10-cm increments) on artificial reefs in the northern Gulf of Mexico by depth zones: Inshore = 18–26 m, Midshore = 26–34 m, and Offshore = 34–41 m.

Catch methods were designed to catch a wide range of Red Snapper sizes and provide an unbiased mean size estimate of resident Red Snapper at each reef site. As expected, hook and line took larger Red Snapper, while the fish trap captured smaller ones. Size differences of catch were attributed to gape-size-limitations for hook and line and opening size of the fish trap (Syc and Szedlmayer 2012). However, catch rates may also be affected by environmental conditions, such as sea state (wind speed and wave height) and fish behavior. Fish behavior may include such factors as desire to feed, location of bait, and successful attack of bait (Stoner 2004). In this study, we attempted to reduce some of these factors by limiting sampling days to calm sea conditions (wave height less than 1 m). In addition, catch rates were not significantly affected by water temperature (affects metabolism and possible feeding rates) or visibility (may affect bait locating ability).

Reef Types

Ecological processes that affect Red Snapper distributions are most likely affected by reef type components (e.g., age, construction material, complexity, size). For example, if Red Snapper show high site fidelity to particular reef sites, then fish age may be correlated to reef age (Topping and Szedlmayer 2011; Syc and Szedlmayer 2012). Key aspects of this reef age–fish age correlation were the absence of fishing mortality and reef age <5 years (Syc and Szedlmayer 2012). In contrast, public reefs sampled in our study were at least 6 years old with the majority over 10 years, which would explain larger fish sizes on present reefs compared with the previous study (Syc and Szedlmayer 2012). However, reef age did not affect Red Snapper densities or size on tanks, pyramids, and big reefs. Likewise, reef age did not affect fish abundance on oil platforms (Stanley and Wilson 1991). Previous studies in the Gulf of Mexico and the Atlantic have shown that artificial reefs typically need around 5 years to reach mature community equilibrium; thus, reefs we sampled were most likely fully colonized before sampling because they were 7–39 years old (Lukens 1981; Bohnsack and Sutherland 1985). Probably more important, these reefs are heavily exploited, and fishing mortality of large Red Snapper may reduce any

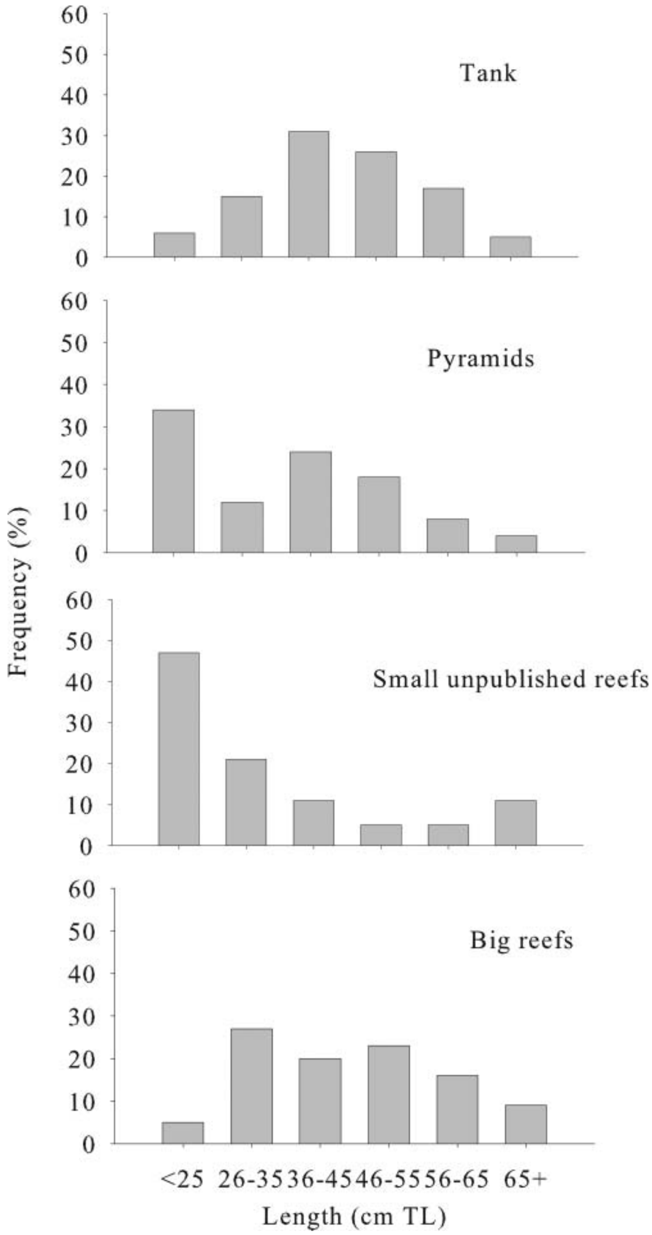

FIGURE 8. Length frequency of Red Snapper (10-cm increments) on artificial reefs in the northern Gulf of Mexico by reef type.

TABLE 2. Environmental measures for artificial reefs in the northern Gulf of Mexico

Variable	Range	Mean	SE
Temperature (°C)	16.7–28.5	23.3	0.6
Dissolved oxygen (mg/L) oxygen	4.3–7.8	6.2	0.2
Salinity (‰)	32.4–36.5	34.9	0.2

reef age–fish age correlation due to fishers targeting older, larger fish.

The relation between reef type and Red Snapper size is consistent with their life history patterns (Gallaway et al. 2009). The youngest Red Snapper first settle onto low-relief habitat, such as relic shell, and then move to larger structured habitat as they grow (Szedlmayer and Conti 1999; Szedlmayer and Lee 2004; Gallaway et al. 2009). We observed few Red Snapper <10 cm TL on the reefs surveyed, and higher numbers of small (10–25 cm) Red Snapper on pyramids and small unpublished reefs in comparison to tanks and big reefs. These patterns suggest that the smaller-sized reefs in the our study provided an intermediate habitat step between low relief shell rubble habitat used by new recruits (Szedlmayer 2011) and larger reefs dominated by larger adults (Gallaway et al. 2009). Previous studies in the laboratory and field both support a Red Snapper size distribution pattern driven by adult conspecifics (Bailey et al. 2001; Mudrak and Szedlmayer 2012).

Mudrak and Szedlmayer (2012) found higher numbers of age 0 Red Snapper when small "recruitment" reefs were placed away from larger reefs occupied by resident adult Red Snapper, indicating a negative effect of adult conspecifics on newly settled Red Snapper. Similarly, juvenile Red Snapper abundance was negatively correlated with the abundance of nearby artificial reefs (Strelcheck et al. 2005). These previous studies indicate that before deploying new artificial reefs, the presence of nearby reefs needs careful evaluation because their presence may actually inhibit Red Snapper recruitment. In contrast, we failed to detect a significant reef-proximity effect on Red Snapper CPUE. One difference in our study compared with other studies was the distances among reefs. In Mudrak and Szedlmayer (2012) reef proximity effects were detected at 15-m distances. Likewise, in Strelcheck et al. (2005), most (80%) distances among reefs were <30 m, while in our study the mean linear distance to the closest reef was >1 km; only 5 of 48 sites within 30 m.

Another ecological interaction that most likely affected Red Snapper demographics is predation. Previous research has shown that predator numbers and predation success were influenced by size and complexity of artificial reefs (Stanley and Wilson 1991; Piko and Szedlmayer 2007). Lingo and Szedlmayer (2006) showed that juvenile Red Snapper were at higher abundances on more complex recruitment reefs (1 m^3). These differences were attributed to reduced predation success on more complex reefs, as well as Red Snapper preference for complex reefs in the presence of predators (Piko and Szedlmayer 2007). Similarly, availability of prey refuges (holes similar to the body size of prey species) indirectly affected density dependence through predation (Hixon and Beets 1993). In our study, pyramids had high vertical relief, but this reef type typically had one large opening (about 30 cm) on each of its three sides. In contrast, small reefs were primarily constructed with either metal cages or concrete rubble and both provided many small refuges and increased complexity. In addition, sizes of Red Snapper on tanks and big reefs were skewed towards larger fish (>25 cm). These fish are less vulnerable to predation (interspecific and intraspecific) and can inhabit larger reefs, where densities of larger potential predators (e.g., sharks) are probably greater (Gallaway et al. 2009).

In addition to predation, food availability may also affect fish distribution. Food availability in turn is probably related to reef type through differences in reef construction material. Initial biofouling rate is greater on rough surfaces, and epibenthic assemblages are significantly different on concrete versus steel (Anderson and Underwood 1994; Andersson et al. 2009). Pyramids were composed primarily of concrete, while tanks were composed of steel. In contrast, materials were mixed for both small reefs (2 concrete reefs and 10 steel reefs) and large reefs (3 concrete reefs, 6 steel reefs, and 3 concrete and steel reefs). Red Snapper diets consist of epibenthos and prey that feed on epibenthos (Ouzts and Szedlmayer 2003; Szedlmayer and Lee 2004). Redman and Szedlmayer (2009) used antifouling paint to show that Red Snapper abundance and fish species diversity were lower on artificial reefs with low epibenthos percent cover. However, Andersson et al. (2009) found that percent cover of metal and concrete pipes were 100% after 1 year, if left untreated, and did not find a difference in fish abundance between the two materials. Published reefs sampled in our study were on average 18 years old and most likely fully covered by the epibenthic community. If concrete reefs supported more epibenthos and Red Snapper densities were driven by bottom–up processes, then pyramids would have had the highest densities of Red Snapper instead of one of the lowest. Thus, it is unlikely that differences in food availability caused differences in Red Snapper abundance among reef types.

Fishing mortality is another important factor that may account for part of the difference between fish density on published versus small unpublished reefs. Fish trap CPUE was highest on the small unpublished reefs compared with all other reef types. The locations of pyramids, tanks, and big reefs are published, and fishing pressure is most likely higher on these published sites than small unpublished reefs. The Red Snapper sport fishery season in the northern Gulf of Mexico has decreased steadily from 194 d in 2007 to just 9 d in 2014 (Gulf of Mexico Fishery Management Council 2014). Those who do fish year-round may target species such as Vermilion Snapper *Rhomboplites aurorubens* or Gray Triggerfish, which require that anglers use smaller hook size and bait; however, smaller Red Snapper are a bycatch of that effort. If small Red Snapper are caught and released repetitively from public sites, then they may become wary of hook-and-line fishing and fish traps (Gilbert et al. 2001; Askey et al. 2006). More importantly, Red Snapper greater than 65 cm were 16 times more abundant on small unpublished reefs than on similarly sized published pyramids. Thus, it appears that unpublished reefs may be providing a refuge from fishing mortality.

Depth Effects

The significant differences observed in Red Snapper densities and mean size with depth is similar to previous findings of increasing fish sizes with increasing depths. Predation may be an important factor in the separation of small and large Red Snapper as related to depth. MacPherson and Duarte (1991) found that 63% of fish species ($N = 44$) in the Atlantic migrated to deeper waters as they got older, showing depth-related diet shifts and lowered metabolism in the colder deeper waters. Along with Red Snapper, other large piscivores (e.g., Greater Amberjack *Seriola dumerili*) are more common in deeper Gulf waters (Stanley and Wilson 1991). The lack of Red Snapper <20 cm TL at our deeper reef sites (34–41 m) supports the contention that smaller fish remain shallow to avoid competition, larger fish and the threat of predation (Linehan et al. 2001). Shallow waters provide a spatial separation from large piscivores, and also opportunities to inhabit alternative nursery habitat (Rozas and Minello 1998; Gallaway et al. 2009; Jaxion-Harm et al. 2012). Most nursery habitats are shallower than adult habitats. Mudrak and Szedlmayer (2012) deployed small (1.5 m^3), low-relief reefs (19–22 m) specifically for recruitment. These reefs were successful in recruiting age-0 and age-1 Red Snapper, but for the most part were at depths shallower than reefs we surveyed. In the present study, both reef size and depth probably explain the absence of newly settled Red Snapper.

Fishing mortality may also be affected by depth. Due to the gradual sloping shelf of the northern Gulf of Mexico, fishers must travel long distances for a 10-m difference in depth. Previous studies have shown that charter boats take distance to shore into consideration when choosing fishing sites and may select closer fishing sites to save on fuel costs (Gordon 1993). In addition, smaller boats, which are more affected by sea conditions, may choose to stay closer inshore (Kanamoto 1996; Womack 2003). Considering these factors, fishing mortality is most likely higher at shallower, inshore reefs than at deeper, offshore sites.

Previous studies have shown that reefs that supported lower densities of fish, supported greater biomass due to larger fish (Bohnsack et al. 1994; Lindberg et al. 2006). This pattern has been attributed to a negative correlation between fish growth and density (Campbell et al. 2011; Lindberg et al. 2006). Tanks and big reefs in deep water (34–41 m) followed this pattern, larger fish (55–80 cm) being found at lower densities. Although prerecruit Red Snapper stay primarily at shallower depths, larger postrecruit fish were found at all depths. However, most Red Snapper on tanks and big reefs located at shallower depths were an intermediate size (40–60 cm) fish. We suggest that greater fishing mortality inshore reduces the number of fish >65 cm on tanks and big reefs. This removal of the largest fish may then cause increased intermediate-sized fish through reduced competitive exclusion, which then show greater hook-and-line CPUE on tanks and greater postrecruit densities on both tanks and big reefs at inshore sites. Pyramid reefs followed the expected ontogenetic migration pattern, postrecruit Red Snapper being more abundant on deeper reefs.

Management Implications

The function of artificial reefs has been described as either simple attractors that make fish easier to catch and drive stocks toward faster depletion (Matthews 1985; Bohnsack et al. 1994) or as enhancers that increase production of resident fish stocks and, thus, benefit populations (Dupont 2008; Gallaway et al. 2009). Matthews (1985) showed that artificial reefs attracted larger, commercially important fishes away from natural reefs, and the artificial concentration of fishes at these sites increased vulnerability to fishing. However, natural reefs are limited (<4% of total benthic habitat; Parker et al. 1983; Dufrene 2005) in the northeast Gulf of Mexico, and historical evidence suggests Red Snapper were not abundant in these water until after artificial reefs were built (Shipp and Bortone 2009). The lack of natural high relief habitat along with lower historical catches suggested that natural reefs in the study area have little effect on Red Snapper densities. While intense fishing on published artificial reefs may deplete postrecruit sized Red Snapper, we suggest that reefs placed further offshore, along with unpublished reefs may enhance densities of the largest (>65 cm) Red Snapper. We also suggest that differences in size distribution of Red Snapper with depth and reef type indicate the necessity of deploying diverse reef types at multiple depths in order to provide habitat for all life stages of Red Snapper. For example, although hook-and-line catch was extremely low on pyramids, these smaller-sized artificial reefs appear to provide valuable habitat for prerecruit Red Snapper. Comparing tanks with big reefs, densities of postrecruit Red Snapper were similar. However, tanks have a smaller volume than big reefs and, thus, hold higher densities per volume of Red Snapper. More importantly, we found that although reef types differ, all artificial reefs examined supported varying levels of Red Snapper, and they all provided valuable structured habitat that for the most part is uncommon on the northern continental shelf of the Gulf of Mexico.

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REFERENCES

Anderson, M. J., and A. J. Underwood. 1994. Effects of substratum on the recruitment and development of an intertidal estuarine fouling assemblage. Journal of Experimental Marine Biology and Ecology 184: 217–236.

Andersson, M. H., M. Berggren, D. Wilhelmsson, and M. C. Ohman. 2009. Epibenthic colonization of concrete and steel pilings in a cold-temperate embayment: a field experiment. Helgoland Marine Research 63:249–260.

Askey, P. J., S. A. Richards, J. R. Post, and E. A. Parkinson. 2006. Linking angling catch rates and fish learning under catch-and-release regulations. North American Journal of Fisheries Management 26:1020–1029.

Bailey, H. K., J. H. Cowan, and R. L. Shipp. 2001. Experimental evaluation of potential effects of habitat size and presence of conspecifics on habitat association by young-of-the-year Red Snapper. Gulf of Mexico Science 19:119–131.

Baine, M. 2001. Artificial reefs: a review of their design, application, management and performance. Ocean and Coastal Management 44:241–259.

Bohnsack, J. A., D. E. Harper, B. David, and M. Hulsbeck. 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs of southeastern Florida, USA. Bulletin of Marine Science 55:796–823.

Bohnsack, J. A., and D. L. Sutherland. 1985. Artificial reef research: a review with recommendations for future priorities. Bulletin of Marine Science 37:11–39.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference, 2nd edition. Springer-Verlag, New York.

Camber, C. I. 1955. A survey of the Red Snapper fishery of the Gulf of Mexico, with special reference to the Campeche banks. State of Florida Board of Conservation Marine Laboratory 12.

Campbell, M. D., K. Rose, K. Boswell, and J. Cowan. 2011. Individual-based modeling of an artificial reef fish community: effects of habitat quantity and degree of refuge. Ecological Modelling 222:3895–3909.

Collins, M. R. 1990. A comparison of 3 fish trap designs. Fisheries Research 9:325–332.

Dance, M. A., W. F. Patterson, and D. T. Addis. 2011. Fish community and trophic structure at artificial reef sites in the northeastern Gulf of Mexico. Bulletin of Marine Science 87:301–324.

Dufrene, T. A. 2005. Geological variability and Holocene sedimentary record on the northern Gulf of Mexico inner to mid-continental shelf. Master's thesis. Louisiana State University, Baton Rouge.

Dupont, J. M. 2008. Artificial reefs as restoration tools: a case study on the west Florida shelf. Coastal Management 36:495–507.

Gallaway, B. J., J. G. Cole, R. Meyer, and P. Roscigno. 1999. Delineation of essential habitat for juvenile Red Snapper in the northwestern Gulf of Mexico. Transactions of the American Fisheries Society 128:713–726.

Gallaway, B. J., S. T. Szedlmayer, and W. J. Gazey. 2009. A life history review for Red Snapper in the Gulf of Mexico with an evaluation of the importance of offshore petroleum platforms and other artificial reefs. Reviews in Fisheries Science 17:48–67.

Gilbert, D. J., J. R. McKenzie, and N. M. Davies. 2001. Evidence from tag recapture experiments that fish learn to avoid fishing gear. Journal of Agricultural Biological and Environmental Statistics 6:281–291.

Gordon, W. R. 1993. Travel characteristics of marine anglers using oil and gas platforms in the central Gulf of Mexico. Marine Fisheries Review 55:25–31.

Gratwicke, B., and M. R. Speight. 2005. Effects of habitat complexity on Caribbean marine fish assemblages. Marine Ecology Progress Series 292:301–310.

Gulf of Mexico Fishery Management Council. 2014. Recreational fishing regulations for Gulf of Mexico federal waters for species managed by the Gulf of Mexico Fishery Management Council. Gulf of Mexico Fishery Management Council, Tampa, Florida. Available: http://www.gulfcouncil.org/fishing_regulations/. (December 2014).

Hixon, M. A., and J. P. Beets. 1993. Predation, prey refuges, and the structure of coral reef fish assemblages. Ecological Monographs 63:77–101.

Hood, P. B., A. J. Strelcheck, and P. Steele. 2007. A history of Red Snapper management in the Gulf of Mexico. Pages 241–256 *in* W. F. Patterson III, J. H. Cowan, G. R. Fitzhugh, and D. L. Nieland, editors. Red Snapper ecology and fisheries in the U.S. Gulf of Mexico. American Fisheries Society, Symposium 60, Bethesda, Maryland.

Jaxion-Harm, J., J. Saunders, and M. R. Speight. 2012. Distribution of fish in seagrass, mangroves and coral reefs: life-stage dependent habitat use in Honduras. Revista De Biologia Tropical 60:683–698.

Kanamoto, Z. 1996. Daily changes in the catch of *Parapristipoma trilineatum* at the hook and line fishing ground of Toshima Fisherman's Union with the special reference to water conditions of Uwa Sea, Japan. Nippon Suisan Gakkaishi 62:565–571.

Lindberg, W. J., T. K. Frazer, K. M. Portier, F. Vose, J. Loftin, D. J. Murie, D. M. Mason, B. Nagy, and M. K. Hart. 2006. Density-dependent habitat selection and performance by a large mobile reef fish. Ecological Applications 16:731–746.

Linehan, J. E., R. S. Gregory, and D. C. Schneider. 2001. Predation risk of age-0 cod (*Gadus*) relative to depth and substrate in coastal waters. Journal of Experimental Marine Biology and Ecology 263:25–44.

Lingo, M. E., and S. T. Szedlmayer. 2006. The influence of habitat complexity on reef fish communities in the northeastern Gulf of Mexico. Environmental Biology of Fishes 76:71–80.

Littell, R. C., W. W. Stroup, and R. J. Freund. 2002. SAS for linear models, 4th edition. SAS Institute, Cary, North Carolina.

Lukens, R. R. 1981. Ichthyofauna colonization of a new artificial reef in the northern Gulf of Mexico. Gulf Research Reports 7:41–49.

Macpherson, E., and C. M. Duarte. 1991. Bathymetric trends in demersal fish size: is there a general relationship. Marine Ecology Progress Series 71:103–112.

Matthews, K. R., 1985. Species similarity and movement of fishes on natural and artificial reefs in Monterey Bay, California. Bulletin of Marine Science 37:252–270.

McGurrin, J. M., R. B. Stone, and R. J. Sousa. 1989. Profiling United States artificial reef development. Bulletin of Marine Science 44:1004–1013.

Minton, R., and S. Heath. 1998. Alabama's artificial reef program: building oases in the desert. Gulf of Mexico Science 16:105–105.

MRD (Marine Resources Division). 2006. Alabama's artificial reefs. Alabama Department of Conservation, MRD, Gulf Shores, Alabama.

Mudrak, P. A., and S. T. Szedlmayer. 2012. Proximity effects of larger resident fishes on recruitment of age-0 Red Snapper in the northern Gulf of Mexico. Transactions of the American Fisheries Society 141:487–494.

Ouzts, A. C., and S. T. Szedlmayer. 2003. Diel feeding patterns of Red Snapper on artificial reefs in the north-central Gulf of Mexico. Transactions of the American Fisheries Society 132:1186–1193.

Parker, R. O., D. R. Colby, and T. D. Willis. 1983. Estimated amount of reef habitat on a portion of the United States south-Atlantic and Gulf of Mexico continental shelf. Bulletin of Marine Science 33:935–940.

Piko, A. A., and S. T. Szedlmayer. 2007. Effects of habitat complexity and predator exclusion on the abundance of juvenile Red Snapper. Journal of Fish Biology 70:758–769.

Redman, R. A., and S. T. Szedlmayer. 2009. The effects of epibenthic communities on reef fishes in the northern Gulf of Mexico. Fisheries Management and Ecology 16:360–367.

Rivas, L. R. 1966. Review of the *Lutjanus campechanus* complex of Red Snappers. Quarterly Journal of the Florida Academy of Sciences 29: 117–136.

Rozas, L. P., and T. J. Minello. 1998. Nekton use of salt marsh, seagrass, and nonvegetated habitats in a south Texas (USA) estuary. Bulletin of Marine Science 63:481–501.

SEDAR (Southeast Data Assessment and Review). 2013. SEDAR 31–Gulf of Mexico Red Snapper stock assessment report. SEDAR, North Charleston, South Carolina.

Shipp, R. L., and S. A. Bortone. 2009. A Prospective of the importance of artificial habitat on the management of Red Snapper in the Gulf of Mexico. Reviews in Fisheries Science 17:41–47.

Stanley, D. R., and C. A. Wilson. 1991. Factors affecting the abundance of selected fishes near oil and gas platforms in the northern Gulf of Mexico. U.S. National Marine Fisheries Service Fishery Bulletin 89:149–159.

Stanley, D. R., and C. A. Wilson. 2004. Effect of hypoxia on the distribution of fishes associated with a petroleum platform off coastal Louisiana. North American Journal of Fisheries Management 24:662–671.

Stoner, A. W. 2004. Effects of environmental variables on fish feeding ecology: implications for the performance of baited fishing gear and stock assessment. Journal of Fish Biology 65:1445–1471.

Strelcheck, A. J., J. H. Cowan, and A. Shah. 2005. Influence of reef location on artificial-reef fish assemblages in the northcentral Gulf of Mexico. Bulletin of Marine Science 77:425–440.

Syc, T. S., and S. T. Szedlmayer. 2012. A comparison of size and age of Red Snapper (*Lutjanus campechanus*) with the age of artificial reefs in the northern Gulf of Mexico. U.S. National Marine Fisheries Service Fishery Bulletin 110:458–469.

Szedlmayer, S. T. 2011. The artificial habitat as an accessory for improving estimates of juvenile reef fish abundance in fishery management. Pages 31–43 *in* S. A. Bortone, F. P. Brandini, G. Fabi, and S. Otake, editors. Artificial reefs in fishery management. CRC Press, Boca Raton, Florida.

Szedlmayer, S. T., and J. Conti. 1999. Nursery habitats, growth rates, and seasonality of age-0 Red Snapper, *Lutjanus campechanus,* in the northeast Gulf of Mexico. U.S. National Marine Fisheries Service Fishery Bulletin 97:626–635.

Szedlmayer, S. T., and J. D. Lee. 2004. Diet shifts of juvenile Red Snapper (*Lutjanus campechanus*) with changes in habitat and fish size. U.S. National Marine Fisheries Service Fishery Bulletin 102:366–375.

Topping, D. T., and S. T. Szedlmayer. 2011. Site fidelity, residence time and movements of Red Snapper *Lutjanus campechanus* estimated with long-term acoustic monitoring. Marine Ecology Progress Series 437:183–200.

Womack, J. 2003. Small commercial fishing vessel stability analysis: where are we now? Where are we going? Marine Technology 40:296–302.

Zar, J. H. 2010. Biostatistical analysis, 5th edition. Prentice Hall, Upper Saddle River, New Jersey.

Zeileis, A., C. Kleiber, and S. Jackman. 2008. Regression models for count data in R. Journal of Statistical Software 27:1–25.