Effects of a New Artificial Reef Complex on Red Snapper and the Associated Fish Community: an Evaluation Using a Before–After Control–Impact Approach

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ARTICLE

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Abstract

Artificial reefs are commonly created with the goal of enhancing fish populations. However, many studies evaluating their effects on these populations have been hindered by a lack of preconstruction data from existing natural habitats and temporal comparisons with control areas. Here, we present findings from a before–after control–impact study designed to assess the effects of a new artificial reef on fish populations in the western Gulf of Mexico. Vertical line and fish traps were used to sample the reef site and a paired control site with soft bottom substrates for 1 year before and 2 years after reef construction. Prior to reef construction in October 2013, and over bare substrates in general, infrequent catches of sea catfishes and small coastal sharks were observed. With the exception of rare occurrences of juvenile Gray Triggerfish *Balistes capriscus* and Red Snapper *Lutjanus campechanus*, which were observed only during the summer recruitment season, the control site exhibited a lack of reef species. In contrast, we documented dramatic increases in the frequency of occurrence and abundance of multiple reef species at the reef site following the addition of structured habitat. Distinct cohorts of Red Snapper could be followed through time suggesting site fidelity, and few fish greater than age 2 years were captured indicating limited migration of older fish from other areas. Given that the reef supported high densities of juvenile

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Red Snapper that were in good condition, growing quickly, and protected from potential shrimp trawl mortality, export of juveniles (i.e., production) to the adult population was evident and likely greater on a per-unit-area basis than for the control site. Our study highlights the potential benefits of artificial reefs to species like Red Snapper; however, future studies should investigate the relative roles of emigration and fishing mortality to better understand the effects of these structures on reef fish population dynamics.

Artificial reefs are commonly created with the goal of enhancing populations of commercially or recreationally exploited marine fishes (Bohnsack and Sutherland 1985; Seaman 2000; Baine 2001; Baine and Side 2003; Broughton 2012). In the northern Gulf of Mexico (GOM), large and active artificial reef programs have resulted in the deployment of thousands of artificial reefs (Minton and Heath 1998; Kaiser and Pulsipher 2005; Gallaway et al. 2009). These artificial structures may benefit reef fish populations as they provide additional hard-bottom "reef" habitat on a shelf dominated by mud and sand substrates (Parker et al. 1983; Dufrene 2005). Several important reef fish species in the GOM including Red Snapper *Lutjanus campechanus* and Gray Triggerfish *Balistes capriscus* commonly reside at artificial reefs where they are captured in directed fisheries, and both stocks are currently considered to be overfished (Gallaway et al. 2009; Simmons and Szedlmayer 2011; SEDAR 2013, 2015). As such, understanding the influence artificial reefs may have on the population dynamics of these species is essential to their sustainable management.

Artificial reefs may confer benefits such as increased recruitment, growth, or survival if they provide additional limiting habitat, increased prey resources, or shelter from predation (Alevizon and Gorham 1989; Bohnsack 1989). The ability of an artificial reef to benefit reef fish populations may also depend on a variety of species- or life-stage-specific behaviors and life history traits, associated fishing mortality, and several aspects of artificial reef design such as reef density, location, and spacing (Bohnsack 1989; Pickering and Whitmarsh 1997; Strelcheck et al. 2005; Addis et al. 2013, 2016; Brandt and Jackson 2013). For example, off the coast of Mississippi, Brandt and Jackson (2013) found that larger Red Snapper were associated with artificial reefs with intermediate spacing and proposed that this reef configuration may have provided foraging benefits leading to increased growth. Similarly, in recent work off southern Texas, Froehlich and Kline (2015) observed larger Red Snapper were associated with lower reef density. Mudrak and Szedlmayer (2012) showed that densities of age-0 Red Snapper were significantly greater on small reefs deployed far (500 m) from large reefs than those deployed near (15 m) large reefs. They suggested the increased density on the far small reefs resulted from reduced predation as large, potential predators were observed on the large reefs. In contrast, another study off northern Florida determined that unreported artificial reefs were unlikely to provide a refuge from fishing mortality for adult Red Snapper and Gray Triggerfish due to the high degree of movement among nearby structures that can occur for these species (Addis et al. 2013, 2016). Clearly, a myriad of factors influence artificial reef function; nevertheless, continued effort to identify artificial reefs that best support the enhancement of exploited reef fish populations and understanding which species and life stages that may benefit is necessary for the effective assessment of these habitats and their future deployments as a management tool.

Evaluating the ecological performance of fishes inhabiting artificial reefs compared with those inhabiting adjacent natural habitats may promote a more comprehensive understanding of the value and function of artificial reefs in supporting marine fish populations (Carr and Hixon 1997; Love et al. 2006; Broughton 2012). Unfortunately, many evaluations of artificial reefs are hindered by a lack of background predeployment data (Brickhill et al. 2005; Cenci et al. 2011). Furthermore, studies that quantify the impact of these structures on fish communities from the onset of construction and those that monitor the recruitment of younger fishes to these habitats are sparse, leaving significant knowledge gaps regarding which artificial reefs best support fisheries production. These monitoring-based approaches are especially absent from the northwestern GOM, where the succession of fish communities following artificial reef construction remains unknown. Given the overfished status of multiple fisheries in the GOM and the expectation of future artificial reef deployments, habitat monitoring studies of this nature are particularly warranted, as such approaches can identify artificial reefs that may disproportionately contribute to the recovery and maintenance of these stocks.

In October 2013, under the guidance of the Texas Parks and Wildlife Department Artificial Reef Program, approximately 200 concrete box culverts and 470 prefabricated reef pyramids were deployed off the Texas coast in the western GOM to create the Corpus Christi Nearshore Reef (CCNR). The construction of the CCNR represented a unique opportunity to better understand artificial reef colonization and recruitment processes. The primary goal of this study was to characterize the reef fish community at nearby natural bottom habitats and the CCNR both before and after reef construction. We specifically evaluated the (1) relative abundance, (2) size structure, and (3) age of fishes recruiting to the CCNR. Our comparisons focused on Red Snapper and, to a lesser extent, Gray Triggerfish given their importance to fisheries in the region.

STUDY AREA

This study occurred within the coastal waters of the Texas continental shelf in the western GOM. The region is characterized by a gently sloping shelf covered in terrigenous sediments consisting of silt and clay muds and a low availability

of natural hard substrates with vertical relief ≥ 1 m (Parker et al. 1983; Rezak et al. 1985). The CCNR site (officially known as MU-775) and a nearby control site were located approximately 15 km offshore near Port Aransas, Texas (Figure 1). The control site was approximately 3.5 km northeast of the CCNR site and was selected to mimic environmental conditions at the CCNR site prior to reef construction (i.e., both sites had water depths of 22 m and ambient bare substrates of sand and silt muds). Both sites are influenced by turbidity stemming from coastal runoff and a persistent but variable nepheloid layer of resuspended sediment (Shideler 1981). Artificial structures consisting of 470 prefabricated concrete pyramids (with embedded limestone rock; 3 m base × 2.4 m height) and 200 concrete box culverts (various sizes; from 1.2 m × 1.2 m to 3 m × 3 m) were deployed across an approximately 11-ha area at the 64.75-ha CCNR site in October 2013. Though not a focus of this study, the 47-m MV Kinta S was also scuttled in this reef block approximately 330 m southeast of the existing structure at the CCNR site in September 2014.

METHODS

Study design and sampling procedure.—This study used a before–after control–impact (BACI) framework to assess the effect of an environmental impact, such as the construction of the CCNR (e.g., Stewart-Oaten et al. 1986; Underwood 1994). The relative abundance of reef fish at the CCNR site and the bare control site was quantified for 1 year before reef construction and 2 years after (i.e., summer 2012–summer 2015; reef deployment occurred October 2013) using vertical line surveys and small fish traps, with the exception that small fish traps were not used until 2 months prior to reef construction. Sampling at both sites was generally conducted once per season (fall: October–December; winter: January–March; spring: April–June; summer: July–September); however, for the first 6 months following reef construction, sampling occurred monthly to better monitor colonization rates. Sampling dates during each quarter were selected based on when suitable sea conditions occurred. At each site, a combination of either random sampling (predeployment) or stratified random sampling (postdeployment) was used. Before
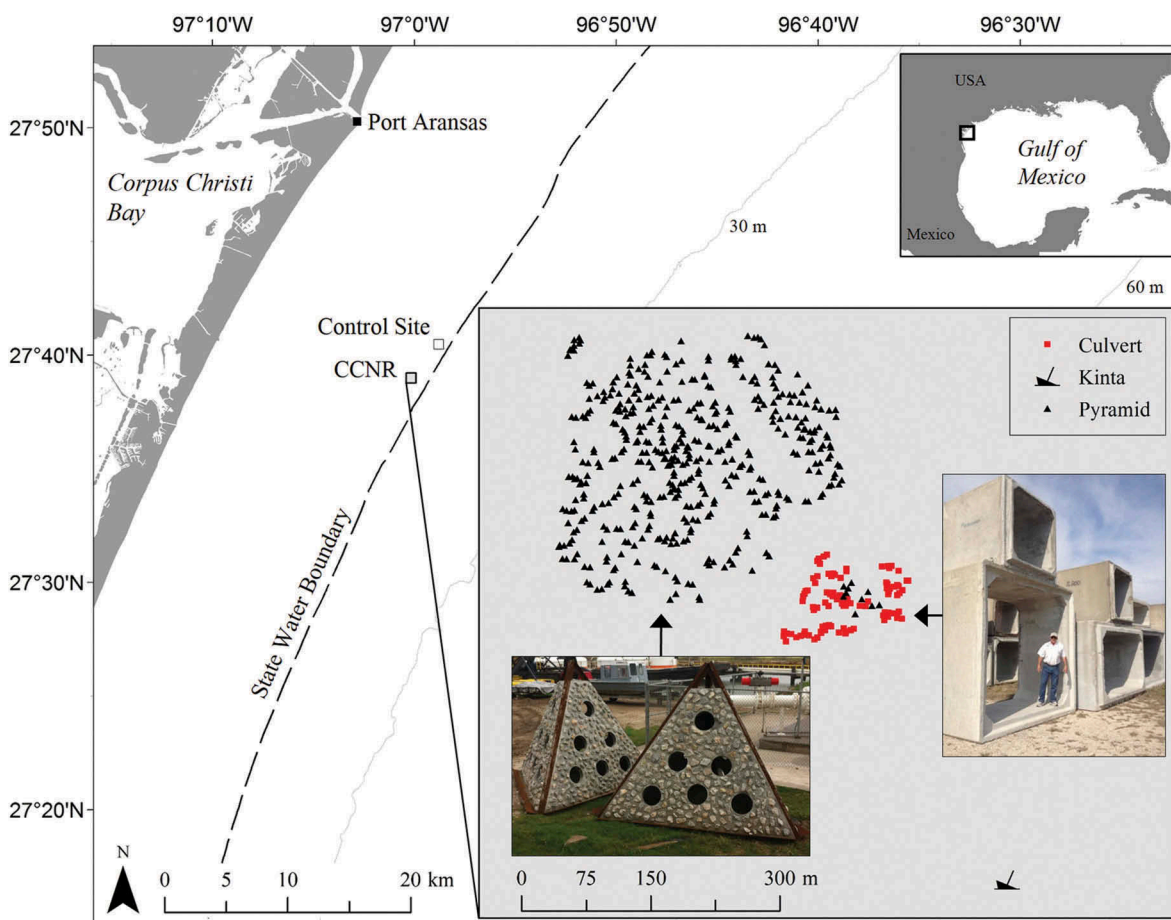
FIGURE 1. Study area near Port Aransas, Texas, in the northwestern Gulf of Mexico; relative location is indicated in the inset map (top right). The Corpus Christi Nearshore Reef (CCNR) site (gray square) and the bare control site (white square) were monitored with vertical line and fish traps for 1 year before reef construction and 2 years after (i.e., summer 2012 through summer 2015). The enlarged reef site displays the configuration of structures deployed at the CCNR site.

reef construction, nine locations for fish trap deployments and three locations for vertical line sets were randomly selected within each site. After reef construction, sampling locations at the CCNR site were selected using stratified random sampling (i.e., stratified by area of structure types). Using this protocol, six pyramids and three culverts were randomly selected for fish trap deployments, while two pyramids and one culvert were randomly selected for vertical line sets. Sampling locations were selected using the Create Random Points tool in ArcMap 10.3.1 (ESRI 2015). To increase the likelihood of spatial independence, a minimum distance criterion of 35 m between sampling locations was specified. On a given sampling day, a trap was deployed at the nine preselected sampling locations at each site (i.e., nine replicate traps at control site; nine replicate traps at CCNR site), while one vertical line set (see below for description) was conducted at each of the preselected sampling locations at each site (i.e., three replicate sets at control site; three replicate sets at CCNR site). Water temperature (°C), dissolved oxygen (DO: mg/L), and salinity (‰) were measured at each site with a Hydrolab DS5 data sonde.

Visual census methods were considered unreliable for quantifying fish abundance due to consistently poor visibility (≤1 m); therefore, we used both small fish traps and vertical lines to help ensure representative samples as both gears may have differing species or size selectivity (e.g., Wells et al. 2008a; Gregalis et al. 2012). Small fish traps (0.97 m long × 0.67 m wide × 0.64 m high), identical to those used by Brandt and Jackson (2013), were used to sample fish inhabiting the study sites. Small mesh size (6 cm stretch measure) and funnel mouth openings (17.5 × 11.5 cm) likely increased selectivity of juvenile fishes. Fish traps were baited with 0.5 kg of cut Atlantic Mackerel *Scomber scombrus* and were allowed to soak for approximately 2 h before retrieval. Vertical lines followed the Southeast Area Monitoring and Assessment Program (SEAMAP) protocol (SEAMAP 2013), and gear consisted of 136-kg-test (300 lb) monofilament mainline connected to a 7.3-m backbone (i.e., leader) constructed with 181-kg-test (400 lb) monofilament. The backbone contained 10 equally spaced 45-kg-test (100 lb) monofilament gangions, each terminating with identical circle hooks (Mustad 39960 D; 8/0, 11/0, or 15/0 sizes; same-sized hooks fished on a backbone) baited with cut Atlantic Mackerel. A 4.5-kg sash weight was attached to the end of the backbone to allow the gear to fish vertically. A vertical line "set" consisted of one deployment of each hook size (i.e., three drops of 10 hooks each; 30 hooks fished per set). Therefore, upon arrival at the sampling location, a randomly selected hook size was deployed over either the port or starboard bow of the vessel and allowed to soak for 5 min. The gear was then retrieved, and a second randomly chosen hook size (of the two remaining) was immediately deployed off the opposite side of the vessel. Following retrieval of this second deployment, the backbone containing the third (unused) hook size was fished.

Hook sizes were rotated such that each hook size was fished on the first, second, and third drop at a site on a given sampling day. Three replicate sets were conducted at each site on each sampling day.

Fishes were identified to species and retained for further processing. In the laboratory, fish were measured (SL, FL, stretched TL; mm), weighed (kg), and sexed, and sagittal otoliths of Red Snapper were extracted. Otoliths were processed following the guidelines of VanderKooy (2009). Briefly, the left otolith of each individual was weighed (g) and then thin-sectioned in the transverse plane (0.5 mm thickness) using an IsoMet 1000 precision sectioning saw. Sections containing the core region were mounted to microscope slides with thermoplastic cement and then viewed under a dissecting microscope with reflected light. Two readers independently counted all opaque annuli on a random subsample of Red Snapper otoliths ($n = 50$), and ages were assigned based on the count of annuli and the degree of marginal edge completion (Allman et al. 2005). Because Red Snapper generally form an annulus sometime between January and June in the northern GOM, fish captured on or before June 30 had their age advanced by 1 year if the otolith displayed a large translucent edge (VanderKooy 2009). For fish captured after June 30, age was equal to the annulus count. Following this standard convention for aging Red Snapper, an annual age-cohort was based on calendar year rather than time since spawning (Jearld 1983; Allman et al. 2002, 2005; VanderKooy 2009). Agreement and precision of age assignments between readers was evaluated using linear regression, the CV (SD/mean), and average percent error (APE).

Data analyses.—Species accumulation curves were used to qualitatively assess the presence of different fish species over time at both the control site and the CCNR site. The CPUE from both vertical line (number of fish per set) and fish trap (number of fish per trap-hour) surveys was calculated for each species at the control site and the CCNR site before and after reef construction. Relative abundance (RA; %) was estimated for each species captured at the CCNR site and was calculated as the proportion of the total fish catch before and after reef construction. Subsequently, the change in relative abundance (ΔRA; %) was calculated by subtracting the before-deployment RA from the after-deployment RA. A positive ΔRA value was interpreted as an increase in relative abundance, while a negative value suggested a decrease in relative abundance (Reese et al. 2008; Hall et al. 2016).

The CPUE data were analyzed in a BACI design using a partially nested hierarchical ANOVA. The model included before–after (BA), control–impact (CI), and their interaction (BA × CI) as main effects. Sampling date was nested within BA and was treated as a random effect to account for temporal dependence among samples. Using this model, changes in CPUE attributable to the construction of the CCNR were evaluated for total numbers of fish (i.e., all species included) and several reef fish species of interest

(e.g., Red Snapper and Gray Triggerfish). Tests for significant main effects were carried out in R 3.2.3 (R Core Team 2015) using functions from the nlme package (Pinheiro et al. 2017). Separate analyses were conducted for traps and vertical line surveys because of their differing CPUE metrics. Prior to testing, CPUE data were log transformed to minimize heteroscedasticity. If the main-effects ANOVA detected a significant BA × CI interaction, Welch's t-test was used to examine potential differences in mean CPUE at the control site and CCNR before versus after reef construction. All tests of significance were conducted using α = 0.05.

Changes in size structure over time were examined for Red Snapper and Gray Triggerfish using length frequency histograms. Length frequencies were plotted by season and included pooled data from vertical lines and fish traps as both gears were used during each season. Because age data were available for Red Snapper, otolith-derived ages were overlain for each fish represented in the length frequency distribution. This allowed confirmation that modal length classes were indeed representing distinct age-groups and facilitated visual tracking of a cohort through time. If it was possible to follow a particular cohort through time, we interpreted this as evidence of site fidelity and continued use of the CCNR over time.

Change in size and age over time was further evaluated using Pearson's correlation coefficient. We specifically tested for changes in Red Snapper mean TL, weight, and age and Gray Triggerfish FL and weight with reef age. Reef age was calculated as the time in years since reef construction. Linear regression was used to estimate Red Snapper growth at the CCNR. Only size-at-age data for fish captured in the 2 years following reef construction were included to help ensure that estimated growth was representative of fish inhabiting the CCNR. To assess the condition of Red Snapper at CCNR, relative weight (W_r) was calculated following the equation of Wege and Anderson (1978):

$$W_r = W/W_s \times 100,$$

where W is the measured weight of a fish and W_s is the predicted standard weight for a fish of the same length estimated from a weight–length regression for the species. Predicted weights (W_s) were calculated from the weight–length regression reported for Red Snapper in the latest benchmark stock assessment (equation 4 in SEDAR 2013). Fish were considered in good condition if their W_r was ≥100. To assess any changes in condition over time, Pearson's correlation coefficient was used to test for a significant relationship of mean W_r and reef age.

RESULTS

Over the course of our study, 504 fish representing 17 species and 11 families were collected (Table 1). Vertical line surveys captured 124 of these fish (24.6%), representing seven species and five families, and fish traps captured the remaining 380 fish (75.4%), which represented 14 species from 11 families. Prior to reef construction, few fish were captured at the control site or the planned CCNR site. For example, vertical lines captured only three ariid catfish (e.g., Gafftopsail Catfish Bagre marinus and Hardhead Catfish Ariopsis felis), while traps captured 28 fish, 21 (75%) of which were Gray Triggerfish. After construction of the CCNR, vertical lines captured 121 fish, while fish traps captured 352 fish. Approximately 94% of these fish were captured at the CCNR site; accordingly, large increases in CPUE of Red Snapper and Gray Triggerfish were observed with both sampling gears after the construction of CCNR (Table 1).

Water quality varied seasonally but was similar at the CCNR and the nearby control site over time. Benthic water temperatures ranged from 13.9°C in winter to 28.7°C in the summer. Evidence of thermal stratification was generally present in the summer months with thermoclines present around the 10–15-m depths. An exception was summer 2015 when no thermocline was present. Benthic DO levels were generally highest in winter (mean = 7.83 mg/L) and lowest in fall (mean = 5.6 mg/L). Hypoxia (i.e., DO < 2 mg/L) was not observed during most of the study but was recorded on one occasion in late spring 2015 (1.7 mg/L) over both the control and reef sites. Benthic salinity levels ranged from 31.8‰ in the fall to 36.4‰ in the summer months.

Assessment of species accumulation curves for both sites revealed the number of species observed following reef construction increased more rapidly at the CCNR site. Prior to reef construction, five species were observed at the planned CCNR site, while four were observed at the bare control site. Notably, Red Snapper and Gray Triggerfish were observed at each site before reef construction. In the first fall and winter after reef construction, only one additional species (Spinner Shark Carcharhinus brevipinna) was observed at the control site, while six additional species were observed at the CCNR site: sciaenids (Sand Seatrout Cynoscion arenarius, Atlantic Croaker Micropogonias undulatus, and Southern Kingfish Menticirrhus americanus), reef fishes (Warsaw Grouper Hyporthodus nigritus and Pigfish Orthopristis chysoptera), and the migratory Cobia Rachycentron canadum. Interestingly, the sciaenid fishes were not observed in samples after the first fall and winter after reef construction. Maximum observed species richness at both the CCNR site (13 species) and control site (eight species) was observed by the first summer after reef construction, and no additional species were observed at either site for the remainder of the study.

Changes in the RA of many species were most pronounced at the CCNR site following reef construction. Vertical lines failed to capture a single Red Snapper before reef construction, but RA increased dramatically to 94.2% in postdeployment samples (Table 2). Although less apparent, several other

TABLE 1. Fish species captured at the CCNR site and control site before and after reef construction. Mean CPUE (fish/set or fish/trap-hour), SE, and sample size (n) are presented separately for vertical lines and fish traps. Sample size (n) represents the number of vertical line sets or fish trap deployments used in calculations.

| Species | Control site | | | | | | CCNR | | | | | |
| | Before reefing | | | After reefing | | | Before reefing | | | After reefing | | |
	Mean	SE	n	Mean	SE	n	Mean	SE	n	Mean	SE	n
Vertical lines (fish/set)												
All fish	0.182	0.122	11	0.030	0.030	33	0.083	0.083	12	3.636	0.762	33
Red Snapper *Lutjanus campechanus*	0	0	11	0	0	33	0	0	12	3.424	0.753	33
Sand Seatrout *Cynoscion arenarius*	0	0	11	0	0	33	0	0	12	0.152	0.098	33
Lane Snapper *Lutjanus synagris*	0	0	11	0	0	33	0	0	12	0.030	0.030	33
Warsaw Grouper *Hyporthodus nigritus*	0	0	11	0	0	33	0	0	12	0.030	0.030	33
Atlantic Sharpnose Shark *Rhizoprionodon terraenovae*	0	0	11	0.030	0.009	33	0	0	12	0	0	33
Gafftopsail Catfish *Bagre marinus*	0.091	0.091	11	0	0	33	0	0	12	0	0	33
Hardhead Catfish *Ariopsis felis*	0.091	0.091	11	0	0	33	0.083	0.083	12	0	0	33
Fish traps (fish/trap-hour)												
All fish	0.126	0.053	27	0.053	0.022	99	0.090	0.031	27	1.077	0.148	99
Red Snapper *Lutjanus campechanus*	0.014	0.010	27	0.002	0.002	99	0.015	0.010	27	0.455	0.091	99
Gray Triggerfish *Balistes capriscus*	0.094	0.043	27	0.004	0.003	99	0.060	0.024	27	0.420	0.092	99
Pigfish *Orthopristis chrysoptera*	0	0	27	0	0	99	0	0	27	0.164	0.043	99
Warsaw Grouper *Hyporthodus nigritus*	0	0	27	0	0	99	0	0	27	0.015	0.007	99
Lane Snapper *Lutjanus synagris*	0	0	27	0	0	99	0	0	27	0.005	0.004	99
Hardhead Catfish *Ariopsis felis*	0.017	0.017	27	0.037	0.021	99	0	0	27	0.005	0.003	99
Atlantic Croaker *Micropogonias undulatus*	0	0	27	0	0	99	0	0	27	0.004	0.004	99
Conger Eel *Conger oceanicus*	0	0	27	0	0	99	0	0	27	0.004	0.004	99
Southern Kingfish *Menticirrhus americanus*	0	0	27	0	0	99	0	0	27	0.002	0.002	99
Cobia *Rachycentron canadum*	0	0	27	0	0	99	0	0	27	0.002	0.002	99
Atlantic Bumper *Chloroscombrus chrysurus*	0	0	27	0.003	0.003	99	0	0	27	0	0	99
Blue Runner *Caranx crysos*	0	0	27	0.002	0.002	99	0.007	0.007	27	0	0	99
Pinfish *Lagodon rhomboides*	0	0	27	0	0	99	0.008	0.008	27	0	0	99
Spinner Shark *Carcharhinus brevipinna*	0	0	27	0.005	0.003	99	0	0	27	0	0	99

species including Sand Seatrout, Lane Snapper *Lutjanus synagris*, and Warsaw Grouper also displayed increased RA in vertical line samples after being absent or undetected before reef construction (i.e., 0% RA). Among trap samples, the greatest changes in RA occurred for Gray Triggerfish (−24% ΔRA), Red Snapper (+23.1% ΔRA), and Pigfish (+13.9% ΔRA). Despite the decline in Gray Triggerfish RA, this species accounted for the greatest RA in trap samples after reef construction (42.3%). Red Snapper RA increased from 16.7% before reef construction to 39.8% after reef construction, while Pigfish RA increased from 0% to 13.9% following reef construction (Table 2). Similar to vertical line samples, other species of recreational or commercial importance also showed increased RA in trap samples after being absent in prereef samples.

Abundance of fish at the control site generally remained low and similar to predeployment levels after reef construction, but large increases in CPUE were observed for several groups at the CCNR site. The construction of CCNR had a significant effect on vertical line total fish CPUE (BA × CI: $F_{1, 73}$ = 16.82, P < 0.001) and trap total fish CPUE (BA × CI: $F_{1, 236}$ = 31.02, P < 0.001; Table 3). Post hoc testing suggested that no differences in CPUE existed at the control site before versus after reef construction for vertical line (t = 1.21, df = 11, P = 0.253) or fish trap data (t = 1.57, df = 32, P = 0.126). In contrast, vertical line and fish trap CPUE increased at the CCNR site following reef construction (t = 4.99, df = 41, P < 0.001 and t = 5.40, df = 60, P < 0.001, respectively; Table 1). There was a significant effect of the CCNR on Gray Triggerfish abundance (BA × CI: $F_{1, 236}$ = 17.33, P < 0.001; Table 3). At the control site, CPUE declined from 0.094 to 0.004 fish/trap-hour after reef construction (t = 2.45, df = 27, P = 0.021; Table 1), while CPUE increased at the CCNR site from 0.060 fish/trap-hour before reef construction to 0.420 fish/trap-hour after reef construction (t = 2.71, df = 63, P = 0.008; Figure 2). Red Snapper abundance estimated from

TABLE 2. Total catch, CPUE (fish/set or fish/trap-hour), relative abundance (RA; %), and change in relative abundance (ΔRA; %) of fish at the CCNR site before and after reef construction.

Species	Before reefing			After reefing			
	Catch	CPUE	RA	Catch	CPUE	RA	ΔRA
Vertical lines (fish/set)							
Red Snapper	0	0	0	113	3.424	94.2	94.2
Sand Seatrout	0	0	0	5	0.152	4.2	4.2
Lane Snapper	0	0	0	1	0.030	0.8	0.8
Warsaw Grouper	0	0	0	1	0.030	0.8	0.8
Hardhead Catfish	1	0.083	100	0	0	0	−100
All fish	1	0.000		120	3.636		
Fish traps (fish/trap-hour)							
Gray Triggerfish	8	0.060	66.7	137	0.420	42.3	−24.4
Red Snapper	2	0.015	16.7	129	0.455	39.8	23.1
Pigfish	0	0	0	45	0.164	13.9	13.9
Warsaw Grouper	0	0	0	5	0.015	1.5	1.5
Hardhead Catfish	0	0	0	2	0.005	0.6	0.6
Lane Snapper	0	0	0	2	0.005	0.6	0.6
Atlantic Croaker	0	0	0	1	0.004	0.3	0.3
Cobia	0	0	0	1	0.002	0.3	0.3
Conger Eel	0	0	0	1	0.004	0.3	0.3
Southern Kingfish	0	0	0	1	0.002	0.3	0.3
Blue Runner	1	0.007	8.3	0	0	0	−8.3
Pinfish	1	0.008	8.3	0	0	0	−8.3
All fish	12	0.090		324	1.077		

vertical lines and fish traps was also significantly affected by the construction of the CCNR (BA × CI: $F_{1, 73} = 17.99$, $P < 0.001$ and BA × CI: $F_{1, 236} = 14.21$, $P < 0.001$, respectively; Table 3). Vertical line CPUE was significantly greater at the CCNR site after reef construction, increasing from 0 to 3.42 fish/set ($t = 6.44$, df = 32, $P < 0.001$; Figure 2A). Red Snapper trap CPUE did not change at the control site before versus after reef construction ($t = 1.23$, df = 27, $P = 0.228$) but increased at the CCNR from 0.02 fish/trap-hour before reef construction to 0.46 fish/trap-hour after ($t = 5.14$, df = 114, $P < 0.001$; Figure 2B). Although Lane Snapper and Warsaw Grouper were not captured prior to reef construction, the construction of the CCNR did not significantly increase their abundance estimated from vertical lines (BA × CI: $F_{1, 73} = 0.35$, $P = 0.558$) or fish traps (BA × CI: $F_{1, 236} = 0.58$, $P = 0.447$ and BA × CI: $F_{1, 236} = 1.41$, $P = 0.237$, respectively; Figure 2; Table 3).

Red Snapper and Gray Triggerfish abundance increased over time after the construction of the CCNR (Figure 3). Vertical line sampling suggested Red Snapper CPUE increased slowly following reef construction, but CPUE increased to approximately 9–11 times greater than CPUE immediately after reef construction by the first summer (Figure 3A). No Red Snapper were captured at the control

site with vertical lines over the course of the study. Red Snapper trap CPUE was more variable than vertical line CPUE but also showed increases in CPUE following reef construction (Figure 3B). Similar to vertical line CPUE, Red Snapper trap CPUE reached the greatest observed levels by the first summer after reef construction—approximately 32 times greater than the highest CPUE observed before reef construction (maximum CPUE before = 0.04 fish/trap-hour; maximum CPUE after = 1.31 fish/trap-hour). Both gear types showed a decline in CPUE the second winter after reef construction followed by an increase to prewinter levels by summer. Hypoxia was observed during this period of lower abundance. Low Red Snapper CPUE (e.g., 0.02–0.04 fish/trap-hour) was observed at the control site but only during the summer season.

Gray Triggerfish were only captured in fish traps, and CPUE increased at the CCNR site following reef construction (Figure 3C). Gray Triggerfish CPUE also displayed a potential seasonal trend in CPUE with declines observed during the first two winters after reef construction followed by increases the following summer and fall. Peak CPUE was observed by the second summer following reef construction, reaching levels approximately nine times greater than the maximum observed the at the CCNR site before reef construction (maximum

TABLE 3. Results of partially nested, hierarchical, two-way ANOVA testing for the effect of constructing the CCNR on overall abundance of fish and several species of economically important reef fish species. Type III tests for the main effects of before versus after (BA), control versus impact (CI), and their interaction (BA × CI) are displayed for both vertical line (fish/set) and fish trap data (fish/trap-hour). No Gray Triggerfish were captured with vertical lines in this study. An asterisk (*) denotes statistical significance ($P \leq 0.05$).

Source	Vertical line			Fish trap		
	df	*F*-value	*P*-value	df	*F*-value	*P*-value
All fish						
BA	12	3.78	0.076	12	1.17	0.301
CI	73	10.02	0.002*	236	32.68	<0.001*
BA × CI	73	16.82	<0.001*	236	31.02	<0.001*
Gray Triggerfish						
BA				12	0.03	0.870
CI				236	14.68	<0.001
BA × CI				236	17.33	<0.001
Red Snapper						
BA	12	5.54	0.037*	12	2.37	0.150
CI	73	17.99	<0.001*	236	14.32	<0.001*
BA × CI	73	17.99	<0.001*	236	14.21	<0.001*
Lane Snapper						
BA	12	0.32	0.583	12	0.26	0.621
CI	73	0.35	0.558	236	0.58	0.447
BA × CI	73	0.35	0.558	236	0.58	0.447
Warsaw Grouper						
BA	12	0.32	0.583	12	1.32	0.274
CI	73	0.35	0.558	236	1.41	0.237
BA × CI	73	0.35	0.558	236	1.41	0.237

CPUE before = 0.18 fish/trap-hour; maximum CPUE after = 1.63 fish/trap-hour). Low abundances of Gray Triggerfish were observed at the control site but only during the summer season (Figure 3C).

Annual age estimates were obtained for 247 Red Snapper captured during this study. Agreement between readers was high (reader 1 age = 1.01 × reader 2 age + 0.01, $r^2 = 0.94$), and variability between age assignments was low (CV = 1.35, APE = 0.95%). Ages ranged from 0 to 4 years and catches consisted primarily of age-2 fish (59.9%) followed by age-1 fish (31.5%). Age-0 Red Snapper (mean TL = 112.8 mm) were first captured in traps during the summer (i.e., August) at the bare control and CCNR site prior to reef construction (Figure 4A). It was possible to identify three cohorts of Red Snapper inhabiting the CCNR over time when assigned ages were overlain onto the seasonal length frequency histograms (Figure 4A). The first cohort consisted of age-0 fish that recruited to the CCNR the first fall after reef construction (i.e., fall 2013). By late fall, more age-0 fish were captured along with several age-1 fish. No age-0 fish were captured at the CCNR after the first fall after reef construction. This cohort was classified as age-1 fish in winter 2014 and then age-2 fish in winter 2015, although length frequency data were

limited that winter. Nevertheless, this cohort was visible again by spring and displayed a modal size of approximately 325–350 mm TL by summer 2015. The second Red Snapper cohort recruited to the CCNR as age-1 fish in fall 2013. This cohort was well represented in the length frequency histograms until fall 2014. Few fish from this cohort remained after fall 2014, and few age-3 fish were represented through summer 2015. The third Red Snapper cohort was apparent in summer 2015 and was represented by a strong supply of new age-1 recruits (Figure 4A). Gray Triggerfish seasonal length frequencies were similar to Red Snapper in that the smallest fish were sampled in the summer prior to reef construction and the first fall following reef construction (Figure 4B). Gray triggerfish appeared to be absent from the CCNR during the winter months as only one fish was captured in the two winters that sampling occurred. Gray Triggerfish had weak representation in the seasonal length frequencies until summer 2015, the second summer after reef construction.

Red Snapper mean age increased from <1 year immediately following reef construction to nearly 2 years by the end of this study ($r = 0.76$, $P = 0.004$). Red Snapper mean length ($r = 0.71$, $P = 0.010$) and weight ($r = 0.78$, $P = 0.003$) also increased as the age of the CCNR increased. Positive

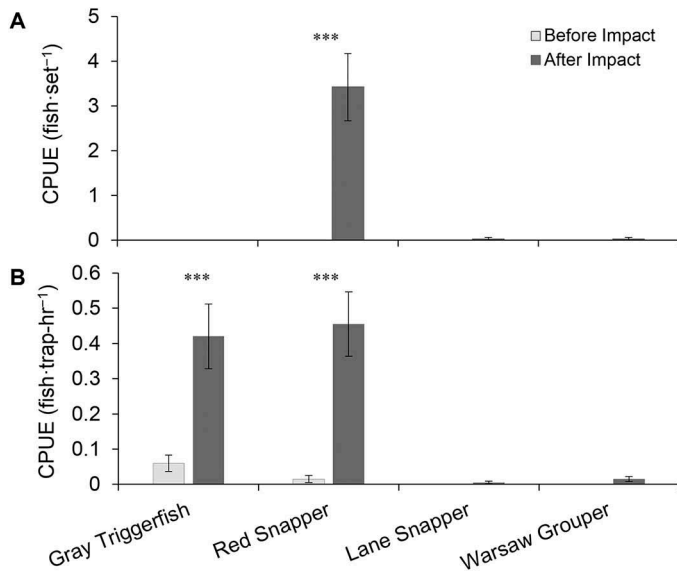FIGURE 2. Mean CPUE (±SE) of select reef fishes at the CCNR site before and after reef construction. Data are displayed separately for vertical lines (**A**; fish/set) and fish traps (**B**; fish/trap-hour). Significant differences ($P \leq 0.05$) are denoted with asterisks (***).

catfishes (Ariidae) and several small shark species (Carcharhinidae). Gray Triggerfish and Red Snapper were the dominant members of the fish community after construction of the CCNR. Although individuals of both of these fisheries species were also captured over bare substrates at the control site and the CCNR site prior to reef construction, they were representative of young of the year based on the sizes at capture and presence only during the summer months —the peak spawning and recruitment season for these species (Gallaway et al. 2009; Simmons and Szedlmayer 2011, 2012). Most notably, Red Snapper and Gray Triggerfish abundances increased substantially at the CCNR following reef construction, while both species remained undetected or occurred infrequently in low abundances at the bare control site. As these changes did not occur at the bare control site, they can clearly be attributed directly to the addition of artificial hard substrate at the newly reefed CCNR site.

Increased abundance of economically important reef fish following the construction of an artificial reef is not uncommon as colonization rates are often rapid (Bohnsack et al. 1991, 1994; Grossman et al. 1997). For example, Bohnsack et al. (1994) observed peak species richness, number of individuals, and biomass within 2 months of reef construction. In
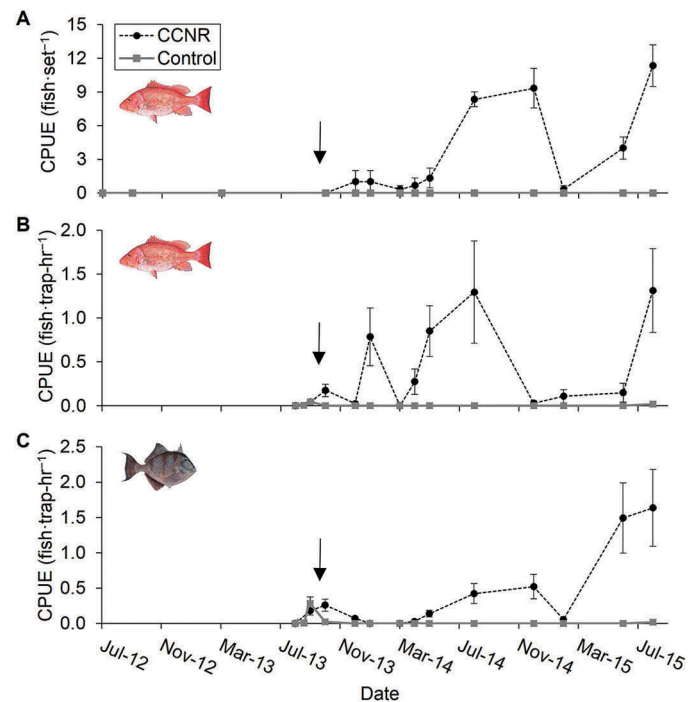
relationships with reef age were also observed for Gray Triggerfish mean FL ($r = 0.69$, $P = 0.060$) and weight ($r = 0.60$, $P = 0.116$), but these relationships were not significant. A linear regression of Red Snapper size-at-age data suggested that fish were growing approximately 124 mm/year [TL = $124.1 \times$ age (years) + 80.3; $r^2 = 0.81$, $P < 0.001$], aligning well with modal sizes from the seasonal length frequency histograms. Evaluation of Red Snapper relative weights suggested that fish were in good health while inhabiting the CCNR (mean $W_r = 126.17$, SE = 0.74). There was no evidence of a relationship between Red Snapper condition (i.e., W_r) and reef age ($r = -0.24$, $P = 0.510$).

DISCUSSION

Understanding the effects artificial reefs have on marine ecosystems and their role in maintaining marine fish populations remains an important issue in modern fisheries management. This research incorporated two experimental features highlighted by Brickhill et al. (2005) as lacking in most previous studies of artificial reef function as fish habitat. Specifically, our study included comparisons with natural control substrates and temporal comparisons of fish age and length data over time. Results from this study suggest that the construction of the CCNR had significant positive impacts on the reef fish community by increasing the abundance of several key species. For example, after reef construction we observed an increased frequency of occurrence for several economically important species at the CCNR, while the control site remained characterized by infrequent catches of sea

FIGURE 3. Relative abundance of Red Snapper and Gray Triggerfish over time. Mean (±SE) Red Snapper CPUE using vertical lines (**A**; fish/set) and fish traps (**B**; fish/trap-hour) is plotted for the CCNR (black circles) and bare control site (gray squares) from summer 2012 through summer 2015. Mean (±SE) Gray Triggerfish CPUE (**C**; fish/trap-hour) at the CCNR (black circles) and control site (gray squares) is also displayed for comparison. The black arrow on each panel represents the time of reef construction.
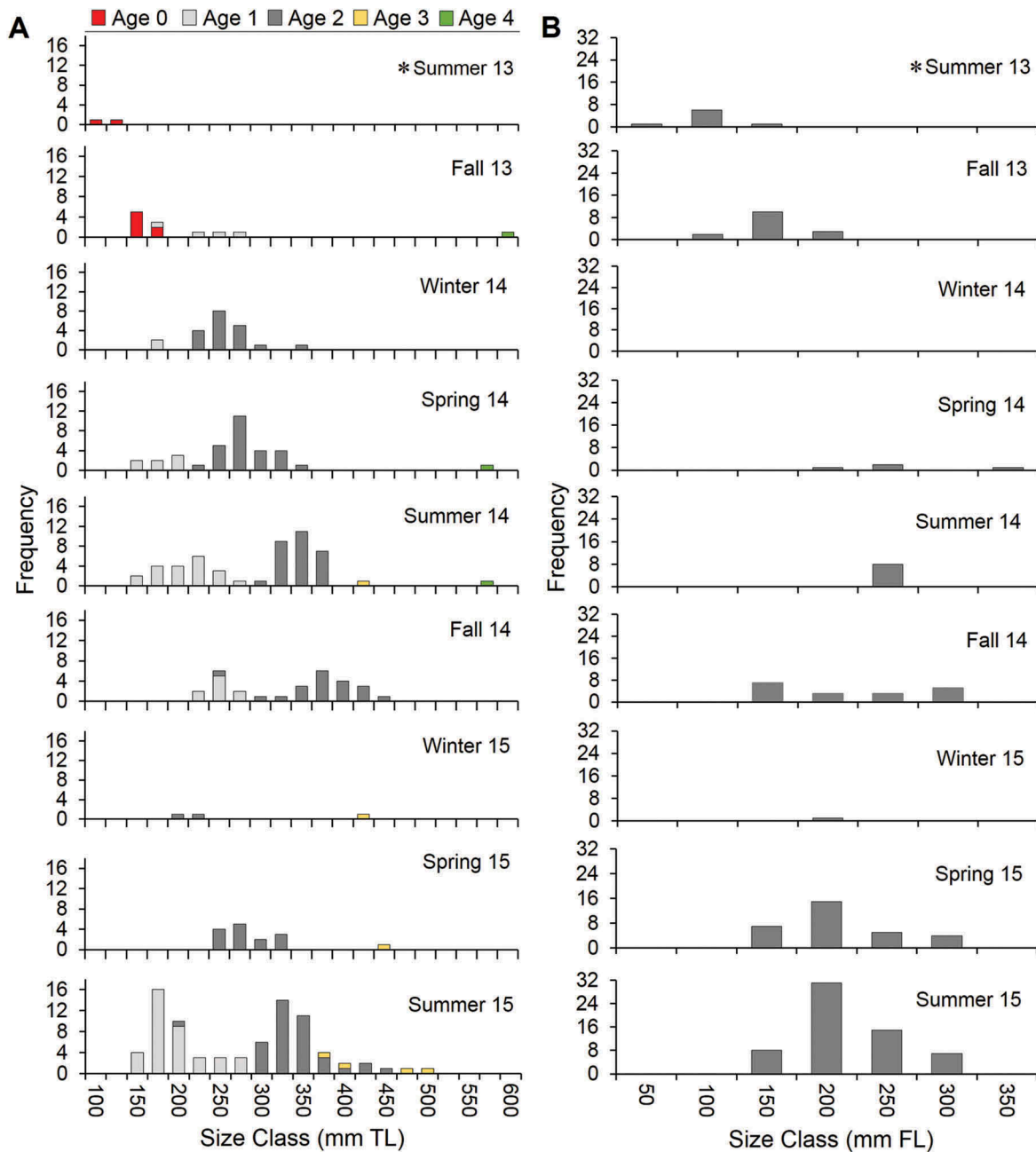
FIGURE 4. Length frequency histograms by season for (A) Red Snapper and (B) Gray Triggerfish captured at the CCNR site from summer 2013 through summer 2015. Red Snapper length frequencies include pooled data from vertical lines and fish traps, while Gray Triggerfish length frequencies include only trap-caught fish as no fish were captured with vertical lines during the study. Red Snapper age data have been overlain onto the length frequencies and confirm the presence of distinct cohorts through time. The astersisks (*) indicate prior to reef construction (summer 2013).

contrast, although rapid colonization was observed, we did not observe peak abundance of Red Snapper until the first summer and fall following reef construction (i.e., ≥11 months after reef construction), and Gray Triggerfish peak abundance was not observed until the second summer following reef construction (i.e., ≥23 months after reef construction). The slower

colonization pattern observed for Gray Triggerfish may be due to the species' preference for encrusting and reef-dwelling invertebrates (Vose and Nelson 1994), which take time to colonize the "clean" structure. Given that the CCNR was constructed in October, cooler water temperatures associated with the fall and winter seasons likely slowed or delayed

growth of Gray Triggerfish prey, which may have contributed to the seasonal decrease in abundance observed for Gray Triggerfish during the winter months. Prey availability on the CCNR structures may have had a lesser effect on Red Snapper as the species relies on prey sources from reef structures as well as open sand or mud bottom habitats (Moseley 1966; Ouzts and Szedlmayer 2003; Szedlmayer and Lee 2004; McCawley and Cowan 2007; Wells et al. 2008b). The slower rates of colonization observed in this study may also stem from the size of the artificial reefs studied as the largest artificial reefs examined by Bohnsack et al. (1994) were approximately 14 m^2, while the CCNR was a much larger complex of artificial structures spanning 11 ha. Consequently, the greater habitat area provided by the CCNR may have contributed to the longer colonization times we observed. Thus, more research is warranted to examine the influence of reef size on the population dynamics of these reef fishes.

Variable patterns in abundance that were observed may be influenced by a combination of environmental conditions, other biological processes like competition, and sampling effort. For example, the presence of a relatively rare hypoxia event in spring 2015 was likely driven by heavy rainfall totals and subsequent runoff observed in spring 2015 (https://water watch.usgs.gov/2015summary/) and may have contributed to lower Red Snapper CPUE, as indicated by previous studies that have also observed depressed catches of juvenile Red Snapper when hypoxia is present (Gallaway et al. 1999). The lone occurrence of hypoxia does not explain the winter declines in CPUE; however, we did observe strong currents during winter sampling. Thus, while it is possible that Red Snapper abundance declined during winter, it is also likely that gear efficiency may have declined during these conditions. The occurrence of several sciaenid species (Sand Seatrout, Atlantic Croaker, and Southern Kingfish) during the first fall and winter following reef construction may be a function of their broad presence over open sand and mud bottoms of the inner shelf (Hoese and Moore 1998). Likewise, their absence in samples after this period may be a result of competition with more reef associated species like Red Snapper and Gray Triggerfish for space or other resources at the CCNR. Finally, the variable patterns in abundance may simply be a function of sampling effort (e.g., sampling was limited by constraints associated with offshore logistics and relatively infrequent [quarterly] sampling trips). Additional sampling gear types (e.g., cameras or visual census) and more frequent sampling would be needed to better evaluate these hypotheses.

Age-0 and age-1 Red Snapper cohorts first recruited to the CCNR during the first fall (2013) and winter (e.g., for 2014 classified as age-1 and age-2 fish) following reef construction. Interestingly, no age-0 fish were captured after the first fall when reef construction occurred, which may be explained in part by gear selectivity and Red Snapper behavior. For example, age frequencies indicated Red Snapper were not fully recruited to trap gear until age 1; therefore, age-0 fish were

likely undersampled in our study. In fact, most studies of age-0 snapper have required the use of trawl gear to successfully sample early juvenile Red Snapper (Holt and Arnold 1982; Gallaway and Cole 1999; Rooker et al. 2004; Wells et al. 2008c). Another explanation for the lack of age-0 Red Snapper in these samples may be a behavioral exclusion of these fish from the immediate reef structures or at least the traps by older Red Snapper (Bailey et al. 2001; Mudrak and Szedlmayer 2012). For example, Bailey et al. (2001) showed older Red Snapper actively excluded age-0 conspecifics from occupying experimental reef structures. These hypotheses are further supported by ancillary trap-camera deployments at the CCNR that show the presence of age-0 Red Snapper away from reef structures, especially during the summer and fall recruitment seasons (M. K. Streich, unpublished data). Several previous studies have also reported recruitment of early juvenile Red Snapper to reefs during this time (Rooker et al. 2004; Mudrak and Szedlmayer 2012; Syc and Szedlmayer 2012). Thus, while it is likely that age-0 fish resided within the boundaries of the CCNR, they were likely undersampled following the first fall recruitment season due to both gear avoidance and behavioral exclusion from the immediate reef structures by older Red Snapper.

Changes in Red Snapper seasonal size structure suggest that a majority of the Red Snapper remained at the CCNR after recruiting to the reef. Furthermore, the ability to track distinct cohorts through time can be interpreted as indirect evidence of site fidelity. This inference is supported by previous tagging studies that indicate juvenile Red Snapper exhibit site fidelity to individual structures over multiple months (Holt and Arnold 1982; Workman et al. 2002). Despite the high susceptibility of older Red Snapper (i.e., ≥age 3) to vertical line gear (e.g., Gregalis et al. 2012), there was little evidence of movement of older fish to the CCNR as only four individuals ≥ age 3 were captured during the first year after reef construction. This finding was somewhat unexpected as age frequencies from this study suggested Red Snapper were fully recruited to the vertical line gear by age 2. In addition, Addis et al. (2013) reported that Red Snapper tagged at small concrete artificial reefs in the northeastern GOM displayed a relatively high degree of movement (120 m/d); therefore, if immigration of older individuals was a significant component of the Red Snapper abundance at the CCNR, one would expect these fish to be better represented in the age structure observed here. Although age-3 Red Snapper were rare throughout this study, seasonal age–length frequencies showed they were most represented in late summer 2015. Assuming the degree of site fidelity suggested above, some of these fish would have recruited to CCNR as age-1 individuals during the first fall after reef construction. We also observed increases in Red Snapper mean length, weight, and age with the age of the CCNR, further supporting the hypothesis that most fish remained at the CCNR after arrival. Thus, it appears that a

majority of Red Snapper at CCNR recruited as age-1 individuals (or at age 0, but undersampled) and remained at the CCNR for up to 2 years after reef construction. Similarly, Syc and Szedlmayer (2012), working in the northern GOM, observed increases in mean length, weight, and age of Red Snapper as the age of the artificial reefs they sampled increased and implied that these positive correlations would not occur if at least some fish did not remain at these habitats for several years. Nevertheless, future studies, similar to that of Workman et al. (2002), using acoustic tagging of small individuals could be used to provide direct estimates of the site fidelity suggested in this study.

This study suggests that Red Snapper were in good condition while inhabiting the CCNR. The relatively high condition index ($W_r = 126$) indicates individuals were able to obtain ample prey resources from the CCNR and surrounding mud bottom, which likely translated into fast growth. Supporting the assertion that Red Snapper at the CCNR were healthy and growing quickly, the growth rate observed for Red Snapper in this study (124 mm/year) was faster than previously reported growth rates for fish of similar age off the Texas coast of 60–90 mm/year (Bradley and Bryan 1975), 90 mm/year (Moseley 1966), and 110 mm/year (Holt and Arnold 1982). Although we did not quantify prey abundance at CCNR, artificial reefs can promote increased growth by providing additional reef-associated prey resources or increasing access to or efficiency of obtaining these resources (Bohnsack 1989; Peterson et al. 2003). Because Red Snapper may obtain significant portions of their prey from surrounding mud or sand bottom (McCawley and Cowan 2007; Wells et al. 2008b), other factors such as reef spacing could also have played a role in the availability of prey resources at CCNR. For example, previous studies have observed that species foraging over open bottoms surrounding reefs can create foraging halos of intense prey depletion surrounding the reefs in which they reside, and the degree of prey depletion may become more severe as reef spacing decreases due to increased foraging overlap (Ogden et al. 1973; Lindberg et al. 1990; Frazer and Lindberg 1994). Spacing of artificial structures at the CCNR was variable with some structures as close as 3 m apart, while farther spacing of at least 75 m was also present between "patches" of structure (Figure 1). Therefore, the spatial configuration of structures or the larger overall footprint (11 ha) at CCNR may have prevented severe foraging halos that could have limited Red Snapper growth. Previous studies have investigated the effects of reef density on Red Snapper size and abundance and have generally found increased abundances and larger fish associated with low to intermediate reef densities (e.g., Strelcheck et al. 2005; Froehlich and Kline 2015). Due to the haphazard deployment of structures at the CCNR, the effect of structure density on Red Snapper growth was not assessed here, but such studies would be beneficial in determining optimal configurations for future artificial reef deployments.

The increased habitat complexity associated with the addition of artificial structure at CCNR likely resulted in differences in survivorship of fishes that recruited to the reef site versus those that remained over the bare mud and sand substrates. This inference is supported by several previous studies that demonstrated greater survival of juvenile fish in high complexity habitats (Connell and Jones 1991; Sale 1991; Wells et al. 2008c). Typically, high survival in high complexity habitats is attributed to decreased predation, which can be a significant factor affecting fish densities at reefs (Connell and Jones 1991; Hixon and Beets 1993). For example, Lindberg et al. (2006) demonstrated that Gag Mycteroperca microlepis selected reef shelters at the expense of maximizing growth. The reef pyramids and box culverts deployed at CCNR have numerous holes and crevices, and thus may benefit reef fish by providing refuges from predation. The habitat complexity associated with the CCNR also resulted in the creation of a de facto no-trawl zone. This aspect of habitat alteration should not be overlooked as Red Snapper suffer significant mortality as bycatch in the shrimp trawl fishery in the GOM (Goodyear 1995; Gallaway and Cole 1999). A study by Wells et al. (2008c) observed that early juvenile Red Snapper had truncated size distributions, increased mortality, and decreased growth over trawled habitats than over nontrawled habitats. Accordingly, the elimination of trawling after reef construction reduced the likelihood of trawl-related mortality for juvenile Red Snapper within the confines of the CCNR compared with juveniles residing over the soft bottom where they remained vulnerable to trawling.

Based on the ages of Red Snapper collected in this study and the reported age at 50% maturity of 4.5 years for Red Snapper in the western GOM (Gallaway et al. 2009; Kulaw et al. 2017), a vast majority of individuals inhabiting CCNR were juveniles. An ontogenetic habitat shift has been suggested for Red Snapper, and juveniles can move from low-relief shell rubble habitats to intermediate-relief structures within the first year (Wells et al. 2008c; Gallaway et al. 2009). These juveniles appear to remain at these reefs for about a year at which time (~age-2 fish) they begin to recruit to high-relief structures such as oil and gas platforms, shipwrecks, and natural hard rock outcroppings. Our findings suggest Red Snapper recruited to CCNR as age-1 fish and moved from the reef at around 2–3 years of age. The lack of older fish at the CCNR may be due in part to a combination of emigration and fishing mortality. For example, the oldest fish may be emigrating from CCNR to larger structured habitats as resources at CCNR are reduced with the advent of age-0 and age-1 recruits each summer and fall season. An alternative but not mutually exclusive explanation is that fishing mortality is largely responsible for the disappearance of fish after age 2, especially considering that the CCNR is located near shore and also open to fishing year round. Moreover, the minimum size limit in Texas state waters is 381 mm (15 in)—the upper size range of age-2 fish in our length frequency histograms; thus, it is likely that removal of these size-classes by fishing also plays some role in the lack of older Red Snapper at CCNR. Certainly, future studies should be developed to determine the relative rates of emigration and fishing mortality to

better understand the apparent decline of individuals over age 2.

Collectively, our results suggest that the construction of the CCNR provided several benefits to reef fish that recruited to the reef. It was evident that CCNR provided valuable habitat for juvenile Red Snapper, likely serving a nursery role for the species. Beck et al. (2001) defined a nursery habitat as one that contributes —on a per unit area basis—greater production and export of juveniles to the adult population than production occurring from other habitats where juveniles occur. Greater contribution to the adult population may occur through any combination of increased density, growth, survival of juveniles, and movement to adult habitats (Beck et al. 2001). Much higher densities of juveniles were observed at CCNR than surrounding soft-bottom habitats after reef construction. Despite increased densities of Red Snapper at the CCNR, growth rates appeared to be at least as fast as those previously reported for the species, and condition of individual fish was high. Although we did not directly estimate survival, age-0 and age-1 fish residing at the CCNR likely experienced some protection from shrimp trawl mortality compared with conspecifics of similar age over the open soft bottom that is typical on much of the inner shelf (Parker et al. 1983). Based on the length frequencies observed after reef construction, the aforementioned benefits likely apply to Red Snapper for up to 2 years after they recruit to CCNR. Finally, though it was apparent that recruitment of new individuals to CCNR (i.e., attraction) was an important process acting at the reef site, export (i.e., production) of juveniles to the adult (fishable) population was also evident and likely greater on a per unit area basis than in the surrounding bare habitats. Thus, while our findings indicate that artificial reefs like the CCNR can benefit reef fish such as Red Snapper and possibly Gray Triggerfish, we recommend future studies directly evaluate the relative roles of emigration and fishing mortality in structuring reef fish populations at nearshore artificial reefs in the GOM.

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