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ARTICLE

The Potential for Unreported Artificial Reefs to Serve as Refuges from Fishing Mortality for Reef Fishes

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Abstract

This study tested the potential for 27 artificial reefs, which were deployed in the northeastern Gulf of Mexico (nGOM) but not reported to the public, to serve as refuges from fishing mortality for reef fishes. Red Snapper *Lutjanus campechanus* ($n = 2{,}114$) and Gray Triggerfish *Balistes capriscus* ($n = 267$) were tagged at a subset ($n = 9$) of the study reefs from winter 2005 to fall 2007 to estimate their site fidelity and mortality rates. Mortality was also estimated from catch curves fit to Red Snapper and Gray Triggerfish age distributions that were estimated at the study reefs both before (2005–2007) and after (2007–2008) the GPS coordinates of the above subset were advertised to the public. Red Snapper site fidelity (*f*), estimated by fitting Burnham's (1993) joint encounter model to tagging data, was 12.8% per year; while estimated instantaneous total mortality (*Z*) of Red Snapper ranged from 0.08 per year during the closed recreational fishing seasons to 1.31 per year during the open seasons. Similar estimates were not available for Gray Triggerfish due to inadequate model structure. Prior to reporting of reef coordinates to the public, estimated *Z* from the catch curve analysis was 1.04 per year for Red Snapper and 0.56 per year for Gray Triggerfish. After reporting, estimated *Z* increased from 0.56 to 0.69 per year for Gray Triggerfish and from 1.04 to 1.14 per year for Red Snapper. The low site fidelity and high fishing effort in the region resulted in no discernible refuge effects for Red Snapper. A modest refuge effect occurred for Gray Triggerfish, fish which have been shown to display high site fidelity (>80% per year) to reefs; however, Gray Triggerfish mortality from fishing quickly increased following the reporting of the study reef coordinates. Our study results indicate that unreported artificial reefs that are deployed in areas of the nGOM where fishing is otherwise allowed are not likely to offer protection to reef fishery species.

Globally, few fishery resources are as imperiled as reef fishes, whose populations are affected by a myriad of anthropogenic impacts, including habitat degradation and overfishing (Hoegh-Guldberg et al. 2007; Munday et al. 2008; Paddack et al. 2009). Reef fishes are particularly vulnerable to overfishing because they aggregate on reef habitats of the coastal zone where catch rates tend to be hyperstable, even when spawning stock biomass (SSB) is fished down to unsustainably low levels (Sadovy and Domeier 2005; Erisman et al. 2011). If fishing mortalities of reef fish fall below threshold

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values, the rebuilding of SSB for taxa such as snappers and groupers remains difficult due to the lack of resiliency displayed by these typically long-lived species (Coleman et al. 2000; Sadovy 2005). These global patterns apply to several reef fish species in the northern Gulf of Mexico (nGOM), where fishes such as Red Snapper *Lutjanus campechanus*, Gray Triggerfish *Balistes capriscus*, Gag *Mycteroperca microlepis*, and Greater Amberjack *Seriola dumerili* are overexploited (SEDAR 2011, 2013, 2014a, 2014b). There are challenges to rebuilding these stocks because it is difficult to simultaneously regulate the diverse commercial and recreational fisheries that either target them directly or catch undersized and/or out-of-season fish as bycatch while targeting other species (Porch 2007).

The deployment of artificial reefs is a nontraditional approach to fishery management that has been used throughout the nGOM, ostensibly to enhance the productivity of exploited reef fish stocks. Although high catch rates often follow artificial reef creation (Bohnsack 1989), there is a vast amount of scientific literature that questions whether artificial reefs enhance fish production or simply aggregate their existing biomass (e.g., Grossman et al. 1997; Pickering and Whitmarsh 1997; Cowan et al. 2011). The current understanding is that the likelihood that an artificial reef will enhance the production of a given species is driven by various environmental, behavioral, and life history factors (Bohnsack 1989; Lindberg 1997; Campbell et al. 2011). This complexity presents an optimization problem to fishery managers who seek to maximize the ecosystem services provided by reef fishes (Lindberg and Seaman 2011).

Artificial reefs may serve as net sinks of reef fish biomass if fishing mortality rates are unsustainably high, regardless of whether the reefs actually enhance productivity (Powers et al. 2003; Campbell et al. 2011). A potential solution to this problem would be to deploy artificial reefs either within marine protected areas (MPAs) (Pitcher et al. 2002) or outside of MPAs, without advertising their locations to the public (Lindberg et al. 2006). The Florida Fish and Wildlife Conservation Commission (FWC) adopted the second approach in 2003 when the agency constructed 525 unpublished artificial reefs that were equally divided among four designated artificial reef zones (~225 km^2 each) off northwest Florida. While these reefs were deployed within the existing artificial reef zones, their locations were not reported to the public with the goal that these reefs would serve as refuges from harvest for exploited reef fishes.

The objective of this study was to evaluate whether unreported artificial reef sites deployed by the FWC serve as refuges from fishing mortality for exploited reef fishes. To do this, we conducted a tagging experiment that estimated the site fidelity and mortality of our model species, Red Snapper and Gray Triggerfish, both of which are estimated to be overfished in the nGOM (SEDAR 2013). We also estimated species-specific mortality from size (converted to age) distributions, both before (2005–2007) and after (2007–2008) a subset of study reef locations was reported to the public, in order to examine the effects of "direct" fishing on mortality.

METHODS

Study area.—A subset of study reefs (n = 27; depth range: 27–40 m) was randomly selected from 115 unreported reefs that were deployed by the FWC, in spring 2003, in the Escambia East-Large Area Artificial Reef Site (EE-LAARS), which is located approximately 24–32 km off Pensacola, Florida (Figure 1). Reefs consisted of three different designs (Table 1). The first, an A-type reef, consisted of a single pyramid module with a concrete frame, open base, and steel

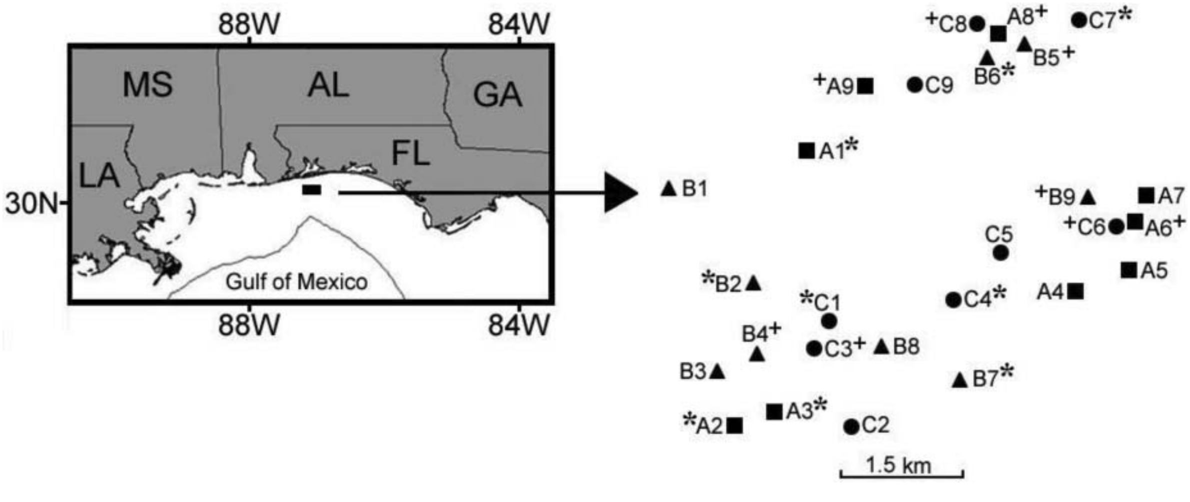
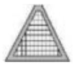


FIGURE 1. Locations of the artificial reef sites (n = 27) within the Escambia East-Large Artificial Reef Site in the nGOM that were included in the study. The shelf edge of the reef is at the 200-m isobaths shown in the inset map. Reef types are as follows: A = pyramid reefs, B = fish haven reefs, and C = reef ball reefs. Asterisks indicate tagging sites, plus signs sites that were reported to the public in spring 2007.

TABLE 1. Dimensions of the artificial reef types deployed in the Escambia East-Large Area Artificial Reef Site that were used in this study.

Reef parameter	Type A: 	Type B: 	Type C: 
Construction material	Concrete/rebar	Concrete	Concrete
Modules per site	1	2	2
Module height (m)	3.05	1.83	1.45
Module base (m)	3.05	3.05	1.83
Reef volume (m^3)	4.09	4.90	2.84

reinforcing bars (rebar), which were set in a lattice configuration along the sides of the reef. The second, a B-type reef, consisted of a pair of concrete modules that shared the same base dimensions as the A-type modules but which were shorter than the A-type reefs, had flattened tops, several triangular openings, and secondary smaller modules that were inserted inside the larger modules. The third, a C-type reef, consisted of a pair of Reef Ball modules which was constructed of concrete, had several wall openings, and an open top. One reef of each type was randomly assigned to one of three treatment groups: control ($n = 9$), tagging ($n = 9$), and experimental ($n = 9$). The coordinates of the study reefs in each treatment group were initially unreported, and the reefs were presumed to be unfished with the exception of the quarterly tagging trips taken by researchers to the tagging reefs. The reefs in the control group remained unreported for the duration of the study to control for the effects of directed fishing on fish mortality, while the reefs in the experimental group were subjected to a period of targeted fishing following the advertisement of their locations to fishers.

Joint encounter model.—The tagging reefs were visited quarterly from winter 2005 to fall 2007, with tagging effort standardized among reefs. Detailed tagging methods are presented in Addis et al. (2013) but are also summarized here. At a given site, we had five anglers target fish with hook and line for 30 min. The hooked fish were brought to the surface at a rate of approximately 1 m/s, and the fish were immediately removed from the hooks and placed into a 475-L holding tank filled with flowing seawater. Fish were removed from the tank, and Gray Triggerfish were measured to the nearest mm in fork length (FL), while Red Snapper were measured to the nearest mm in total length (TL). Fish were tagged with an internal anchor tag, which was inserted into a small incision (<5 mm) in the abdominal cavity, and then they were released. The anchor tags were each marked with the word “REWARD,” an identifying tag number, and a toll free number to report the tag recovery. To encourage fishermen to report their tag recoveries, the tagging study was advertised and included the placement of advertisement posters in the marinas and tackle shops between Bay St. Louis, Mississippi and Panama City Beach, Florida. Those who reported a tag recovery received a \$10 reward per tag and were entered into a \$500 annual lottery of all tag returnees.

Following the encounter history format used in Program MARK (version 7.1; White and Burnham 1999; Cooch and White 2014), tagged fish that were recaptured and released by

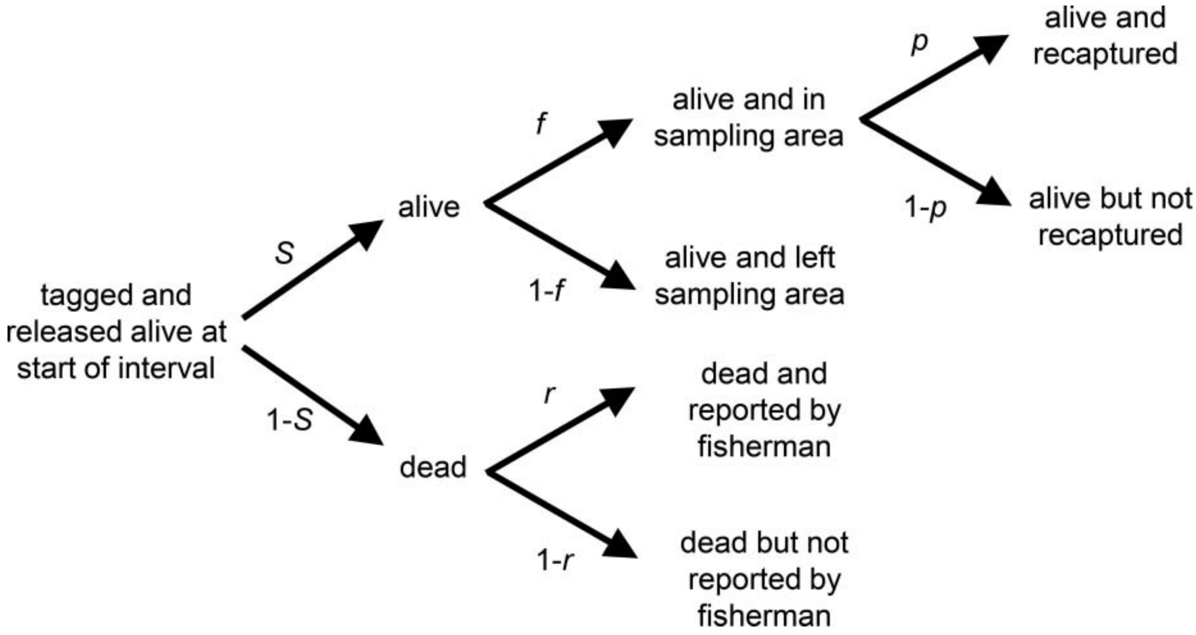


FIGURE 2. The “fate diagram” of the Burnham joint encounter model for a newly tagged individual released on a study reef. The fate of an individual was governed by the following probabilities: S = the probability of surviving the interval, r = the probability of being harvested and reported dead, f = the probability of not permanently emigrating, and p = the probability of recapture, which is conditional on being both alive and within the sampling area.

researchers were expressed as "live encounters," while those harvested and reported by fishers were expressed as "dead encounters." The individual encounter histories for tagged fish were inputted into Burnham's (1993) joint encounter model to estimate interval-specific survival rates. Parameters estimated by the model included survival (S), the probability of recapture (p), the probability of a tagged fish being harvested and reported dead (r), and the probability of permanent emigration of fish to tagging reefs (i.e., site fidelity, f) (Figure 2). Here, S was the annual probability that a tagged fish alive in year i survived until the following year (i+1), taking into account the likelihood of permanent emigration of individuals from the study reefs (Burnham 1993). Random emigration and immigration were both assumed in the joint encounter model because they can affect p by creating a joint probability that an animal is at risk of capture and is captured. The model additionally estimated the probability that a surviving individual in year i+1 would not permanently emigrate (f) from the study area (White and Burnham 1999).

Several plausible models were fit in Program MARK to determine the most parsimonious model. Plausible models included those in which S, p, and f were either held constant or allowed to vary among tagging intervals. The same models were also fit with a hurricane effect as an individual covariate (coded as "1" for those fish at liberty during a hurricane and "0" for fish that were not at liberty) because the center of Hurricane Dennis (maximum winds = 235 km/h) passed within 10 km of the tagging reefs before making landfall in northwest Florida on July 10, 2005 (Addis et al. 2013). The hurricane covariate was considered for all parameters.

Capture–recapture data are frequently overdispersed; therefore, goodness-of-fit tests were computed by applying a parametric bootstrap procedure in Program MARK to the global models of each species (Cooch and White 2014). Simulation results (1,000 runs) permitted the estimation of the overdispersion (variance inflation) factor ($\hat{c}$), which was calculated as the observed deviance estimate divided by the mean simulated deviance (Cooch and White 2014). All models were then adjusted with $\hat{c}$, and model selection was determined using Akaike information criterion that was automatically adjusted by Program MARK for sample size and overdispersion (i.e., the quasi-AIC: QAIC_c; Burnham and Anderson 2002). The highest ranking model had the lowest QAIC_c, and the differences in QAIC_c (ΔQAIC_c) were used to rank and identify other likely models.

Interval specific estimates of S from the most parsimonious joint encounter model were converted to total instantaneous mortality (Z) with $Z = -\ln(S)$. Instantaneous fishing mortality (F) was estimated by subtracting an estimate of instantaneous natural mortality (M) from Z. Red Snapper and Gray Triggerfish M was previously estimated as 0.10 and 0.27 per year, respectively, using the methods of Hewitt and Hoenig (2005) and the maximum observed longevities of these species in the nGOM (SEDAR 2011, 2013).

Estimating mortality for fished and unfished reefs.—Quarterly video sampling of reef fish communities was conducted with a remotely operated vehicle (ROV) at the study reefs (n = 27) from fall 2005 to summer 2008. The point-count method described by Patterson et al. (2009) was used to estimate fish abundance at each reef, and fish size distributions were estimated with a red laser scale that was attached to the ROV (Patterson et al. 2009; Dance et al. 2011). To examine the mortality response of each model species to the fishery, the coordinates of a subset of experimental reefs (n = 9) were advertised to the fishing public in May 2007 via a press release from the FWC's Artificial Reef Program and a posting on the FWC's artificial reef database, which is publicly accessible online.

We used catch-curve analysis to estimate the Z of Red Snapper and Gray Triggerfish at the study reefs, during prereporting (fall 2005 through winter 2007) and postreporting (spring 2007 through summer 2008) of the experimental reef coordinates to the public. Red Snapper size-at-age data, some previously reported by Patterson et al. (2001a; n = 1,755) and some more current (W. F. Patterson III, unpublished data; n = 468), were used to create an age-length key (ALK), using the methods of Ricker (1975). The ALK was then used to estimate ages of laser-scaled fish that were observed in the ROV video samples. Once ages were assigned to individual laser-scaled Red Snapper, age distribution estimates for each sampling quarter and reef- were expanded to total numbers of fish observed at those specific times and sites. The cumulative age distributions were computed by summing the total number of individuals at a given age among all reefs, for both the pre- and postreporting periods. Sizes at age were estimated for the laser-scaled Gray Triggerfish with an ALK that was developed during a 2011 stock assessment for Gray Triggerfish in the GOM (SEDAR 2011). The ALK was derived from Gray Triggerfish in the eastern GOM that were sampled between 2001 and 2010 and aged with dorsal spines (n = 2,565; Johnson and Saloman 1984; Fioramonti 2012). Ages were estimated from bias-corrected FL of laser-scaled Gray Triggerfish, and subsequent cumulative age distributions were computed for both the pre- and postreporting periods as described above for Red Snapper.

The estimated numbers at age were ln-transformed and plotted against age, for both Red Snapper and Gray Triggerfish, at all study reefs during the prereporting period. Catch curves were computed for the ages fully recruited to the study reefs (age 2+): Z was the negative slope from the regression of ln-transformed numbers at age. The catch curves also were fit to the species-specific age distributions that were estimated at the control reefs and experimental reefs during the postreporting period to examine how the reporting of coordinates affected Z. As indicated above, the estimates of F were computed from Z by subtracting species-specific M. Fishing mortality rates that were species specific and conditional (m) were derived from the instantaneous mortality rates, using $m = 1 - e^{-F}$ (Ricker 1975), in order to estimate the percent differences in mortality between the control and experimental reefs.

TABLE 2. Synopsis of the quarterly Red Snapper and Gray Triggerfish tagging efforts at nine artificial reef sites in the nGOM from March 2005 to December 2007. Recaptures were made at the tagging sites during subsequent tagging events. Recoveries are tagged fish that were reported dead by fishermen either during the intervals between the tagging events or within 90 d of the last tagging event.

	Red Snapper			Gray Triggerfish		
Tagging date	*N* Tagged	Recaptures	Recoveries	*N* Tagged	Recaptures	Recoveries
Mar 26, 2005	174		2	66		0
Jun 9, 2005	153	10	12	26	9	2
Sep 27, 2005	314	3	2	15	0	6
Dec 22, 2005	137	1	2	12	0	1
Apr 5, 2006	96	6	22	12	0	0
Jun 8, 2006	123	5	24	12	3	2
Sep 15, 2006	351	3	1	32	2	2
Dec 6, 2006	225	3	7	27	4	1
Apr 19, 2007	64	0	29	7	1	4
Jun 18, 2007	83	4	31	18	5	2
Sep 11, 2007	183	6	3	20	4	1
Dec 8, 2007	211	3	2	20	2	1

RESULTS

Joint encounter model estimates.—A total of 2,107 Red Snapper and 267 Gray Triggerfish were tagged during the study (Table 2). Among the tagged Red Snapper, 44 live fish were recaptured at the release sites on subsequent tagging trips, and 137 dead harvested fish were reported by fishermen (Table 2). Among tagged Gray Triggerfish, 30 live fish were recaptured at the release sites on subsequent tagging trips, and 22 dead harvested fish were reported by fishermen (Table 2).

The Gray Triggerfish joint encounter global model was structurally inadequate to produce parameter estimates due to lack of fit, severe overdispersion ($\hat{c} > 5.00$), poor model convergence, and low effective sample size ($n = 297$) (Anderson et al. 1994; Burnham and Anderson 2002). For Red Snapper, the goodness of fit test for the global model (i.e., time-varying *S, p,* and *f*) was significant ($P < 0.001$), providing evidence that the model failed to represent the data adequately. Therefore, model selection was based on a QAIC$_c$ that adjusted for

TABLE 3. Model fitting summary for the joint encounter models. Models were fit to tag-recapture data for Red Snapper that had been tagged on unreported artificial reefs off northwest Florida, 2005–2007. The best-fitting models were selected using sample size and overdispersion-adjusted QAIC$_c$. The analyses were based on 2,114 tagged Red Snapper and 181 recaptures over 12 quarterly tagging intervals.

Model[a]	Description	QDev[b]	Np[c]	QAIC$_c$	ΔQAIC$_c$[d]
	No hurricane covariate				
$S(t)\ p(.)\ r(.)\ f(.)$	Temporal variation in *S*	58.9	15	590.2	0.0
$S(t)\ p(t)\ r(.)\ f(.)$	Temporal variation in *S* and *p*	47.6	24	597.2	7.0
$S(t)\ p(.)\ r(.)\ f(t)$	Temporal variation in *S* and *f*	50.2	25	601.9	11.7
$S(t)\ p(t)\ r(.)\ f(t)$	Temporal variation in *S, p,* and *f*	44.1	31	608.0	17.8
$S(.)\ p(.)\ r(.)\ f(.)$	All parameters constant	105.7	4	614.8	24.6
	With hurricane covariate				
$S(t)\ p(.)\ r(.)\ f(.)$	Temporal variation in *S*	587.5	15	617.7	27.5
$S(t)\ p(t)\ r(.)\ f(.)$	Temporal variation in *S* and *p*	576.6	24	625.2	35.0
$S(t)\ p(.)\ r(.)\ f(.)$	Temporal variation in *S* and *f*	579.1	25	629.8	39.6
$S(t)\ p(t)\ r(.)\ f(t)$	Temporal variation in *S, p,* and *f*	573.0	31	635.9	45.7
$S(.)\ p(.)\ r(.)\ f(.)$	All parameters constant	638.5	4	646.5	56.3

[a] Model parameters are as follows: S = survival, p = capture probability, r = probability of a tagged fish being harvested and reported dead, and f = site fidelity of tagged fish to tagging reefs; (.) indicates a constant value for a given parameter and among tagging intervals, while (t) indicates interval-specific estimates.

[b] QDev = QDeviance, which is the difference between the –2log(likelihood) of the current model and the –2log(likelihood) of the saturated model, with the saturated model being the model with the number of parameters equal to the sample size.

[c] Np = the number of estimated parameters in the model.

[d] ΔQAIC$_c$ = the difference between the QAIC$_c$ of the current model and the QAIC$_c$ of the model with the lowest QAIC$_c$.

TABLE 4. Joint encounter model estimates of survival (*S*) and its standard error (SE) for Red Snapper tagged at unreported artificial reef sites off northwest Florida. Tagging date = the date at the start of a given interval; the end date of the last interval was March 7, 2008. The % interval within fishing season = the percentage of days between a particular tagging event and the subsequent tagging event that occurred during the recreational Red Snapper fishing season (April 21–October 31 during the study period). Red Snapper total instantaneous mortality (*Z*) per year was calculated from our *S* estimates using the methods of Ricker (1975).

Tagging date	% Interval within fishing season	*S*	SE	*Z* per year
Mar 26, 2005	65.3	0.78	0.25	0.25
Jun 9, 2005	100.0	0.43	0.22	0.84
Sep 27, 2005	39.5	0.87	0.17	0.14
Dec 22, 2005	0.0	0.91	0.11	0.09
Apr 5, 2006	75.0	0.21	0.20	1.55
Jun 8, 2006	100.0	0.27	0.25	1.31
Sep 15, 2006	43.9	0.96	0.08	0.04
Dec 6, 2006	0.0	0.90	0.09	0.11
Apr 19, 2007	96.7	0.29	0.26	1.24
Jun 18, 2007	100.0	0.35	0.29	1.04
Sep 11, 2007	43.2	0.90	0.12	0.11
Dec 8, 2007	0.0	0.92	0.09	0.08

overdispersion and sample size ($\hat{c} = 3.28$, $n = 2{,}158$). The highest ranking model ($\Delta\text{QAIC}_c = 0$) included the constant parameters *p, r*, and *f*, with temporal variation in *S* ($\text{QAIC}_c = 590.2$; Table 3). The addition of a hurricane covariate did not improve the model rank ($\Delta\text{QAIC}_c = 27.6$, Table 3) and was therefore not retained in the final model. Estimates of Red Snapper survival (SE) for each tagging interval ranged from 0.21 (0.11) to 0.96 (0.04) per year; and survival was negatively correlated to the percent of the interval within the recreational fishing season (Pearson's $r = -0.85$, $P < 0.001$). The range in these values reflects open (April 21–October 31) versus closed (November 1–April 20) recreational fishing seasons for Red Snapper (Table 4), with corresponding *Z* estimates ranging 1.55–0.04 per year. Estimates (SE) of the probability of recapture, the probability of a tagged fish being harvested and reported dead, and the site fidelity were 0.02 (0.01), 0.13 (0.05), and 0.13 (0.12) per year, respectively. A site fidelity estimate of 0.13 per year indicated that 87% of the Red Snapper that survived in a given year were projected to emigrate permanently from the study reefs.

Mean *Z* derived from the joint encounter model results was 1.06 per year among those periods ($n = 3$) when the Red Snapper recreational fishing season was 100% open and 0.09 per year among those periods ($n = 3$) when the Red Snapper season was 100% closed (Table 4). With an estimated *M* of 0.10 per year, mean *F* would then be 0.96 per year during the open Red Snapper season and nil during the closed season.

Mortality estimates for fished and unfished reefs.—The catch curves fit to ln-transformed abundance at age at all 27 prereporting study reefs (prior to spring 2007) produced an estimated *Z* of 1.04 per year for Red Snapper and 0.56 per year for Gray Triggerfish (Table 5; Figure 3). The catch curves for the postreporting time period estimated *Z* as 0.99 per year on the control reefs ($n = 9$) and 1.14 per year on the experimental reefs ($n = 9$) for Red Snapper, and 0.54 (control) and 0.69 (experimental) per year for Gray Triggerfish (Table 5; Figure 3). The percent difference in *m* between the control and experimental reefs during the postreporting time period was 9.4% for Red Snapper and 36.6% for Gray Triggerfish, differences which were based on $M = 0.10$ and 0.27 per year, respectively.

DISCUSSION

Overall, our study results demonstrate that unreported artificial reef sites provide some level of refuge from fishing mortality for Gray Triggerfish, while sites offer little refuge for Red Snapper. For Red Snapper, mean *Z* estimated from the tagging data during the open recreational fishing seasons (1.06 per year) was remarkably similar (i.e., within one SE) to the *Z* estimated from the catch curves of all the reefs during the prereporting period (1.04 per year), as well as the *Z* estimated from the control reefs during the postreporting period (0.99

TABLE 5. Catch-curve model results computed from the estimated age distributions of Red Snapper and Gray Triggerfish at both the control ($n = 9$) and experimental ($n = 9$) artificial reefs before and after advertising the locations of the experimental reefs to the public. SE = standard error of the estimated slope, hence Z. *Z* and instantaneous fishing mortality (*F*) are given per year.

Model	*Z* (slope)	SE	*F*
Red Snapper			
All before	1.04	0.08	0.94
Control after	0.99	0.03	0.89
Experimental after	1.14	0.12	1.04
Gray Triggerfish			
All before	0.56	0.06	0.29
Control after	0.54	0.07	0.27
Experimental after	0.69	0.11	0.42

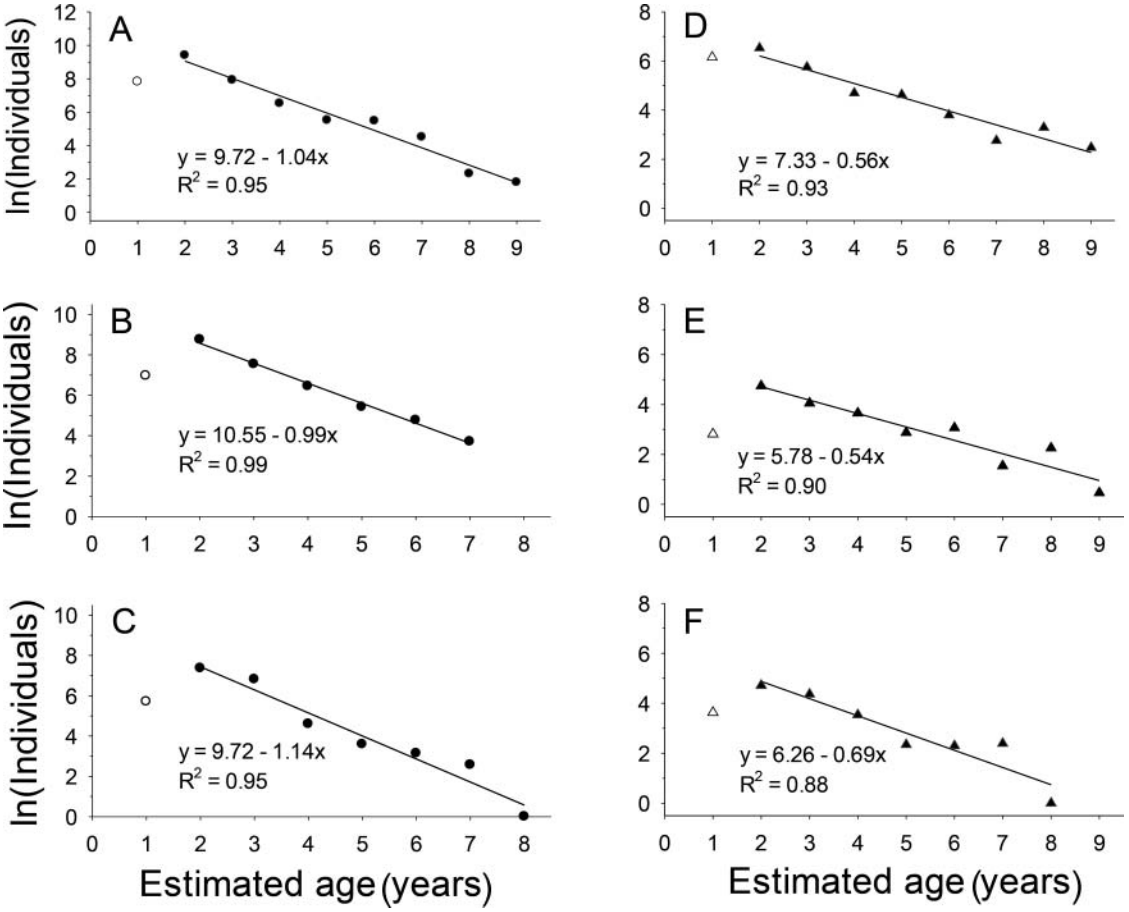


FIGURE 3. Catch curves fit to the estimated age distributions of Red Snapper observed at **(A)** all 27 study reefs from January 2005 to April 2007, **(B)** control reefs ($n = 9$), and **(C)** experimental reefs ($n = 9$) from May 2007 to December 2008. Panels **(D)**–**(F)** are the corresponding catch curves for Gray Triggerfish.

per year). Estimated Z was greater at the experimental reefs than the control reefs during the postreporting period, but F was only marginally higher ($<10\%$) at the experimental reefs following the advertising of their coordinates to the public. Regardless of the treatment group, time period, or estimation method, the estimated F:M was > 8:1 for Red Snapper. That is a remarkably high value given that F at maximum sustainable yield is approximately equal to M for many of the typical bony fishes (Zhou et al. 2012). However, our high fishing mortality rates for Red Snapper are consistent with the estimates from the recreational fishery in the eastern GOM, 2005–2008 (SEDAR 2013). That comparison is important because the recreational fishery generally targets age classes similar to those predominantly observed in this study, while little commercial effort occurs at smaller (<100 m^2) artificial reefs in the system (Porch 2007; SEDAR 2013).

No fishing activity was observed at our study reefs during our quarterly ROV sampling or tagging trips, and no fishing line or tackle was observed in the ROV video samples, both of which can foul the control and/or experimental reefs during prereporting (Dance et al. 2011). This is in stark contrast to our observations at the tagging and experimental reefs during postreporting, where fishing line and tackle conspicuously fouled the reef modules. Therefore, while it cannot be stated definitively that no fishing effort occurred at the control reefs or experimental reefs before spring 2007, there was no obvious evidence that fishing occurred there during that time. If little to no fishing pressure existed at the unreported study reefs, as it is inferred here, then a high regional F alone could not explain the pattern of high F:M observed for Red Snapper. However, low site fidelity implies that Red Snapper move freely among the reefs in this region, thus tagged fish were likely exposed to a substantial amount of regional fishing pressure. Furthermore, low site fidelity and a mean extent of movement of 29–37 km, as previously reported for tagged Red Snapper (Szedlmayer and Shipp 1994; Patterson et al. 2001b; Addis et al. 2013), indicate that our estimates of F for Red Snapper, derived from catch curve analysis, should be thought of as regional rather than local.

Similar to those for Red Snapper, the stock assessment results for Gray Triggerfish indicated that overfishing was occurring in the GOM during our study period (SEDAR 2011). However, higher site fidelity likely translated into a greater refuge effect for this species at the unreported reef sites, as suggested in previous studies (Ingram 2001; Ingram and Patterson 2001). Our sample sizes were too small to estimate Gray Triggerfish site fidelity with the tagging data, but Ingram (2001) estimated Gray Triggerfish site fidelity to be $>80\%$ per year for fish tagged on artificial reefs in the nGOM off Alabama. Furthermore, Addis et al. (2013) reported that mean (SD) movement exhibited by tagged Gray Triggerfish was 8.8 (3.3) km per year, while mean movement of Red Snapper was 37.1 (6.6) km per year. Unreported artificial reefs deployed in heavily fished reef zones may provide some refuge for fish that have high site fidelity (e.g., Gray Triggerfish); however, species that exhibit greater dispersion and low site fidelity (e.g., Red Snapper) will likely be exposed to more fishing pressure. Our results indicate that the F:M for Gray Triggerfish was approximately 1:1 at all reefs (prereporting) and at all control reefs (postreporting); while F:M was 1.6:1 at the experimental reefs (postreporting). When experimental reefs were exposed to increased F from the fishery, we observed more dramatic differences for the Gray Triggerfish between the pre- and postreporting. This contrast may be indicative of how unreported artificial reefs can offer some level of protection for a species that has high site fidelity.

Similar estimates of Red Snapper Z from our tagging versus catch-curve methods corroborate these two approaches. Furthermore, estimates of Z and F:M reported here are consistent with the Red Snapper stock assessment results from the eastern GOM during our study period (SEDAR 2013). The same pattern is true for Gray Triggerfish: F:M estimated at the experimental reefs during the postreporting period (1.6:1) is comparable to the mean F:M estimated for the Gray Triggerfish stock in the GOM, 2005–2008 (1.7:1) (SEDAR 2011). The strongest verification of our methods for estimating mortality is the fact that the closed season Z (i.e., $F = 0$, hence $Z = M$), which was estimated for Red Snapper with the joint encounter tagging model, was nearly identical (0.09 versus 0.10 per year) to the estimate obtained with the Hewitt and Hoenig (2005) method (Porch 2007). Our estimates of M are also consistent with the estimate of 0.11 per year obtained from an acoustic telemetry study of adult Red Snapper in the same region (Topping and Szedlmayer; 2013).

Our study results have important implications for the management of reef fishes in the nGOM and other regions where high fishing mortality exists. Deploying artificial reefs in no-take MPAs can be effective at protecting reef fishes from fishing pressure (Pitcher and Seaman 2000; Pitcher et al. 2002), but MPAs must be sufficiently large to protect spawning stock biomass from fishing effort outside of the MPA boundaries (Allison et al. 1998; Walters et al. 2007). Spillover effects, both of recruits and adults that move beyond MPA boundaries, are important mechanisms for sustaining fisheries (Halpern 2003; Kerwath et al. 2013), but if movement of adults to surrounding fished areas is too great, then MPAs may be ineffective management tools for exploited fishes (Crowder et al. 2000; Walters 2000). Therefore, the ability of unpublished artificial reefs to act as protected areas and enhance the SSB of reef fishes is likely compromised when unpublished reefs are deployed in artificial reef permit zones that are not protected from fishing (such as EE-LAARS). Given the proximity of our study reefs to nearby public artificial reefs, even modest (<5 km) movement could have exposed either Red Snapper or Gray Triggerfish to fishing pressure. Furthermore, the eventual detection of unreported reef locations is likely inevitable due to modern sonar and geopositioning that most fishermen employ to target reef fishes in the nGOM. Therefore, it is currently difficult to develop a scenario for which unreported artificial reef sites could provide a long-term refuge from fishing mortality, even for fishes such as Gray Triggerfish, which display high site fidelity and limited postsettlement movement, unless those reefs are deployed in areas that are totally protected, such as MPAs.

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