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Habitat-specific performance of vertical line gear in the western Gulf of Mexico: A comparison between artificial and natural habitats using a paired video approach

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ABSTRACT

Gear performance is often assumed to be constant over various conditions encountered during sampling; however, this assumption is rarely verified and has the potential to introduce bias. We used fishery-independent vertical line surveys to evaluate whether gear efficiency and selectivity is similar while assessing reef fish populations at oil and gas platforms, artificial reefs, and natural banks in the western Gulf of Mexico. We conducted 192 vertical line sets with cameras placed on a subset of these deployments to validate any differences in efficiency among habitat types. Red snapper (*Lutjanus campechanus*) accounted for 93% of the catch. No difference in red snapper CPUE among habitats was detected. When evaluating fish size, 8/0 and 11/0 hooks sampled significantly larger red snapper at natural banks than at artificial habitats. While CPUE was similar among all hooks at artificial habitats, CPUE at natural banks was lower for shallower hooks and increased towards the bottom hooks along the backbone. At all habitats, red snapper TL decreased from shallow to deep hook positions. Simultaneous camera deployments revealed other processes affecting efficiency such as bait removal and depredation. Vermilion snapper (*Rhomboplites aurorubens*) were effective at removing bait while avoiding capture. Perhaps related to this observation, Red snapper CPUE was negatively correlated with the vermilion snapper video index of abundance. Video confirmed gear saturation was prevalent (70% of deployments), occurring more frequently on artificial habitats. Furthermore, the time fished was effectively “shorter” at artificial habitats as the number of available baited hooks declined rapidly. These results point towards higher relative abundance at artificial habitats; however, the prevalence of saturation indicates vertical line CPUE may not always be proportional to true abundance, hindering our ability to detect differences at the scale examined in this study. Vertical line surveys should evaluate the prevalence of saturation as inferences regarding relative abundance may be compromised when this information is unknown.

1. Introduction

For many exploited fish populations, stock assessments and management commonly rely on fishery-dependent data. However, such data can often be biased by fisher behavior (e.g., targeting of specific portions of the population), management regulations, and gear selectivity (Hilborn and Walters, 1992). Fishery-independent sampling can control for some of these issues and efficiently provide indices of abundance and other biological data over a variety of spatial and temporal scales, which is critical for effective management (Yoccoz et al., 2001). However, fishery-independent methods may still suffer from the same inherent catchability and size selectivity biases because the gears used are often nearly identical to the gear used in the fishery (e.g., longlines,

traps; Ellis and DeMartini, 1995; Harvey et al., 2012; Santana-Garcon et al., 2014). Thus, evaluations of gear performance that can help to identify survey biases are needed, and if possible, should be conducted under a range of environmental conditions and at various habitats that may be encountered during sampling.

Fishery-independent surveys commonly supply indices of abundance that are derived from catch per unit effort (CPUE). The usefulness of these indices relies on the assumption that changes in CPUE reflect proportional changes in actual abundance (Hilborn and Walters, 1992; Quinn and Deriso, 1999). Furthermore, this approach assumes that gear efficiency and selectivity remain constant across space, time, habitat types, and environmental conditions, which is often not the case nor verified (Hilborn and Walters, 1992; McAuley et al., 2007; Rozas and

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Minello, 1997). Certain gears such as longlines may be particularly prone to violating this assumption, because efficiency declines as the number of hooks remaining unoccupied and baited declines during the soak time (Somerton and Kikkawa, 1995). Identifying and accounting for such bias is crucial for estimating the relationship of the survey index with absolute population abundance; however, estimates of absolute population size are often unavailable or expensive to obtain. Nevertheless, with recent advances in remote monitoring (e.g., remote underwater video) and the use of paired gear comparisons, calibration and refinement of surveys designed to index abundance can be achieved (Bacheler et al., 2013a, 2014; Parker et al., 2016; Rodgveller et al., 2011).

In the U.S. Gulf of Mexico (GOM), recent stock assessments for red snapper (*Lutjanus campechanus*) have recommended additional fishery-independent sampling to elucidate regional and sub-regional (e.g., habitat) differences in red snapper demographics (SEDAR, 2013). Accordingly, a vertical line survey was recently developed to characterize the spatial and temporal distribution of commercially and recreationally important reef fish species (Gregalis et al., 2012; SEAMAP, 2013). One particular goal of the survey includes generating an index of abundance for red snapper at both unstructured and structured (i.e., natural hard bottom and artificial structure) habitat types while also providing fishery-independent biological data on size structure, age, growth, and reproduction (Gregalis et al., 2012; SEAMAP, 2013). While this gear is efficient in obtaining such data from patchy reef habitats, there are nuances with selectivity that can influence assessments. For example, Gregalis et al. (2012) evaluated the performance of vertical lines to sample reef fish at artificial (e.g., military tanks and reef pyramids) and unstructured habitats (i.e., bare substrate) off the coast of Alabama. They showed that peak catch rates occurred with five minute soak times and demonstrated the species selectivity of vertical lines by

using a remotely operated vehicle (ROV) survey prior to the vertical line soak. Vertical line hook size selectivity has also been estimated for red snapper and vermilion snapper (*Rhomboplites aurorubens*) at natural habitats in the GOM (Campbell et al., 2014). While these two studies have provided important information on the performance and selectivity of vertical lines, vertical line gear performance among habitat types remains underdeveloped. This is particularly important given that the survey spans natural and artificial habitats – two habitats that can have dramatically different physical characteristics (e.g., vertical relief, habitat area). If vertical lines fish habitats differently, data generated from the survey (i.e., CPUE index of abundance; size structure) may not be comparable across habitats. For example, because the gear fishes vertically in the water column, the efficiency or size selectivity of shallower hooks may be different at natural habitats given their greater distance from the structure.

The goal of this study was to evaluate the performance of vertical lines, following Southeast Area Monitoring and Assessment Program (SEAMAP) specifications, to survey red snapper at three ‘reef’ habitats commonly found over the western GOM shelf. While other studies using vertical line gear have uncovered important data concerning red snapper population dynamics, our aim with this work was to provide information necessary for calibrating vertical line estimates of relative abundance. Given the previous work of Gregalis et al. (2012) and Campbell et al. (2014), we were specifically interested in testing the effects of hook size and hook position on the red snapper vertical line index of abundance (i.e., CPUE) and size among habitat types. Finally, we used simultaneous camera deployments to 1) compare an alternative video-based index of abundance with the vertical line index of abundance, and 2) evaluate other factors such as depredation and gear saturation that may alter vertical line efficiency between habitats.

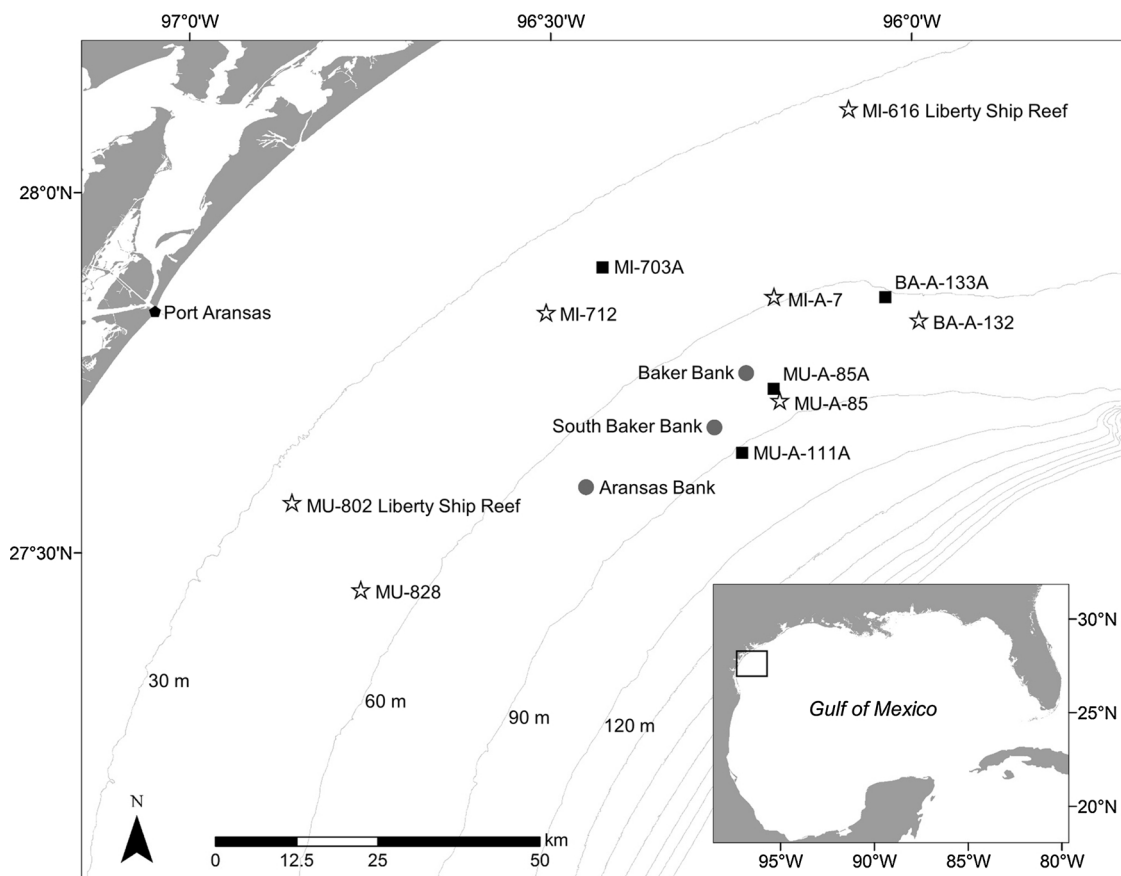


Fig. 1. Locations of artificial reefs (stars), standing platforms (black squares), and natural banks (gray circles) surveyed with vertical longlines from 2012 to 2015 in the western Gulf of Mexico. Gray contour lines indicate relevant bathymetry (30 m isobaths). Inset map displays study area relative to the Gulf of Mexico.

2. Materials and methods

2.1. Study area

Vertical line surveys were conducted at natural banks, standing oil and gas platforms (hereafter “standing platforms”), and artificial reefs off the south Texas coast in the western GOM (Fig. 1). With the exception of the structured habitats sampled in this study, the continental shelf in the region is dominated by open expanses of 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). A persistent nepheloid layer of varying thickness covers the Texas continental shelf, likely influencing the ecology of these habitats (Rezak et al., 1985; Shideler, 1981; Tunnell et al., 2009). Artificial reefs in this study were developed under the Texas Parks and Wildlife Department’s Artificial Reef Program and consisted of decommissioned oil and gas structures (i.e., Rigs-to-Reefs [RTR] structures) or Liberty ships (two sites). Natural banks in this study were part of a group of geomorphic features collectively known as the South Texas Banks (Rezak et al., 1985). Bottom depths at sampling sites ranged from 32 to 90 m. Generally, the natural banks offered less vertical relief (mean = 15 m; range = 12–17 m) than either artificial reefs (mean = 22.8 m; range = 5–53 m) or the standing platforms, which extended from the water’s surface to depth. In addition, the areal extent (i.e., footprint) of artificial reefs and standing platforms surveyed was much less than that of natural banks (~0.001 km² compared to ~0.75 km², respectively). Nevertheless, all of these habitat types are well-known to harbor large populations of red snapper.

2.2. Sampling procedure

Red snapper were sampled with standardized vertical lines from October 2012 through August 2015 during daylight hours (i.e., 10:00–16:00 local time). Vertical line gear followed specifications of SEAMAP (SEAMAP, 2013) and consisted of commercial grade “bandit” reels spooled with 136-kg-test (300 lb) monofilament mainline, which terminated in a 7.3-m backbone (i.e., leader) constructed with 181-kg-test (400 lb) monofilament (Fig. 2). The backbone contained 10 equally-spaced 45-kg-test (100 lb) monofilament gangions (60.9 cm

apart; 45.7 cm long), each terminating with identical circle hooks (Mustad® 39960D; 8/0, 11/0, or 15/0 sizes; same-sized hooks fished on a backbone) baited with cut Atlantic mackerel (*Scomber scombrus*). 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. Therefore, upon arrival at the sampling location, a randomly selected hook size was deployed over the side of the vessel and allowed to soak for 5 min. Hook sizes were then rotated in a random fashion, such that a different hook size was fished on the first, second, and third drop at a site. We conducted three replicate sets at each site visited on a given sampling day. At standing platforms and RTR artificial reefs, each set was conducted around the artificial structure. Because natural banks were considerably larger than artificial structures, a grid with cells the size of the sampling area at artificial sites was overlain onto multibeam imagery of the natural bank in ArcMap 10.3.1 (ESRI, 2015). Grid cells were sequentially numbered, and a single cell was randomly selected for sampling before each sampling trip using a random number generator. Locations for the three vertical line sets were then randomly allocated within the selected grid cell using the ‘Create Random Points’ tool in ArcMap. Upon retrieval of the gear, the fate of each hook was recorded (i.e., fish, bait, no bait, no hook), and captured fishes were identified to species. Species of interest, such as red snapper, were given a temporary tag labeled with a call number and retained on ice for later processing. In the laboratory, fish were measured (SL, FL, TL; mm), weighed (TW; kg), and sexed. Other tissues and hard parts including stomachs, gonads, and sagittal otoliths were also extracted and stored for other studies.

To further evaluate gear efficiency, a video camera (GoPro® Hero3+) was attached to a random subset of vertical line deployments to estimate species frequency of occurrence, relative abundance, and record species interactions with the gear. The camera was attached to the terminal end of the mainline and faced downward towards the backbone (SEAMAP, 2013; Figs. Figure 2B and Figure 3). In the laboratory, video was downloaded from cameras and viewed by two independent viewers. Fish were identified to the lowest possible taxon, enumerated, and recorded each time they entered the field of view. Counts of the two viewers were compared and reviewed only if the counts differed by > 5%. In such cases, disagreements based on

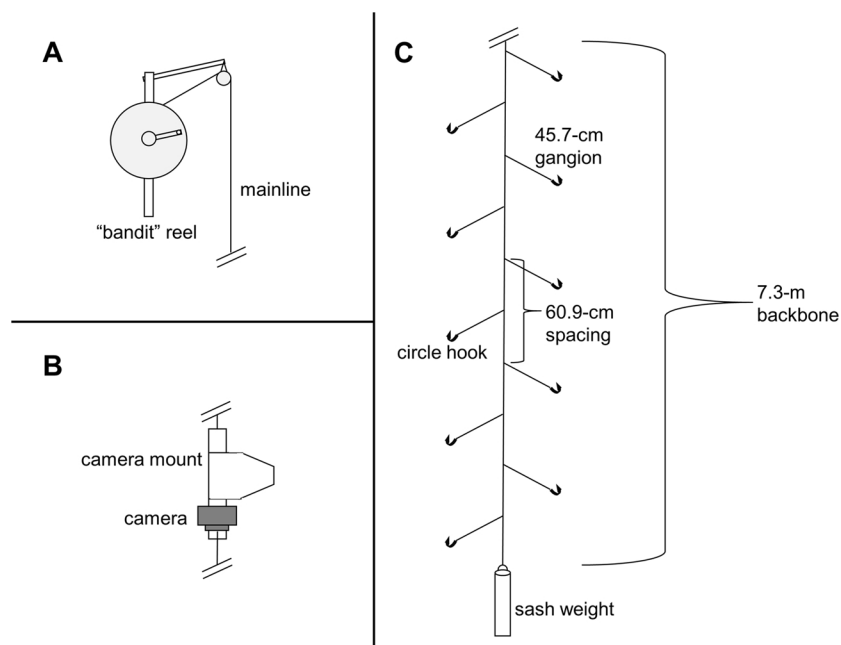


Fig. 2. Schematic of SEAMAP vertical line configuration depicting the (A) “bandit” reel and mainline, which was attached to (B) an optional camera mount with downward-facing camera, and the (C) backbone with 10 equally spaced gangions terminating in 8/0, 11/0, or 15/0 circle hooks.

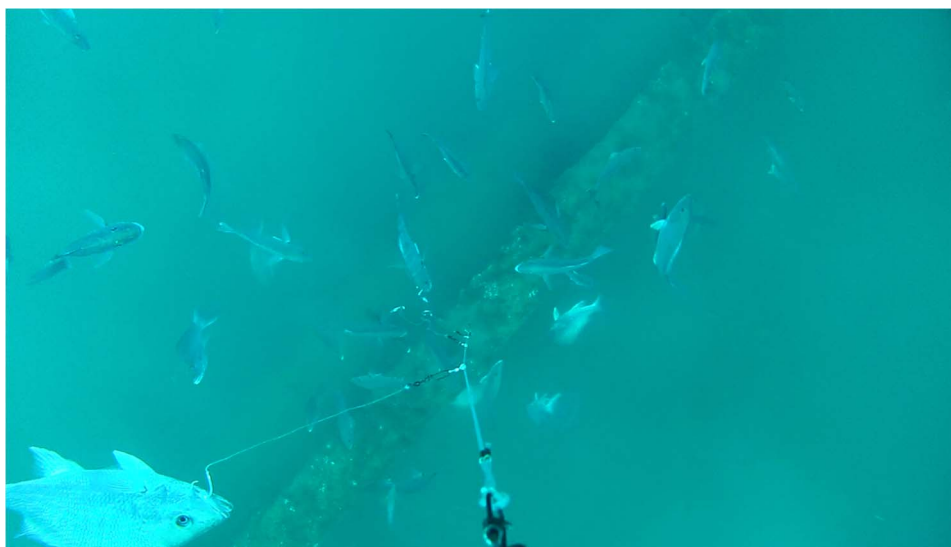


Fig. 3. Screenshot from GoPro® camera during simultaneous vertical line-video deployment. On this particular deployment at an artificial reef, video revealed gear saturation (i.e., no unoccupied, baited hooks remaining; capture probability goes to zero) occurred 17 s into the soak.

counting errors were resolved jointly to reach consensus. If reader counts were > 20 , counts were averaged to save processing time (e.g., reader 1 = 20 and reader 2 = 21, so average of 20.5 used). For each survey, we generated a MinCount for each species that was observed during the five minute soak time. The MinCount, also commonly referred to as MaxN, is a conservative metric that minimizes the probability of double counting. It represents the maximum number of individuals on the screen at any one time during the survey, and its use as an index of abundance is widespread throughout the literature (Ajemian et al., 2015; Campbell et al., 2015; Ellis and DeMartini, 1995; Wells and Cowan, 2007). Video samples were excluded from further analyses if they were unreadable (e.g., too turbid, which we defined as being unable to see at least 3.5 m [> 5 of the 10 hooks on the backbone]). Hereafter, we refer to these paired vertical line and camera deployments as vertical line-video data.

2.3. Data analyses

Catch per unit effort was calculated as the number of fish per hook per five minutes ($\text{fish-hook}^{-1} \cdot 5 \text{ min}^{-1}$). Because the vertical line was deployed multiple times at a site, a linear regression was used to test for an effect of sequential deployments on CPUE at each habitat. To verify cameras had no effect on red snapper CPUE, we compared CPUE from deployments with and without cameras using Welch's t -test. A nested ANOVA was used to test the interactive effects of hook size and habitat type on red snapper CPUE and total length (TL), with hook size (8/0, 11/0, 15/0), habitat (artificial, natural, standing), and their interaction as main effects. To account for variability among sites, site was nested within habitat. All data were assessed for normality and homogeneity of variance using normal probability plots and residual examination, and if necessary, were transformed prior to testing. If significant interactions were detected, post hoc ANOVAs were conducted to test the effect of each factor while holding the level of the other factor constant (e.g., testing hook size effect at natural habitats). Tukey's HSD was used if these post hoc ANOVAs detected significant effects of hook size or habitat. To evaluate the possibility that physical differences in each habitat (e.g., immediate vertical relief) influence the performance of vertical lines, ANCOVA was used to test for a relationship between hook position (i.e., 1–10; shallow to deep) and mean red snapper CPUE or TL by habitat type. If vertical lines fished similarly at each habitat, we predicted that no patterns would exist in red snapper CPUE and TL by hook position (i.e., equal catchability per hook). All tests were carried out using $\alpha = 0.05$.

Using the vertical line-video data, we compared species frequency of occurrence and the red snapper video index of abundance (i.e., MinCount) between habitat types. Welch's t -test was used to test for differences in the mean red snapper video index of abundance between artificial reefs and natural banks. We documented interspecific interactions with the vertical line gear and used a Kendall tau (τ) test to assess association between red snapper CPUE and MinCounts of other species attracted to the vertical line. Because true abundance data were not available, we used the paired vertical line-video data to compare the red snapper video index of abundance (i.e., MinCount) to the vertical line index of abundance (i.e., CPUE). We expected a linear relationship between the two indices if they both indexed true abundance equally well (Bacheler et al., 2013a). To test whether the relationship between the two indices was linear or nonlinear, a linear model ($\log\text{-transformed CPUE} = b \times \log\text{-transformed MinCount}$), a Beverton-Holt model ($\log\text{-transformed CPUE} = [a \times \log\text{-transformed MinCount}] / [b + \log\text{-transformed MinCount}]$), and an exponential model ($\log\text{-transformed CPUE} = a^{\log\text{-transformed MinCount}}$) were fit to the data for artificial reefs and natural banks. Akaike's information criterion (AIC; Akaike, 1973) with the small-sample bias adjustment (AIC_c; Hurvich and Tsai, 1989) was used to determine the best-fitting model for each habitat. The model with the lowest AIC_c indicated the best model; however, models within two AIC_c units of the best model were also considered to have strong support (Burnham and Anderson, 2002). If the best-fitting model was the same for each habitat, a likelihood ratio test was used to determine if the relationship between indices could be described by a single curve (i.e., habitats pooled).

Longlines are prone to the effects of gear saturation (Beverton and Holt, 1957; Somerton and Kikkawa, 1995); therefore, we visually assessed if saturation occurred on each paired vertical line-video deployment of the gear. We considered saturation to occur if all 10 hooks were either occupied by a fish or no longer baited before the vertical line was retrieved (Somerton and Kikkawa, 1995). Fisher's exact test was used to determine if saturation occurred at equal frequencies at the two habitat types. Saturation (by the definition above) effectively reduced capture probability to zero. Thus, time of saturation was recorded and used to estimate the effective time fished for each deployment (i.e., effective time fished = saturation time – deployment start time). Welch's t -test was used to evaluate the null hypothesis of equal time fished between habitats. The time when the MinCount was observed for each deployment was also recorded and compared between habitats using Welch's t -test.

Table 1

Species composition of vertical line catch by habitat type for surveys conducted off the Texas coast from 2012 to 2015. Total catch is sorted in decreasing order of abundance. Bold numbers below column headings denote number of sets (i.e., effort).

Species	Artificial	Standing	Natural	Total catch
Red snapper <i>Lutjanus campechanus</i>	76	59	57	192
Vermilion snapper <i>Rhomboplites aurorubens</i>	904	621	508	2033
Almaco jack <i>Seriola rivoliana</i>	33	13	33	79
Gray triggerfish <i>Balistes capriciscus</i>	5	14	2	21
Greater amberjack <i>Seriola dumerili</i>	15	3	–	18
Blue runner <i>Caranx crysos</i>	4	1	3	8
Warsaw grouper <i>Hyporthodus nigritus</i>	1	–	5	6
African pompano <i>Alectis ciliaris</i>	1	3	–	4
Gray snapper <i>Lutjanus griseus</i>	1	2	–	3
Scamp <i>Mycteroperca phenax</i>	2	–	–	2
Cobia <i>Rachycentron canadum</i>	–	2	–	2
Gag <i>Mycteroperca microlepis</i>	1	–	–	1
Graysby <i>Cephalopholis cruentata</i>	–	1	–	1
Lane snapper <i>Lutjanus synagris</i>	–	1	–	1
Pinfish <i>Lagodon rhomboides</i>	1	–	–	1
Red hind <i>Epinephelus guttatus</i>	–	1	–	1
Rock hind <i>Epinephelus adscensionis</i>	–	1	–	1
Silky shark <i>Carcharhinus falciformis</i>	1	–	–	1
	–	1	–	1

3. Results

3.1. Vertical line sampling

Over the course of the study, we conducted 192 vertical line sets (573 backbones fished) and captured 2184 fish representing 20 species and 7 families. Red snapper comprised the vast majority of the catch (2033 fish; 93.1% of catch; Table 1). Vermilion snapper were the next most commonly captured species (3.6% of catch); none of the remaining species made up more than one percent of the catch. Red snapper ranged in size from 251 to 855 mm TL, averaging 551 mm TL (SE = 6.9; n = 128 backbones with red snapper) at natural banks, 519 mm TL (SE = 5.8; n = 144 backbones with red snapper) at standing platforms, and 517 mm TL (SE = 6.3; n = 202 backbones with red snapper) at artificial reefs. Red snapper CPUE averaged 0.355 fish·hook⁻¹·5 min⁻¹ during the study. There was no evidence of enhanced catch with sequential deployments of the gear at artificial reefs (t = -0.20, df = 225, p = .843), natural banks (t = 0.88, df = 169, p = .380), or standing platforms (t = 0.73, df = 173, p = .466). Cameras were affixed to 166 backbones fished during the study. There was

no evidence that camera presence affected red snapper CPUE (t = 0.59, df = 316, p = .556).

There was no evidence of an interactive effect of habitat and hook size on red snapper CPUE (F_{4,553} = 1.82, p = .123), indicating that the effect of hook size on CPUE was similar among habitat types. While mean CPUE was greater at artificial reefs (LS mean = 0.400 fish·hook⁻¹·5 min⁻¹; SE = 0.04) and standing platforms (LS mean = 0.367 fish·hook⁻¹·5 min⁻¹; SE = 0.05) than natural banks (LS mean = 0.293 fish·hook⁻¹·5 min⁻¹; SE = 0.06), these differences were not significant (F_{2,11} = 1.17, p = .347). Hook size did have an effect on red snapper CPUE (F_{2,553} = 10.58, p < .001), with 11/0 hooks having greater catch rates than either 8/0 or 15/0 hooks. Both habitat and hook size

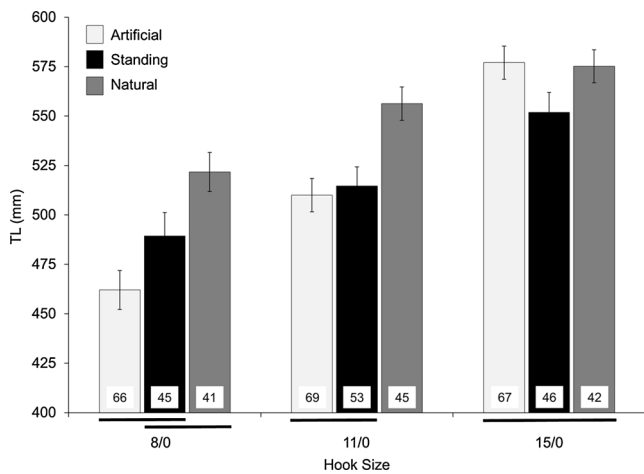


Fig. 4. Mean TL (mm) of red snapper by hook size and habitat type. Within each hook size, means that do not share a black horizontal bar are significantly different (α = 0.05). Error bars represent ± 1 SE and numbers at base of bars indicate sample size for that hook size and habitat combination (n; number of backbones with red snapper).

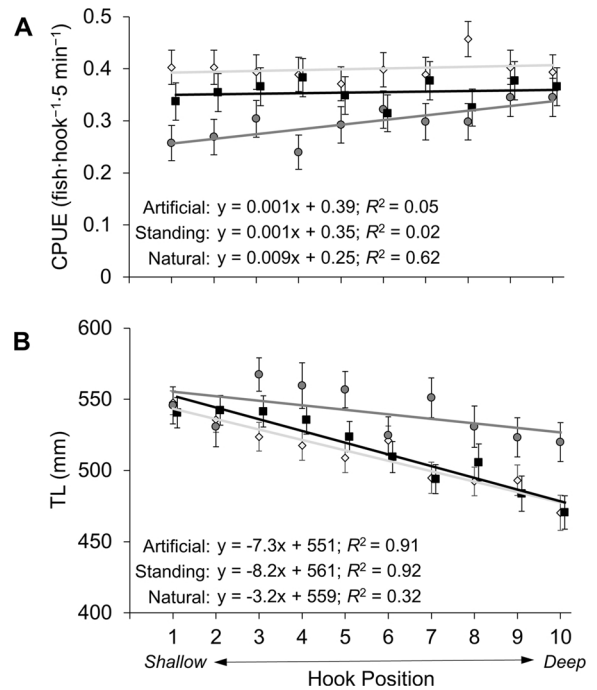


Fig. 5. Plots of (A) mean red snapper CPUE (fish·hook⁻¹·5 min⁻¹) and (B) TL (mm) by hook position for artificial reefs (light diamonds), standing platforms (black squares), and natural banks (gray circles). Hook position along the backbone runs from “1” at the top or shallowest position to “10” at the bottom or deepest position. Error bars represent ± 1 SE.

Table 2

Paired vertical line-video deployment frequency of occurrence (FO) for species either captured on vertical lines or seen on video at artificial reefs and natural banks off the Texas coast, 2012–2015. Species are sorted in order of decreasing video FO. Sample sizes for each habitat indicate the number of paired vertical line-video deployments.

Species	Video FO (FO%)	Vertical line FO (FO%)
Artificial (n = 27)		
Red snapper <i>Lutjanus campechanus</i>	27 (100)	27 (100)
Vermilion snapper <i>Rhomboplites aurorubens</i>	20 (74)	3 (11)
Greater amberjack <i>Seriola dumerili</i>	16 (59)	–
Almaco jack <i>Seriola rivoliana</i>	13 (48)	–
Gray triggerfish <i>Balistes capriscus</i>	9 (33)	1 (4)
Great barracuda <i>Sphyrna barracuda</i>	6 (22)	–
Lookdown <i>Selene vomer</i>	3 (11)	–
Sandbar shark <i>Carcharhinus plumbeus</i>	3 (11)	–
Spotfin hogfish <i>Bodianus pulchellus</i>	3 (11)	–
Blue runner <i>Caranx crysos</i>	2 (7)	–
Crevalle jack <i>Caranx hippos</i>	2 (7)	–
Gray snapper <i>Lutjanus griseus</i>	2 (7)	–
Rainbow runner <i>Elagatis bipinnulata</i>	2 (7)	–
Scamp <i>Mycteroperca phenax</i>	2 (7)	–
Spanish hogfish <i>Bodianus rufus</i>	2 (7)	–
Spinner shark <i>Carcharhinus brevipinna</i>	2 (7)	–
Warsaw grouper <i>Hyporthodus nigritus</i>	2 (7)	–
African pompano <i>Alectis ciliaris</i>	1 (4)	–
Bermuda chub <i>Kyphosus saltatrix</i>	1 (4)	–
Blacktip shark <i>Carcharhinus limbatus</i>	1 (4)	–
Cobia <i>Rachycentron canadum</i>	1 (4)	–
Natural (n = 31)		
Red snapper <i>Lutjanus campechanus</i>	29 (94)	21 (68)
Greater amberjack <i>Seriola dumerili</i>	18 (58)	1 (3)
Vermilion snapper <i>Rhomboplites aurorubens</i>	16 (52)	5 (16)
Sandbar shark <i>Carcharhinus plumbeus</i>	9 (29)	–
Black grouper <i>Mycteroperca bonaci</i>	8 (26)	–
Almaco jack <i>Seriola rivoliana</i>	3 (10)	–
Scamp <i>Mycteroperca phenax</i>	2 (6)	–
Sharksucker <i>Echeneis naucrates</i>	2 (6)	–
Bar jack <i>Caranx ruber</i>	1 (3)	–
Blue angelfish <i>Holacanthus bermudensis</i>	1 (3)	–
Cobia <i>Rachycentron canadum</i>	1 (3)	–
Crevalle jack <i>Caranx hippos</i>	1 (3)	–

influenced red snapper TL, but not in an additive fashion as their interaction was significant ($F_{4,454} = 3.04$, $p = .017$). Post hoc ANOVAs suggested that the effect of habitat was significant for the 8/0 hook ($F_{2,149} = 7.09$, $p = .001$) and 11/0 hook ($F_{2,164} = 6.62$, $p = .002$), but not for the 15/0 hook ($F_{2,152} = 2.07$, $p = .130$). The 8/0 hook sampled significantly smaller red snapper at artificial reefs (mean = 462 mm TL; SE = 9.8; $n = 66$) than natural banks (mean = 522 mm TL; SE = 14.4; $n = 41$), while the 11/0 hook sampled larger individuals at natural banks (mean = 556 mm TL; SE = 9.9; $n = 45$) than either artificial reefs (mean = 510 mm TL; SE = 9.0; $n = 69$) or standing platforms (mean = 515 mm TL; SE = 9.1; $n = 53$; Fig. 4).

Analysis of mean red snapper CPUE by hook position suggested marginal evidence of different efficiencies by habitat type (ANCOVA slopes: $F_{2,24} = 3.05$, $p = .066$). There was no relationship between CPUE and hook position at artificial reefs ($t = 0.63$, $df = 8$, $p = .547$) or standing platforms ($t = 0.37$, $df = 8$, $p = .719$); however, a significant relationship was evident at natural habitat ($R^2 = 0.62$, $t = 3.62$, $df = 8$, $p = .007$), with CPUE increasing from the top to bottom hook position (i.e., shallow to deep along the backbone; Fig. 5A). Red snapper mean TL by hook position also varied by habitat type (ANCOVA slopes: $F_{2,24} = 5.20$, $p = .013$). At all three habitat types, mean TL was greatest at the shallowest hook positions and declined towards the deeper hook positions along the backbone (Fig. 5B). This relationship was not significant at natural habitats ($t = -1.92$, $df = 8$, $p = .091$); however, mean TL significantly decreased from shallow to deep hook positions at artificial reefs ($R^2 = 0.91$, $t = -9.09$, $df = 8$, $p < .001$) and standing platforms ($R^2 = 0.92$, $t = -9.26$, $df = 8$, $p < .001$).

3.2. Paired vertical line-video deployments

Of the 166 paired vertical line-video deployments, 108 (65%) were not useable primarily due to high turbidity. Only five useable vertical line-video deployments were available from standing platforms, and we included these samples in the artificial reef group after determining there was no difference in red snapper mean CPUE or MinCount between these two habitats (Welch's t -test, $p > .05$). Thus, useable vertical line-video deployments were divided fairly evenly between natural ($n = 31$) and artificial habitats ($n = 27$). The use of paired video permitted identification of 21 species on artificial reefs and 12 species at natural banks (Table 2). Of these, only red snapper, vermilion snapper, greater amberjack (*Seriola dumerili*), and gray triggerfish (*Balistes capriscus*) were captured on vertical lines. Video identified 22 additional species that were not captured on vertical lines during paired gear deployments. With the exception of red snapper on artificial reefs, which were seen and captured on all 27 deployments, species frequency of occurrence was greater on video. For example, vermilion snapper were observed on 74% of the vertical line deployments at artificial habitats and 52% of the deployments at natural banks, but were only captured on 11% and 16% of these deployments, respectively. The red snapper video index of abundance was significantly greater on artificial reefs than natural banks ($t = 2.45$, $df = 40.9$, $p = .018$). MinCounts averaged 22.4 (SE = 3.7) at artificial reefs and 11.9 (SE = 2.1) at natural banks. MinCounts as high as 89 on artificial and 52 on natural habitats were recorded.

Several species interactions with the vertical line were observed on video including bait removal, depredation events, and hooked fish

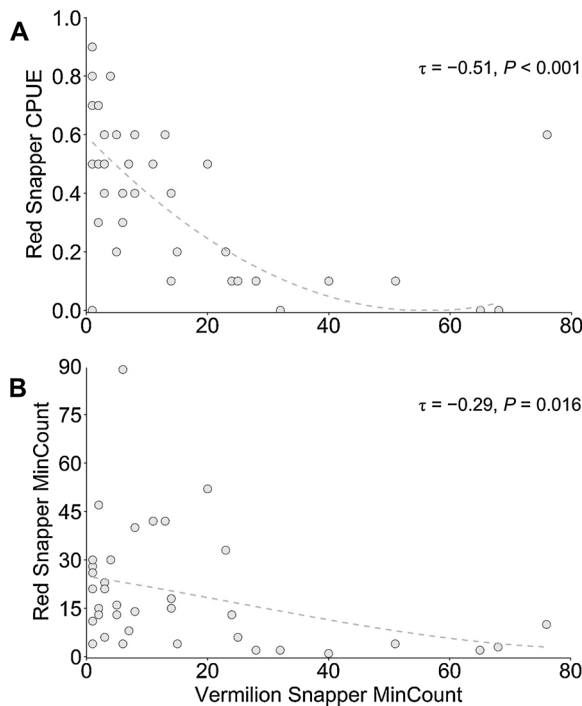


Fig. 6. Scatterplots depicting the negative correlation between (A) Vermilion Snapper MinCount and red snapper CPUE (fish·hook⁻¹·5 min⁻¹) and (B) Vermilion Snapper MinCount and red snapper MinCount. Loess best-fit line depicting negative association between variables is shown in gray.

escaping after initial capture. Video observations suggested that large groups of vermilion snapper (MinCounts up to 76) and gray triggerfish (to a lesser extent due to their lower frequency of occurrence) were effective at removing bait from hooks while avoiding capture. When vermilion snapper were observed on video (n = 36), there was moderate, but significant ($\tau = -0.51, p < .001$), negative correlation between vermilion snapper MinCount and red snapper CPUE (Fig. 6A). There was also a weak negative correlation ($\tau = -0.29, p = .016$) between the vermilion snapper MinCount and red snapper MinCount during these deployments (Fig. 6B). Of all hooks fished during this study (5730 hooks), 41% returned without bait, and no bait was observed falling off the hook during the paired video deployments. Depredation of captured individuals was observed on 19% (11 of 58) of all paired video deployments, with 19 individual depredation events observed (i.e., multiple depredation events occurred during some deployments). Predators of the vertical line catch included great barracuda (*Sphyraena barracuda*; 2 events), greater amberjack (8 events), sandbar shark (*Carcharhinus plumbeus*; 8 events), and Warsaw grouper (*Hyporhamphus nigritus*; 1 event). Depredation of the catch occurred more

Table 3

Models fit to assess relationship of red snapper log-transformed video index of abundance and log-transformed vertical line index of abundance. K = number of estimated parameters; AICc = Akaike's information criterion with small sample bias adjustment; $\Delta AICc$ = AICc difference; w_i = Akaike weight.

Model	Log likelihood	K	AICc	$\Delta AICc$	w_i
Artificial					
Linear	-16.8	2	38.1	0.0	0.62
Beverton-Holt	-16.0	3	39.1	1.0	0.38
Exponential	-22.0	2	48.4	10.3	0.00
Natural					
Linear	-26.6	2	57.7	0.0	0.60
Exponential	-27.6	2	59.7	1.9	0.23
Beverton-Holt	-26.6	3	60.2	2.5	0.18

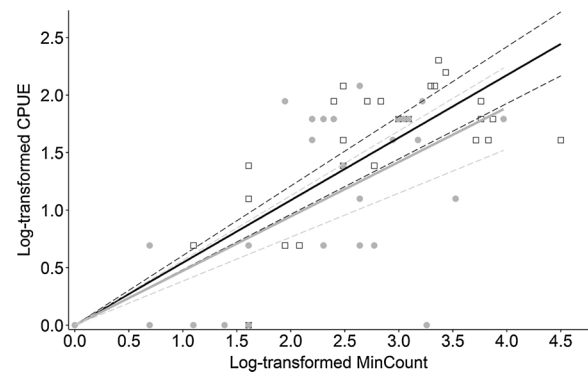


Fig. 7. Log-transformed CPUE (fish 10·hooks⁻¹·5 min⁻¹) versus log-transformed MinCount for red snapper at artificial (black line and open squares) and natural habitats (gray line and circles). The dashed lines for each habitat indicate 95% confidence limits for the slope estimate.

frequently at artificial habitats (33% [9 of 27] of deployments) than natural habitats (6% [2 of 31] of deployments). Greater amberjack most commonly preyed upon vermilion snapper (7 of 8 events), while sandbar sharks exclusively preyed upon red snapper (8 of 8 events). In total, depredation of captured red snapper was observed on 10% (6 of 58) gear deployments and more frequently at artificial habitats (19% [5 of 27] of deployments) than natural habitats (3% [1 of 31] of deployments). Generally, predators removed the entire fish from the gangion. On all 11 deployments where depredation was observed, no fish remains (e.g., head only) were present upon retrieval of the gear, but hooks were missing on 63% (7 of 11) of these deployments. Only four red snapper were observed escaping from hooks after initially being captured, indicating escapement from the vertical line was minimal.

The relationship between the red snapper log-transformed video index of abundance and log-transformed vertical line index of abundance was positive. The linear model best fit the data for artificial reefs ($t = 18.11, df = 26, p < .001$), although evidence for the Beverton-Holt model was strong as well (Table 3). Similarly, a linear model best fit the data for natural banks. While the slope for artificial reefs ($b = 0.54$) was greater than the estimate for natural banks ($b = 0.47$), 95% confidence intervals for these estimates overlapped (Fig. 7). The likelihood ratio test confirmed the slope estimates were not significantly different ($\chi^2 = 1.80, df = 1, p = .179$).

Paired vertical line-video deployments suggested that gear saturation was prevalent at both habitats. In total, vertical lines became saturated on 70% (41 of 58) of the deployments. Saturation occurred as quickly as 15 s at artificial reefs and 18 s at natural banks, and average time to saturation was similar between habitats (1.3 and 1.6 min, respectively). Frequency of gear saturation differed between habitats (Fisher's exact test, $p = .041$), with saturation estimated to occur more often at artificial reefs (85% [23 of 27] of deployments) than natural banks (58% [18 of 31] of deployments). Effective time fished also differed between habitats ($t = -2.5385, df = 56, p = .014$), averaging 1.9 min (SE = 0.33) at artificial reefs and 3.1 min (SE = 0.36) at natural banks. Red snapper MinCounts were observed at similar times into the soak ($t = 0.88, df = 52.7, p = .384$), averaging 2.5 min into the soak at artificial reefs and 2.2 min at natural banks.

4. Discussion

Gear efficiency usually varies among habitat types (Rozas and Minello, 1997; Wells et al., 2008); therefore, comparisons of gear performance among various habitats a survey is likely to encounter are essential for understanding the habitat-specific biases of a gear to make accurate assessments. Knowledge of these biases is necessary for interpreting observed trends and making accurate inferences regarding the population of interest. In this study, we evaluated the performance

of standardized SEAMAP vertical line gear to sample red snapper at artificial reefs, standing platforms, and natural banks through traditional catch-based comparisons and via the use of a simultaneous camera survey. Our results indicate several differences in gear performance that could affect inferences regarding relative abundance and size structure of red snapper inhabiting these habitats. Most notably, the use of simultaneous video comparisons provided insight into several processes affecting vertical lines during the gear soak and revealed that gear saturation was prevalent on most (70%) of the vertical line deployments. High rates of gear saturation can undermine relative abundance estimates by causing hyperstability in vertical line CPUE – especially at high levels of true abundance – resulting in poor estimates of true changes in population abundance. Thus, caution should be applied when using vertical line CPUE to compare relative abundance of red snapper among the habitats sampled in this study and especially with those where fish occur in lower abundance.

Catch-based comparisons suggested no interactive effects of hook size and habitat on red snapper CPUE, suggesting similar catch efficiencies for each hook size regardless of habitat type. In contrast, the effect of habitat type on red snapper size differed depending on the hook size fished. Specifically, mean red snapper TL was significantly smaller at artificial reefs than natural banks for the 8/0 hook and 11/0 hook, but not for the 15/0 hook. Circle hook size does influence size selectivity; however, selectivity curves are broad, and small hooks can sample the largest fish (Campbell et al., 2014; Patterson et al., 2012). This phenomenon was particularly evident for the 8/0 hook, which sampled fish at natural banks that were on average 60 mm larger than fish at artificial reefs. Furthermore, the 8/0 hook sampled larger fish at natural banks than the 11/0 hook sampled at either of the artificial habitats. Given a similar size distribution of fish at two theoretical habitats, one would expect that the same size hook would sample the same size fish. Thus, a plausible explanation for the differences in this study is that natural banks support greater proportions of large fish than artificial habitats in the region (Karnauskas et al., 2017; Streich et al., 2017a), which serves to increase the mean TL sampled by 8/0 and 11/0 hooks.

Our observation of differing trends in mean CPUE and TL by hook position (i.e., height along backbone) suggest differences in gear efficiency and potentially selectivity among habitats. No trends in CPUE by hook position should be evident if each hook fished on a backbone has similar efficiency (i.e., equal CPUE per hook position). This pattern was observed at artificial reefs and standing platforms, but at natural banks, CPUE tended to be lower for the shallower hooks. This implies that these hooks were less efficient at capturing red snapper than hooks near the bottom of the backbone. Consequently, the observed trend at natural habitats, but not at artificial habitats, may invalidate the assumption of equal or constant efficiency among survey conditions (Hilborn and Walters, 1992) and would likely result in underestimation of relative abundance at natural habitats. Understanding why these patterns in catch rates and size exist was beyond the scope of this study, but we hypothesize that differences in the immediate vertical relief at each habitat and relative predation risk may have played a role given that numerous studies have demonstrated the importance of predation and the availability of refuges in structuring fish habitat use (e.g., Hixon and Beets, 1993; Lima and Dill, 1990; Werner et al., 1983). For example, while natural banks in this study had relief up to 17 m, this relief was spread over a much larger area than the relief at artificial reefs. As a result, the immediate relief in the vicinity of a backbone (e.g., 10 m²) fished at a natural bank was quite low (< 2 m) compared to a backbone that was fished at an artificial reef or standing platform (with relief spanning the entire length of backbone). Thus, the lower CPUE at natural banks may be function of fish tending to avoid the more open water where the shallow hooks fished and instead stay toward the structure provided near the bottom. A similar mechanism may explain the increase in TL towards the top of the backbone (i.e., larger fish more likely to forage away from benthic substrate) at all habitats. Regardless,

future investigations into trends in catchability and selectivity by habitat type and depth are warranted.

While previous studies have suggested that red snapper densities are greater at artificial habitats than either bare substrate or natural reef habitats (Patterson et al., 2014; Streich et al., 2017b; Wilson et al., 2006), vertical line CPUE from this study suggested no differences in relative abundance among habitats. Although this could indicate that true densities are not different among the habitats sampled here, the use of paired video indicated that saturation occurred more frequently at artificial habitats. Gear saturation often occurs when the true density of fish is so high that the number of effective hooks (i.e., baited and unoccupied) approaches zero before the gear is retrieved. At such high abundances, CPUE becomes an insensitive indicator of true abundance (Ricker, 1975; Robinson et al., 2015). In addition to a higher frequency of saturation at artificial habitats, the effective time fished was less at artificial habitats, suggesting that effort (5 min) was overestimated more often at artificial habitats. Furthermore, the red snapper video index of abundance was greater at artificial reefs and MinCounts up to 89 were observed around a single backbone. While vertical line saturation also occurred at natural banks, these data suggest that CPUE was underestimated more frequently (and more severely given the differences in time fished) at artificial habitats. As such, our ability to detect differences in relative abundance among habitat types using vertical line CPUE was severely impaired.

The addition of cameras to a subset of vertical line deployments in this study provided a better characterization of the fish community attracted to vertical lines as well as information on species selectivity that would not have been possible based on vertical line catch alone. While only three species were captured on vertical lines at each habitat type during paired vertical line-camera deployments, video data showed that many additional species were attracted to the gear. Frequency of occurrence for all species was higher on video, consistent with previous studies employing paired video techniques (Bacheler et al., 2013a; Harvey et al., 2012). Although red snapper were frequently observed on video and also captured, other species like Vermilion Snapper were captured far less frequently than they were observed – a likely artifact of hook size selectivity (Campbell et al., 2014), gear avoidance, and behavior. In addition, other important species groups such as groupers were never captured during paired vertical line-video deployments despite video confirmation of their presence around the gear. Gears that reduce the frequency of zero catches generally allow for an index of abundance with reduced variability, which is more desirable for stock assessment purposes (Maunder and Punt, 2004). Thus, while vertical lines can generate a useful index of abundance for red snapper (Gregalis et al., 2012), a video-based index, or at least one calibrated with similar video-based methods may be more suitable when determining indices of abundance for these other important species that are often present, but not sampled with vertical line gear.

Other aspects of gear performance such as bait loss and interspecific interactions with the gear would also have gone overlooked without the use of video. The negative correlation between vermilion snapper MinCount and red snapper CPUE is a concern if the goal of the vertical line survey is to generate an index of abundance for red snapper. Hook competition occurs when several species attack the gear such that the CPUE of one species is reduced by the catch of the other (Rodgveller et al., 2008; Rothschild, 1967). While not hook competition per se, given that vermilion snapper were rarely captured, vermilion snapper had a relatively high frequency of occurrence (52–74% depending on habitat), and video documented their effectiveness in removing bait from hooks. Thus, vertical line efficiency may be substantially reduced in areas with high vermilion snapper abundance. The negative correlation between vermilion snapper MinCount and red snapper MinCount suggests that fewer red snapper are present when vermilion snapper are abundant, which could also contribute to lower red snapper CPUE. Nevertheless, the observation of no bait falling off hooks supports the

notion that bait removal by red snapper and other species like vermilion snapper is an important contributor to depressed red snapper CPUE. Depredation of longline catch may also substantially affect CPUE estimates (Ward et al., 2004). In this study, vertical-line video data revealed depredation occurred on nearly 20% of the deployments and showed prey selectivity of some predators (e.g., greater amberjack on vermilion snapper; sandbar shark on red snapper). Sandbar sharks were identified as a primary predator of captured red snapper and usually removed the entire fish from the gangion, suggesting that depredation may be underestimated on deck. Depredation was also likely underestimated on video because of the visibility constraints imposed by the nepheloid layer on some deployments. Importantly, video data suggested that depredation of captured red snapper was more common at artificial reefs (19% of deployments) than natural banks (3% of deployments), providing another potential driver of habitat-specific vertical line efficiency. Collectively, these two gear interactions represent important sources of variability in red snapper CPUE estimates, and we recommend the use of paired vertical line-video deployments to estimate their prevalence.

An index of abundance is commonly assumed to be proportional to true abundance (Maunder and Starr, 2003; Rodgveller et al., 2011). True abundance data were unavailable for our study, and we instead compared vertical line index (i.e., CPUE) with the video index (i.e., MinCount), finding a positive linear relationship for both artificial and natural habitats. Our finding of a linear relationship suggests that both methods may index true abundance equally well and are comparable across the habitats we sampled. It does not necessarily indicate they index true abundance linearly (Bacheler et al., 2013a). In fact, it is likely that both vertical line CPUE and video MinCount index true abundance in a nonlinear fashion, especially given the high prevalence of vertical line saturation observed on video. This inference is also supported by several studies that have demonstrated a non-linear relationship of the MinCount index with true abundance (i.e., MaxN; Campbell et al., 2015; Schobernd et al., 2014). Specifically, the MinCount is hyperstable at high abundances, a trait Campbell et al. (2015) attributed to the inability of the MinCount to account for the increasing number of individuals outside the camera field-of-view as true abundance increases. This was almost certainly the case with our vertical line-video derived MinCount, and we recommend future comparisons of this index with vertical line CPUE at locations that may have lower abundances of red snapper to assess the stability of this relationship.

4.1. Conclusions

The increasing call for fishery-independent monitoring data in modern fisheries management requires the identification of efficient and accurate methods of data collection. Our results indicate that several factors affect vertical line CPUE for red snapper, and these factors may affect the efficiency of vertical lines at artificial and natural habitats differently. While catch data alone did not suggest differences in relative abundance among habitats, several lines of evidence including the higher prevalence of gear saturation and the shorter time to saturation at artificial habitats point to higher relative abundance at these habitats compared to the natural banks in this study. Given the prevalence of gear saturation at both habitats and the linear relationship with the video-based MinCount, as currently deployed, vertical line CPUE in our region is most likely not proportional to true abundance. Thus, while vertical lines are effective in obtaining a large number of red snapper for life-history studies of age, growth, and reproduction (Gregalis et al., 2012; this study), vertical line estimates of red snapper CPUE should be used with caution when attempting to discriminate relative abundance at the scale (i.e., habitat) assessed in our study. Vertical line CPUE may be useful for identifying differences in relative abundance at larger spatial scales or when large gradients or shifts in

true abundance are expected, such as during the colonization of new artificial reefs (Streich et al., 2017c). Gear saturation, especially long before the gear is retrieved (as was the case in our study), results in severely biased CPUE estimates, and many have advocated the use of time-to-capture data rather than CPUE to correct for this problem (Hovgård and Lassen, 2000; Kaimmer, 2004; Somerton and Kikkawa, 1995). Acquiring such data requires information on the fate of each hook over time, which can be obtained using hook timers that activate when a fish strikes the hook and record the time since that strike. While the use of hook timers may be unfeasible over the large-scale SEAMAP vertical line survey, experimental trials could shed additional light on the dynamics of vertical line gear saturation and provide an alternative index of abundance for comparison and calibration (Somerton and Kikkawa, 1995). The frequency of vertical line saturation is likely to increase given the Gulf of Mexico red snapper stock is rebuilding (SEDAR, 2013); therefore, at the very least, paired cameras should be used to assess the prevalence of saturation. Finally, numerous calibration methods are available to standardize catch or CPUE by modeling the effects of explanatory variables that influence gear efficiency (Bacheler et al., 2013b; Maunder and Punt, 2004). These approaches seem suitable for standardizing vertical line CPUE and could employ data generated from the paired camera (e.g., depredation, interspecific MinCount [vermilion snapper]) to enhance the utility of a vertical line index of abundance for assessment purposes.

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