Estimation of Red Snapper Abundance on Oil and Gas platforms in the northern Gulf of Mexico based on Mark-Recapture Methods

by

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Abstract

Estimating abundance of sport and commercial reef fish species remains difficult due to cryptic habitat use patterns. One such species is Red Snapper, *Lutjanus campechanus,* which is perhaps the most important fish species in the Gulf of Mexico. It is known that Red Snapper are an important component of fish assemblages on oil and gas platforms, yet there are few quantitative abundance estimates for this species on platforms. This lack of abundance information for Red Snapper on these structures has led to questions concerning the required removal of platforms after production has ceased, i.e., if platforms are removed will the loss of reef habitat and associated Red Snapper populations cause further reductions on a stock that is already heavily fished. To address this question, the present study applied a Petersen mark-recapture study with Bailey's modification to estimate Red Snapper abundance on platforms in the northern Gulf of Mexico. Abundance estimates were based on the number of tagged Red Snapper available for recapture, the number of recaptures, and the total number caught at the time of recapture efforts. Red Snapper were captured with hook and line (8/0 circle hooks), weighed (nearest 0.1 kg), measured (fork length, standard length, total length mm), tagged with Floy internal anchor tags, and released at 20 different platforms on the continental shelf in the northern Gulf of Mexico (Texas, Louisiana, and Alabama) from February 2017 through October 2018. To allow for full recovery of Red Snapper from tagging effects there was a minimum time period of one week between mark and recapture efforts. The number of tagged Red Snapper available for recapture on each platform was adjusted for emigration, tagging artifacts, natural mortality, fisher non-reporting, and tag retention based on a separate Red Snapper telemetry study on the same study platforms. The adjusted Bailey's mean (±SE) abundance estimate = 1,228 ± 602 with a mean weight (±SE) = 1.63 ± 0.04 kg and ranged from 107 to 10,698 Red

Snapper per platform. The adjusted abundance of Red Snapper by platform were compared among geographical region (continental shelf areas off Alabama, Louisiana, and Texas), depth zone, distance-from-shore, and year. An interaction of region and year had a significant effect on Red Snapper abundance. Depth and distance-from-shore had significant correlations with Red Snapper abundance on platforms, but these results were most likely due to differences in platform size. Mean fishing mortality (F) of Red Snapper on all sampled platforms was low compared to other platform studies with F = 0.19 (0.11 - 0.25; 95 % CL) and ranged from 0 to 0.47 among platforms. Red Snapper tagged on platforms in 2017 had an F = 0.32 (0.23 – 0.44, 95 % CL) and Red Snapper tagged on platforms in 2018 also had an F = 0.32 (0.17 – 0.55, 95 % CL). There were 533 platforms at 18 to 46 m depths in the northern Gulf of Mexico between 2017 - 2018. Thus, based on the adjusted mean Bailey's abundance per platform there were 654,524 Red Snapper with a weight of 1.05 million kg on platforms at these depths during the present study time period. This total abundance and weight estimate indicated that Red Snapper on platforms accounted for 5% of the 2016 Gulf of Mexico Red Snapper biomass. Thus, it is unlikely that the explosive removal of platforms will have a significant effect on Red Snapper stock size in the Gulf of Mexico.

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Introduction

Red Snapper Fishery and Management

Red Snapper, *Lutjanus campechanus*, have significant value to both commercial and recreational fisheries in the Gulf of Mexico. Distinguished by a lifespan of more than 50 years (Szedlmayer and Shipp 1994; Render 1995) and a high fecundity (Collins et al. 2001; Jackson et al. 2006), this gonochoristic, demersal fish occurs on the continental shelf of the Gulf of Mexico and Atlantic coasts (Szedlmayer and Shipp 1994; Render 1995; Wilson and Nieland 2001; Baker et al. 2001; Gallaway et al. 2009).

A comprehensive history of the Gulf of Mexico Red Snapper fishery was reviewed by Goodyear (1995), beginning with the inception of the Red Snapper commercial fishery over 150 years ago off Pensacola, FL and indicating a decline in local stocks as early as 1855 (Stearns 1883; Goodyear 1995). For both the years 1900 and 1965 the total catch of Red Snapper in the Gulf of Mexico reached a peak of around 6,000 metric tons. Following historic high landings in 1965, commercial and recreational catch declined. Commercial landings decreased from 6,389 to 1,015 metric tons (t) between 1965 and 1991, and recreational landings decreased from 4,734 to 581 metric tons (t) between 1979 and 1990 (Goodyear 1995; Schirripa and Legault 1997). In 1984, the first attempt at managing the exploited Red Snapper stock was implemented (Hood et al. 2007; Agar et al. 2014; SEDAR 2018). This was soon followed by various management strategies that divided the total allowable catch (TAC) between the commercial and recreational sectors, increased minimum size limits, set recreational daily bag limits, required permits to limit participation in the fishery, and progressively restricted the fishing season over the next couple decades (Agar et al. 2014). Despite these efforts, Red Snapper were still undergoing overfishing more than 20 years after initial management efforts were in place (Schirripa and Legault 1999;

SEDAR 2005). At the present time of writing, the Gulf of Mexico Red Snapper stock is recovering, and overfishing has not occurred since 2005 (SEDAR 2018).

Management of Oil and Gas Platforms in the Gulf of Mexico and their use as Artificial Reefs

The first offshore oil structure on the Outer Continental Shelf was constructed in 1942, and in the following 55 years more than 5,500 oil and gas platforms (platforms) were installed, with the majority off the Louisiana and Texas coasts (Pulsipher et al. 2001). Federal and international law requires the removal of decommissioned structures within a year of ceased production, and at present the total number of platforms is declining as more are being removed than being built (Pulsipher et al. 2001; Kaiser 2006; Figure 1). At the time of the present study, there were approximately 1,945 platforms in the Gulf of Mexico (K. McCain, LGL Ecological Research Associates, personal communication).

Since the first platforms were built, they have attracted various nekton assemblages by providing reef habitat that was rare on the continental shelf in the Gulf of Mexico (Sonnier et al. 1976; Render 1995; Stanley and Wilson 1997, 1998, 2000; Seaman et al. 1989; Pulsipher et al. 2001; Dufrene 2005). Fish densities can be 20 to 50 times higher near platforms versus in the surrounding open water (Dauterive 2000) and standing platforms have been documented to seasonally serve as critical habitat for a substantial number of commercially and recreationally important fishes, with one previous estimate indicating 1,900 to 28,100 fish species associated with platforms depending on seasonality (Stanley and Wilson 1997).

Explosive removal of platforms is the most common method of removal (Gitschlag et al. 2000). At close range (<230 m), the shock wave from these explosives causes high mortalities of reef fishes associated with platform structure (Continental Shelf Associates 2004). Thus, the removal of platforms results in a loss of recreational and commercial fishing opportunities (Wilson et al. 1987). These concerns were addressed with the establishment of the National Artificial Reef Plan (NARP) in 1985 to help maintain platforms as artificial reefs after production ends (Dauterive 2000). The first platform was converted to an artificial reef in 1979, and since that time artificial reef programs have been perceived as successful. Presently, all coastal states on the Gulf of Mexico manage active artificial reef programs, with platforms and other structures providing enhanced reef habitat (Minton and Heath 1998; Dauterive 2000; Gallaway et al. 2009).

Red Snapper on Platforms

Red Snapper show high natural mortality as age-0 and age-1 juveniles before they enter the directed fishery at around age-2 and are then harvested throughout their lifespan (Gallaway et al. 2009). Juvenile Red Snapper settle over all substrate types, but quickly (within their first year) aggregate on low-relief reef structure and then move to higher relief reef habitat as they grow (Szedlmayer and Lee 2007; Gallaway et al. 2009; Syc and Szedlmayer 2012). Red Snapper have been observed to show high site fidelity (72 - 82 %) and residency (12 to 23 months) to artificial reef habitats (Schroepfer and Szedlmayer 2006; Topping and Szedlmayer 2011; Williams-Grove and Szedlmayer 2016).

Platforms attract high densities of nekton assemblages, and Red Snapper comprise a substantial proportion of these assemblages in the Gulf of Mexico. For example, explosive removal and hydroacoustic surveys both observed that Red Snapper were a dominant component of the platform fish community (Stanley and Wilson 1996; Nieland and Wilson 2003; Gitschlag et al. 2000). Gallaway et al. (2009) calculated a total estimate of 3.1 million Red Snapper at platforms in the western Gulf of Mexico. Similarly, recreational fisher catch rates off Louisiana indicated that Red Snapper were the most common species harvested (Stanley and Wilson 1990), and SCUBA diver visual surveys and video surveys also identified Red Snapper as a dominant species of the fish assemblages around platforms (Sonnier et al. 1976; Bull and Kendall 1994; Render 1995). Most Red Snapper residing at platforms were age 2 to 7 (Render 1995; Wilson and Nieland 2004; Gitschlag et al. 2003; Gallaway et al. 2009).

Petersen Mark-Recapture Methods

Reliable abundance estimates are imperative for managing exploited fish stocks and for evaluating the effects of environmental variables. Mark-recapture studies are common methods for estimating animal abundance. The Petersen mark-recapture method encompasses marking and releasing animals during an initial capture period and after a set time interval recapturing marked animals. This method assumes the proportion of marked individuals in the subsequent sample can accurately estimate the proportion of marked animals in the total population (Petersen 1896; Ricker 1975; Seber 1982).

Petersen mark-recapture studies have been conducted on numerous fish species from a wide range of habitats; for example, Rainbow Trout *Oncorhynchus mykiss* (Rosenberger and Dunham 2005; Gresswell et al. 1997), Cutthroat Trout *Oncorhynchus clarki* (Gresswell et al. 1997), Atlantic Sturgeon *Acipenser oxyrinchus* (Peterson et al. 2000), Coho Salmon *Oncorhynchus kisutch* (Peterson and Cederholm 1984), Sockeye Salmon *Oncorhynchus nerka* (Carlson et al. 1998), and Black Sea Bass *Centropristis striata* (Sedberry et al. 1999). The

Petersen mark-recapture method has also been extended to multiple tagging and recapture efforts (Seber 1982) in Southern Bluefin Tuna *Thunnus maccoyii* (Polacheck et al. 2006).

Despite the high relative abundance of Red Snapper on platforms in the Gulf of Mexico, there are few quantitative abundance estimates and it remains unclear how platforms contribute to stock size. If platforms are important habitat for Red Snapper, the decreasing number of platforms may cause a reduction in stock size (Gitschlag et al.2000). Estimating Red Snapper abundance on platforms remains difficult due to many factors including: density variability across the platform structure, difficulty in sampling the interior of the platform, and changing communities across a wide continental shelf (Stanley and Wilson 1996, 1997, 2000; Sonnier et al. 1976; Gallaway et al. 2009). Traditional methods for estimating fish abundance are difficult, if not impossible, on platforms. For example, trawling cannot sample over the structured habitat, and video surveys can only provide abundance estimates within visible range. Diver (SCUBA) visual estimates are limited by visibility and often high densities that are difficult to count. Hydroacoustic surveys can address many of these difficulties with the exception that it is still difficult to sample the interior of platforms that may contain substantial proportions of fish densities (Stanley and Wilson 1997). Hydroacoustic surveys are also dependent on the extrapolation of fish proportions from diver visual surveys or video recordings, which are again affected by water clarity. To address these difficulties, the present study estimated Red Snapper abundance on platforms in the northern Gulf of Mexico using Bailey's modification of the Petersen estimator (Petersen 1896; Bailey 1951; Ricker 1975; Seber 1982). Importantly, the present study applied estimates of fisher non-reporting, emigration, tag shedding, and tagging artifacts from a separate Red Snapper telemetry study on the same platforms to address the known difficulties in fish mark-recapture studies.

Methods

Study Platforms. ― The present study sampled 21 platforms located from Alabama to Texas in the northern Gulf of Mexico. Stratified random sampling was used to select different platforms for sampling in 2017 and 2018 among 1,945 known platforms (Figure 1). Platforms were stratified based on distance from shore (20 - 60 km, 61 - 100 km, 101 - 140 km) and depth zones (coastal = 10 to 30 m depths, offshore = 31 to 75 m depths; Table 1). Ten platforms were sampled in 2017, and ten new platforms were sampled in 2018 (Figure 2). Platform-2 was sampled in both years in addition to the ten new platforms sampled in 2018. One platform was not sampled in 2017 due to an explosive removal event; another platform was sampled as an alternative. Sampled platforms included five located off Texas, nine off Louisiana, and six off Alabama.

Mark Recapture Methods.― Red snapper were captured with hook-and-line (8/0 circle hooks), anesthetized in a 70-L container with 150mg/L of tricaine methanesulfonate (MS-222) for one minute, then measured (mm, fork length [FL], standard length [SL], and total length [TL]) and weighed (nearest 0.1 kg). Red Snapper were tagged with an internal anchor tag (Floy FM-95W) that was inserted into a small (4 mm) incision made with a No. 11 scalpel in the peritoneal cavity. Betadine was applied over the incision area to reduce infections from tagging. Tagged fish were transferred to a recovery tank and not released until active fin and opercula movement were observed. If water temperatures at the surface exceeded 27°C, ice was added to the recovery tank to reduce high temperature stress. Once recovered, one to six tagged Red Snapper were placed in a remotely opening release cage (85 x 61 x 63 cm; Piraino and Szedlmayer 2014; Williams et al. 2015) made with vinyl coated 16-gauge wire mesh. The release cage was lowered to the seafloor within 20 m of the capture platform, where tagged Red Snapper exited on their own initiative. This cage release method reduced barotrauma and predation mortality and substantially increased survival of tagged fish (Piraino and Szedlmayer 2014; Williams et al. 2015). Cages were retrieved after a minimum of 15 minutes. If a tagged fish was still in the cage after retrieval, the fish was released at the surface. If a tagged Red Snapper died in the release cage, it was removed from the present study. All cage releases were fitted with an attached GoPro4 camera to confirm Red Snapper departure and initial survival.

Platform-30 and platform-9 included multiple tagging and release dates. On platform-30, Red Snapper were tagged and released on 17 February 2017 (n = 33) as described above. Red Snapper tagged and released on 10 March 2017 (n = 31), were tagged with dart-tags inserted into the epaxial muscle without anesthesia and released on a weighted line (Topping and Szedlmayer 2011). GoPro cameras were attached to the drop weight line to monitor fish behavior, environmental conditions, and predation mortalities that may have occurred prior to fish release. Platform-9 included transmitter tagged fish that were part of another study. Red Snapper were tagged with transmitters and internal anchor tags on 3 July 2017 (n = 6) and 4 July 2017 (n = 7; Everett et al. In review). All other Red Snapper at platform-9 were tagged on 18 July 2017 (n = 30) with internal anchor tags. At platform-5.2, Red Snapper (n = 11) were measured, weighed, and tagged without anesthesia, and released at the surface within 15 m of the platform.

A minimum period of one week between tagging and recapture efforts allowed for recovery of tagged Red Snapper. All recapture sampling applied the same hook-and-line methods as described above for tagging. As many Red Snapper were caught as possible during recapture efforts, but recapture efforts were limited to daylight hours. Captured Red Snapper without tags were surface released, while recaptured tagged Red Snapper were weighed (0.1 kg), measured (FL mm), and the fish retained for later age analysis.

In the present study a high monetary reward ($150) was offered for the return of tagged Red Snapper captured by fishers to increase the accuracy of fishing mortality estimates (Topping and Szedlmayer 2013). Identification numbers, contact information, and reward notices were placed on all tags. In addition, reward notices were posted at local marinas, tackle stores, bait shops, and on the Auburn University web page.

Analysis

Abundance Estimation

The Petersen index is perhaps the simplest mark recapture technique to estimate abundance in closed populations, i.e., no births, mortalities, emigration, or immigration (Petersen 1896; Ricker 1975; Seber 1982). The method usually involves one sample period where captured fish are marked and released, and a second sample period where fish are collected and examined for marks. Abundance can then be estimated based on the assumption that the proportion of marked fish that are recaptured out of the total captured accurately estimates the proportion of marked fish in the population. The Petersen abundance equation is:

$$N = \frac{MC}{R}$$

where M is the number of fish initially marked, C is the total number of fish captured and examined for tags in the recapture effort, R is the number of marked fish captured in the recapture effort, and N is the abundance estimate at the time of marking.

The abundance of Red Snapper was estimated with the Bailey's modification of the Peterson mark-recapture method at each platform (Bailey 1951). Bailey's modification provides an unbiased abundance estimate when recaptures of marked fish are greater than 3 (Ricker 1975). Although it is noted that the Petersen formula remains a consistent estimation of N and tends to be increasingly accurate as sample sizes are increased, both Bailey (1951) and Chapman (1951) note that this formula tends to overestimate the true population size in cases of direct sampling. Thus, Bailey's modification (Bailey 1951) accounted for this Petersen overestimation with the following:

$$N = \frac{M(C+1)}{R+1}$$

Platform-9 had both transmitter-tagged fish and Floy-tagged Red Snapper. Transmitter-tagged fish that were caught, tagged, and released prior to recapture efforts (n = 13) were included in the abundance estimates at this platform. Platform-30 and platform-9 also had multiple tagging and recapture efforts, and these were combined and analyzed as one tagging and recapture effort for each platform.

As stated previously, two tagging methods were used at platform-30: either epaxial muscle tagging with drop weight releases or internal anchor tagging in the peritoneal cavity with cage releases. At this platform, one Red Snapper was recaptured from each method. To determine if recapture rates differed between the two methods, the number of returns by fishers with anchor tags (n = 15) was compared to the number of returns with dart tags (n = 13). The comparison did not detect a significant difference in return rates between the two tagging methods (X^2 = 0.14, df = 1, P = 0.705), thus tagging methods were pooled. Weights for back tagged fish were computed from a weight to length relation (SEDAR 31):

$$\text{WW (lbs)} = 4.47\text{E-}04 * \text{Max TL (in)}^{2.994}$$

The number of Red Snapper available for recapture (M_R) was adjusted for emigration rates, tagging artifact mortality, fisher non-reporting, and tag retention rates from a concurrent Red Snapper telemetry study on platforms (Everett et al. In review), and fisher removal of tagged Red Snapper for the time between marking and recapture efforts. For all platforms, upper and lower 95% confidence intervals were calculated by treating recaptures as a Poisson variable and obtaining direct upper and lower limits from the chart provided by Ricker (1975) and substituting these values into the original equation (Avyle and Hayward 1999).

Abundance estimates of Red Snapper on platforms were compared with a generalized linear model (PROC GLIMMIX, SAS version 9.4) with negative binomial distributions and log link functions among regions (continental shelf areas off Alabama, Louisiana, and Texas), years, depth zones, and distance-from-shore (Huelsenbeck and Crandall 1997; Seavy et al. 2005; Bolker et al. 2009). Year and region were considered fixed factors, while depth zones and distance-from-shore were considered covariates. Each covariate was tested for two-way interactions and stepwise simplification was performed based on the corrected Akaike information criterion value. If a significant interaction on abundance was detected, regression analysis was used to show relations of covariates on abundance. The most appropriate model was determined by comparing all combinations of these variables and selecting the model with the lowest corrected Akaike information criterion value and detection of significant effects. The reduced model included year and region as fixed factors in the generalized linear model, and separate linear regression analyses for depth and distance-from-shore.

A closure test was used to evaluate the probability of a closed population (i.e., no immigration or emigration) for the sampled platforms, with recapture histories adjusted for tagging artifacts, tag retention, emigration, fishing mortality, and natural mortality (Stanley and Burnham 1999). Platforms with less than 10 tagged Red Snapper were not included in analysis due to small sample size.

Tag Retention, Emigration, and Tagging Artifacts

A tag retention rate of 2.67 % for 87 days was applied in the present study based on combined data on tag retention from previous telemetry studies (Topping and Szedlmayer 2011, Williams – Grove and Szedlmayer 2016; Everett et al. In review). Emigration, tagging artifacts, and fisher non-reporting rates were obtained from a telemetry study of Red Snapper on the same platforms over the same time period as the present study (Everett et al. In Review) and applied to abundance estimates in the present study. Emigration, tagging artifact, tag retention, and fisher non-reporting rates were applied based on the time interval between initial tagging and recapture dates for each platform (7 - 87 days, Table 1). A 100% fisher return rate was assumed for the present study based on the comparison of fisher return rates to fishery independent mortalities determined from telemetry (Everett et al. In review).

A known fate model in the MARK program was used to estimate annual fishing mortality (F) for each platform selected for mark-recapture abundance estimates. A staggered entry start date and conditional probabilities were used in instances where individuals “entered” at different times during the study period. Annual estimates for each platform were based on monthly time intervals beginning with the initial tagging month and ending one year from the initial tagging month. Only fishing mortalities that occurred within one year of initial tagging were included. For fish tagged on platforms in 2017, each 2017 platform had a unique 12-month history interval. This same procedure was used to calculate fishing mortality of fish tagged on platforms in 2018. All fishing mortality estimates were adjusted for tag loss, tagging mortality, natural mortality, and fisher non-reporting. Fishing mortality was calculated with the following equation:

$$F = -\ln(S)$$

Where survival is based on the probability of surviving *F* over a year. Fishing mortality was compared with a generalized linear model (PROC GLIMMIX, SAS version 9.4) among regions, distances-from-shore, depth zones, and years (Huelsenbeck and Crandall 1997; Seavy et al. 2005; Bolker et al. 2009). As described above for abundance comparisons, the reduced model was based on the lowest corrected Akaike information criterion value for *F* comparisons and included year and region as fixed factors, and separate linear regression analyses for depth and distance-from-shore.

Cage Release Efficiency

Cage release efficiency was determined by comparing the number of successful releases to unsuccessful releases as determined from video recordings of tagged fish releases. A successful release was defined as an event where the cage door opened completely, and fish exited the cage. Instances where it could not be determined if the cage door opened due to low visibility conditions were not included in the analysis.

Results

Red snapper (n = 786) were tagged and released at 20 different platforms in the northern Gulf of Mexico from February 2017 to October 2018 (Figure 2). Among the 21 platforms where Red Snapper were tagged and released, recaptures were obtained at 12 platforms. Platforms with less than 10 tagged Red Snapper were removed from analysis (n = 4, Table 1). Recapture efforts were completed between 7 to 87 days after tagging (Table 1). Red Snapper mean (± SE) abundance = 1,228 ± 602 (0 – 2,504, 95% CL) and ranged from 107 to 10,698 fish per platform (Table 2). Platform-4 had the highest abundance with 10,698 Red Snapper and platform-67 had the lowest abundance with 107 Red Snapper (Table 2).

Red Snapper abundance (±SE) in 2017 (least square mean =1,508 ± 387) was significantly higher than in 2018 (least square mean = 430 ± 97; $F_{1,11}$ = 13.5, P = 0.004). Region did not have a significant effect on Red Snapper abundance (±SE) on platforms (Texas least square mean = 1,295 ± 329, Alabama = 652 ± 249, and Louisiana = 617 ± 140; $F_{2,11}$ = 2.6, P = 0.12). However, there was a significant interaction detected between year and region, with Red Snapper abundance on platforms significantly higher in 2017 off Texas platforms (4,199 ± 1,507) compared to 2018 off Alabama platforms (234 ± 109), 2017 off Louisiana platforms (449 ± 125), and off Texas platforms the following year (399 ± 144; $F_{2,11}$ = 21.8, P = 0.002; Figure 3).

There was a significant increase in Red Snapper abundance on platforms as depth increased (R^2 = 0.24, $F_{1,15}$ = 4.6, P = 0.048, Figure 4). Similarly, there was a significant increase in abundance as distance-from-shore increased (R^2 = 0.36, $F_{1,15}$ = 8.3, P = 0.011, Figure 5).

Modified recapture histories used in the program CloseTest (Stanely and Burnham 1999) indicated violation of closure at most (85%) sampled platforms ($P \leq 0.05$) due to evidence of additions to the population. This test is based on the “NR vs JS” component test, which tests the fit of the no recruitment model against the Jolly Seber open population model. Platforms-5.2, 31, and 50 did not show evidence of additions or losses to the population during the sample period.

Fishers reported 141 captures of tagged Red Snapper out of 786 that were released in the present study over the study time period (Table 3). Fishing mortalities varied among platforms and F ranged from 0 to 0.47 (Table 3). No tagged Red Snapper were caught at platforms-9, 18, 19, 57, and 58 (i.e., F = 0), while platform-30 and platform-66 had the highest fishing mortality with F = 0.47 (Table 3).

Fishing mortality on platforms was not significantly affected by region ($F_{2,11}$ = 2.68, P = 0.11), year ($F_{2,11}$ = 0.24, P = 0.63) or an interaction of region and year ($F_{2,11}$ = 2.19, P = 0.16, Figure 6). There also was not a significant relation between depth and fishing mortality (R^2 = 0.12, P = 0.17, Figure 7), or distance-from-shore and fishing mortality (R^2 = 0.01, P = 0.73, Figure 8).

Due a lack of significant effects based on region and year, data were pooled among all platforms for all regions and years and fishing mortalities calculated. Annual survival (S) among all platforms was 0.74 (0.65 - 0.81; 95% CL) with F = 0.31 (0.21 - 0.42; 95% CL). Fish that were tagged in 2017 had an annual S = 0.72 (0.64 - 0.79; 95% CL) and F = 0.32 (0.23 - 0.44; 95% CL). Fish that were tagged in 2018 had an annual S = 0.73 (0.58 - 0.84; 95% CL) and F = 0.32 (0.17 - 0.55; 95% CL).

Platforms off Alabama had the highest fishing mortality, with an annual S = 0.68 (0.58 - 0.77; 95% CL) and F = 0.38 (0.26 - 0.55; 95% CL), followed by Louisiana platforms with an annual S = 0.76 (0.70 - 0.81; 95% CL) and F = 0.28 (0.21 - 0.36; 95% CL). Texas platforms had the lowest fishing mortality with an annual S = 0.85 (0.79 - 0.89; 95% CL) and F = 0.16 (0.11 - 0.23; 95% CL).

Cage release methods used in the present study have been shown to substantially increase fish survival after release by reducing barotrauma and predation effects (Piraino and Szedlmayer 2014, Williams et al. 2015). However, the cage efficiency rate (the percentage of fish not released because they failed to exit the cage) has not yet been previously reported. Among the 240 recorded cage release attempts in the present study, 19 fish failed to exit the cage and 5 attempts were undetermined, indicating a cage release efficiency rate of 91.9%.

Discussion

The Petersen method

The Petersen estimator remains a simple way to estimate population abundance for mark-recapture studies. Bailey's modification of the Petersen index gives more accurate estimation of abundance when the number of recaptures is small (Bailey 1951). Avoiding the limitations of the Petersen method is frequently difficult and often violations of assumptions occur that may result in significant biases (Seber 1982). However, the present study addressed many of these biases. Although mark recapture studies are inherently difficult for species that are widely distributed and challenging to directly observe, Red Snapper may be the most ideal fish species to sample with mark-recapture due to their high (70%) probability of presence on platforms after one year (Everett et al. In review).

Validation of Abundance Estimates

In order to obtain valid abundance estimates with the Bailey's modification of the Petersen mark-recapture method, several assumptions must be met: 1) sampling must occur in a closed population, i.e., the population size does not change over the time period of the study; 2) tagged and untagged fish have an equal probability of being captured in the recapture effort; 3) tagged fish do not lose their tag prior to the recapture effort; 4) tagged fish are not missed in the recapture sample; and 5) tagged fish become randomly mixed with untagged fish prior to the recapture effort. Any violation of these assumptions will result in underestimating or overestimating the true population size. An advantage of the present study was that most of these violations were minimized due to the short time periods between mark and recapture and that all abundance estimates were based on captures by the present study (i.e., fishery independent).

Despite evidence of a violation of closure for sampled sites, bias was minimized by accounting for losses of tagged Red Snapper between tagging and recapture efforts due to tag-retention, emigration, and tagging artifacts based on concurrent and previous telemetry studies of Red Snapper (Topping and Szedlmayer 2011; Piraino and Szedlmayer 2014; Williams-Grove and Szedlmayer 2016; Everett et al. In review). Longer time periods between initial marking and recapture efforts (up to 87 days) may have contributed to violations of closure attributed to immigrations to sampled platforms that were not recognized, though these were likely minimal due to overall low rates of emigration between mark and recapture efforts.

There was no significant difference detected in recaptures of Red Snapper with dart tags in the epaxial muscle that were released with drop weights compared to anchor tags in the peritoneal cavity with cage release at platform-30. Phelps and Rodriguez (2011) report 90.4% retention rate of internal anchor tags but mention that inflammation and hemorrhage commonly occurred at the site where internal anchor tags were inserted into the body. Additionally, the short time period of the present study (7-87 days) minimized bias due to tag loss events, ensuring that all marked fish were likely accounted for in the recapture sample and very few Red Snapper lost their tags prior to recapture events.

Before recapture efforts, tagged Red Snapper were provided a minimum of 7 days to recover from tagging effects and to allow full mixing of tagged Red Snapper with untagged Red Snapper. However, most (71 %) platforms had greater than 30 days between tagging and recapture efforts. Effects of handling and tagging fish has the potential to alter fish behavior after release (Mesa and Schreck 1989), which could reduce their catchability if not allotted enough time to fully recover before recapture efforts. A 7-day period before recapture efforts was supported by previous studies that indicated a 24 - 48-hour recovery period was adequate for full recovery after tagging (Mesa and Schreck 1989; Peterson et al. 2004).

The 100% fisher reporting rate was higher than previous Red Snapper telemetry studies on fisher reporting rates (> 62 %) for captured fish (Topping and Szedlmayer 2013, Williams-Grove and Szedlmayer 2016), but based on the most recent telemetry study of Red Snapper on the same platforms as the present study the 100% fisher reporting rate was applied (Everett et al. In review).

There is evidence that the growth rates of tagged fish are similar to those observed in wild Red Snapper in the Gulf of Mexico (Szedlmayer and Shipp 1994; Patterson et al. 2001). Phelps and Rodriguez (2011) report that Red Snapper survival did not differ between tagged and untagged fish (independent of tag type) and that weight gain among tagged and untagged fish were similar after the tagging process. Gitschlag and Renaud (1994) report that the observed condition of released fish at the surface correlated with mortality estimates from cage tagging experiments and reported that most Red Snapper mortalities occurred shortly after capture. The use of predator protection cages in the present study allowed fish to exit the cage on their own initiative once they had recovered from the tagging process, thus lowering predation rates when tagged fish were most vulnerable just after release (Piraino and Szedlmayer 2014).

The present study was dependent on the successful recapture of tagged Red Snapper. There is some evidence that Red Snapper catchability may differ among individuals and fish initially tagged and released may have a higher probability of recapture during recapture efforts. For example, several Red Snapper tagged and released off coastal Alabama were recaptured more than two times, and one Red Snapper that was captured and released in August 2006 was recaptured three times over the following two years (Szedlmayer, unpublished data). Similarly, a previous study reported 42 Red Snapper had multiple recaptures among 2,932 total tagged individuals off coastal Alabama (Patterson et al. 2001). If catchability varies among individual Red Snapper, fish that were not captured during tagging efforts may have a lower catchability compared to tagged fish and would result in an underestimate of Red Snapper abundance.

Variation in Abundance Estimates

In the present study the adjusted mean abundance of Red Snapper on platforms was lower than a previous estimate of Red Snapper abundance on platforms but was within the calculated range (mean = 1,884, range 905 – 4,632, Cowan and Rose 2016). Stanley and Wilson (2000) estimated Red Snapper numbers at two petroleum platforms sampled off Louisiana were 1,264 (depth = 60 m) and 2,642 (depth = 20 m) fish based on ROV point count surveys and dual-beam hydroacoustics. However, more platforms were sampled in the present study and were in a variety of depth zones (29 - 42 m). Gitschlag et al. (2000) estimated mean Red Snapper mortality (assuming mortality = abundance) on nine platforms on the Louisiana and Texas coasts in depths of 14-32 m to be 515 Red Snapper (range 24-1,193) between the years 1993-1999 using explosive removals. In that study all platforms were sampled from May to September. The lower abundance of Red Snapper at the platforms studied by Gitschlag et al. (2000) compared to the mean abundance estimate in the present study may be attributed to low visibility conditions and predation of dead fish that were not collected by divers immediately after the explosive removal events. Shark predation is highly intense immediately following an explosive removal (Osowski and Mudrak, personal observation, 2017) and the collection of all affected fish after such a removal before there are extreme losses due to predation is highly unlikely. Loss of dead fish could result in large underestimates of Red Snapper mortalities which would explain differences in the mean Red Snapper abundance estimate in the present study and the estimated mean Red Snapper mortality. Stanley and Wilson (1997) reported platform WC 352 to have a mean Red Snapper abundance of 3,090 Red Snapper based on monthly hydroacoustic and visual point count surveys between January 1991 - May 1992. However, Red Snapper abundance varied substantially over the sampling period (521 - 8,202) and sampling only occurred on one platform off the coast of Louisiana in 22 m of water. It is also likely that visual point count surveys conducted by ROV's and scuba divers may have been overwhelmed by the large fish assemblages observed at platform WC 352 to accurately estimate fish species proportions to calculate fish abundances accurately. The present study sampled more platforms in different regions at a variety of depths to obtain a more accurate Red Snapper abundance estimate that can be applied to the entire northern Gulf of Mexico region.

Wilson and Neiland (2004) estimated Red Snapper abundance at depths of 20 - 100 m on platforms was 1.2 - 7.2 million individuals based on both hydroacoustic and video surveys from the years 1994, 1996 - 1998, and 2000, and platform explosive removals conducted in 1998. Similarly, another study estimated a total of 3.1 million Red Snapper on a total of 4,000 platforms present in 1995 in the western Gulf of Mexico (Gallaway et al. 2009) based on nine explosive removals in 1993, 1994, 1995, 1998, and 1999 (Gitschlag et al. 2003). Based on the adjusted Bailey's mean abundance estimate in the present study there were 654,524 Red Snapper with a total weight of 1.05 million kg on the 533 platforms within the depth range of 18 to 46 m in the northern Gulf of Mexico (K. McCain, LGL Ecological Research Associates, personal communication). These estimates were approximately 2 to 14 times lower than the previous estimates of Red Snapper on platforms in the northern Gulf of Mexico (Gitschlag et al. 2003;

Wilson and Neiland 2004; Gallaway et al. 2009). Differences in Red Snapper abundance estimates was likely due to the reduced number of platforms in the northern Gulf of Mexico at present time compared to previous studies. Gallaway et al. (2009) estimated Red Snapper abundance based on 4,000 platforms that were present in 1995. Similarly, Red Snapper abundance estimates by Wilson and Neiland (2004) were based on approximately 2,500 platforms in the northern Gulf of Mexico. If this previous platform number of 2,500 is applied to the present study's mean abundance estimate, Red Snapper increased to 3 million, which is within the range reported by Wilson and Neiland (2004). If the 4,000-platform number was applied, Red Snapper increased to 4.9 million (Gallaway et al. 2009). This higher estimate in the present study compared to Gallaway et al. (2009) was again likely due to underestimates when counting fish mortalities after explosive removals.

In the present study platforms off Louisiana had the lowest Red Snapper abundance compared to Alabama and Texas. In contrast, Karnauskas et al. (2017) reported higher relative abundance of Red Snapper at platforms off Louisiana compared to other areas in the northern Gulf of Mexico based on vertical longline samples. However, it is difficult to evaluate this earlier report because there is a lack of validation that vertical longline catches reflect the true abundance on sites sampled. In addition, direct platform sampling was only carried out over a limited spatial domain with the majority of sampled platforms located off the Louisiana coast. Furthermore, estimates of relative abundance differences between platforms and natural habitats were extended beyond the area upon which predictions were based (Karnauskas et al. 2017). The present study encompassed increased sampling efforts on platform habitats over a larger geographical range and abundance estimations were based on more almost 800 tagged and released Red Snapper, thus providing a more accurate Red Snapper abundance estimate on platforms.

In 2017, there were 593,229 Red Snapper with a weight of 949,166 kg, and in 2018 there were 464,776 Red Snapper with a weight of 743,642 kg on platforms at depths of 18 - 46 m in the northern Gulf of Mexico. Differences in abundance estimates by year and region could be due to recent changes from federal to state directed management. In 2017, the federal-NOAA fishing season was 3 days. In 2018, states bordering the Gulf of Mexico took over setting seasonal limits for the recreational Red Snapper fishery in both state and federal waters. In 2018, Alabama increased the Red Snapper season to 47 days starting 1 June 2018, Louisiana increased the season to 54 days, and Texas increased the season to 82 days. These seasons were substantially longer than the original 3-day federal recreational season in 2017. These increases in fishing season were reflected in higher mean Red Snapper abundance on platforms in 2017 than 2018, and an interaction effect with both Alabama and Texas having higher mean Red Snapper abundance in 2017 compared to 2018. The latest projections of the Red Snapper stock have indicated that the Western regions will have greater Red Snapper abundances, while the Eastern regions will have declines in Red Snapper abundance (SEDAR 2018). Abundance estimates on 2018 platforms support this prediction, with Texas and Louisiana platforms having higher abundance estimates than platforms in Alabama.

Abundance estimates varied widely with some platforms having around one hundred Red Snapper (platform-67) to more than 1,000 Red Snapper (platform-4 platform-30, platform-2, platform-49; Table 2). There was also a difference in Red Snapper abundance among years for Platform-2 (sampled in both 2017 and 2018), where abundance estimates differed by more than a thousand individuals between years. Previous studies have also reported large variation in Red

Snapper abundance with time (Stanley and Wilson 1997, 2000; Reynolds et al. 2018). For example, Red Snapper abundance showed high monthly variation on a Louisiana platform, with total fish abundance ranging from 2,000 to 28,000 fish over the course of one year (Stanley and Wilson 1996). A more recent study also indicated that Red Snapper abundances dominated both standing and toppled platforms but had seasonal variations (Reynolds et al. 2018).

Previous studies have also observed that Red Snapper densities vary by depth on platforms (Stanley and Wilson 1998, 2000; Wilson et al. 2003) and Red Snapper have been observed to stratify by size along the vertical gradient of a platform (Render 1995). In the present study all deep platforms (31 to 75 m depths) had an estimated abundance of more than 500 Red Snapper (Table 2). Additionally, depth of platforms (R^2 = 0.24, Figure 3) and distances-from-shore (R^2 = 0.36, Figure 4) appeared to account for some variation in Red Snapper abundance on platforms. Platform-4 was the deepest site with the greatest-distance-from-shore and had the highest Red Snapper abundance (10,698) followed by platform-30 (1,818; Figures 3 and 4). However, these two platforms were also substantially larger than all other platforms sampled in the present study, i.e., platform-4 had five separate platform stands and platform-30 had three separate platform stands. If platform-4 and platform-30 were removed from regression analysis, there were no longer significant effects of depth and distance-from-shore on Red Snapper abundance. Stanley and Wilson (1998, 2000) sampled three different platform sites with dual-beam hydroacoustics between 1994 - 1996 and reported significantly higher fish densities near the surface and bottom on two platform sites (water depth = 60 m and 22 m) on the continental shelf, but fish densities were close to zero below 100 m on a platform on the continental slope off Louisiana (depth = 219 m). However, on this slope platform 88% of fish still resided at 0 to 60 m depths (Stanley and Wilson 1998). Similarly, Wilson et al. (2003) reported that most Red

Snapper occurred between 40 - 50 m based on an ROV survey on a standing platform located in 90 m water depth on the Texas-Louisiana border. This platform had an estimated Red Snapper abundance of 400 fish in June 1999, similar to the Red Snapper abundance at platform-13 and platform-18 off Louisiana in the present study (Table 2). Additionally, platform-4 was located in 46 m water depth and had the highest Red Snapper abundance of all sites sampled, thus supporting these previous studies that indicated Red Snapper abundance was highest for platforms located at 40 - 50 m depths (Stanley and Wilson 1998; Wilson et al. 2003).

Out of seven platforms randomly selected for mark-recapture at the deeper depths (31 to 75 m depth contours), tagged Red Snapper were only recaptured on platform-13. Increased platform depth has been shown to negatively affect Red Snapper release condition (Patterson et al. 2001); thus, platform depth and location may have contributed to difficulty in the initial tagging effort at these deeper platforms during late summer months and may have been compounded by the Red Snapper fishing season.

Karnauskas et al. (2017) reported high relative abundance (CPUE) of Red Snapper on platforms in depths of 50 to 90 m off the Louisiana coast, but isolated high CPUE of Red Snapper at varying depths off Alabama. The present study observed varying Red Snapper abundances on platforms off coastal Alabama, but samples size was limited (n = 3) and further quantitative efforts are needed to better define Red Snapper distribution patterns off Alabama platforms. Additionally, a previous study reported that Red Snapper had the highest abundance during January and February and the lowest Red Snapper abundance in May and October on platforms off Louisiana (Stanley and Wilson 1997). Similarly, platform-30 in the present study was sampled between February and March of 2017 and had the second highest abundance among all platforms. Other studies report decreased Red Snapper abundance in winter months although these earlier studies were based on visual and photographic surveys that may have been affected by low visibility and high turbidity conditions (Gallaway 1980). In the present study, most (62 %) tagging and recapture efforts occurred in the summer months (June - August), and the abundance estimates may differ if platforms were sampled at different times of the year.

Abundance estimates from the present study closely mirror those reported in a concurrent hydroacoustics survey of northern Gulf of Mexico platforms (LGL, et al. 2018). This recent study estimated Red Snapper abundances at deeper depths (31 – 90 m) to be 2,980 (875 – 10,152, 95% CL) Red Snapper per platform, which was very similar to the 3,127 mean Red Snapper abundance at deeper depths (31 – 75 m) from the present study. Similarly, the median estimate of Red Snapper abundance with hydroacoustic methods for ten platforms at the same depths as the present study was 1,015 fish (LGL, et al. 2018), which is again very close to the mean Red Snapper abundance estimate of 1,228 in the present study. However, hydroacoustic surveys indicated that there were 359 Red Snapper on shallow platforms in 10 – 17 m depths, and 1, 015 Red Snapper on shallow platforms in 18-20 m depths. The present study estimated that shallow platforms (depth = 10 – 30 m) had a mean abundance estimate of 643 Red Snapper, which is different than the hydroacoustic abundance estimates reported at these depths. However, this mean abundance estimate of Red Snapper at shallow platforms in the present study is very close to the mean Red Snapper abundance estimate of both categories of shallow platforms from these hydroacoustic surveys.

Based on the Red Snapper mean abundance estimate of 1,228 fish per platform in the present study and the presence of 1,945 oil platforms in the Gulf of Mexico at the time of this study, there were 2.4 million Red Snapper with a total weight of 3.8 million kg residing on platforms. The latest Red Snapper stock assessment indicated that there were 75.6 million kg of

Red Snapper in 2016 in the Gulf of Mexico (SEDAR 52). Thus, Red Snapper on platforms only accounted for 5 % of the total Red Snapper biomass in the Gulf of Mexico. This indicates that the explosive removal of platforms may not have a significant effect on the Red Snapper stock size in the Gulf of Mexico. However, platforms are open ecological systems where fish abundances can fluctuate greatly due to immigration-emigration, recruitment, fishing mortality, seasonal patterns, storm events, and physical disturbances (i.e., low water temperatures and hypoxic conditions) and caution is advised when attempting to extrapolate abundance estimates of Red Snapper on platforms in the present study to wider shelf distributions.

Fishing Mortality on Platforms

Fishing mortality rates on platforms in the present study were lower than previous Red Snapper telemetry studies on platforms. A concurrent telemetry study on Red Snapper on the same study platforms as used in the present study reported an F = 0.86 (Everett et al. In review). Another telemetry study reported fishing mortality of Red Snapper on platforms to range between 0.36 and 6.7 (Peabody 2004). In the present study, the estimated fishing mortality for Red Snapper was F = 0.32 in both 2017 and 2018, which is lower than both of these previous fishing mortality estimates based on telemetry but higher than fishing mortality estimates on smaller submerged artificial reefs (F =0.27 - 0.44; Topping and Szedlmayer 2013; Williams-Grove and Szedlmayer 2016). Fishing mortality on platforms in 2017 was similar to fishing mortality estimates of conventionally tagged Red Snapper tagged in 2017 on large and small artificial reefs off Alabama (F = 0.30, Mudrak and Szedlmayer, In press). This difference was not surprising, as platforms likely had higher fishing effort compared to other artificial reefs because platforms structures are highly visible, making it easier to locate Red Snapper aggregations (Gordon 1993; Everett et al. In review).

In the present study, five platforms were observed with $F = 0$, and these platforms were all located greater than 30 km from the nearest coastline. Additionally, 75 % of the higher fishing mortality estimates ($F > 0.20$, $n = 12$) in the present study were from shallow platforms ($n = 9$). However, a significant relation was not detected for either depth or distance-from-shore on fishing mortality. Also, no significant region, year or interaction effects were detected on fishing mortality. In contrast, a previous study did indicate a significant decline in fishing mortality as distance from shore increased (Mudrak and Szedlmayer, In press). The lack of significant year, region, distance-from-shore or depth effects on fishing mortality in the present study were likely due to small sample sizes.

The present study assumed a 100 % fisher reporting rate based a concurrent telemetry study and a high reward of $150 for tag returns (Everett et al. In review). However, a previous study indicated that no reward amount would elicit a Red Snapper capture reporting rate of 100% (Brown and Wilkins 1978). Thus, if the actual reporting rate was less than 100% fishing mortality estimates would be underestimated.

Conclusions

The Red Snapper fishery is one of the most controversial fisheries in the northern Gulf of Mexico, and increasingly restrictive regulations encompassing decreased bag limits, shorter fishing seasons, and lower catch limits have led to increasing opposition from fishers regarding the fishery management process. Thus, accurate abundance estimates are critical for accurate stock assessments, sustainable fisheries, local economies, and stakeholder livelihoods in the northern Gulf of Mexico. The present study provides the first quantitative (rather than relative) Red Snapper abundance estimate on platforms in the northern Gulf of Mexico that will contribute to the management of this species. In the present study, mean Red Snapper abundance on platforms varied from previous abundance estimates on platforms. Differences in abundance estimates were likely due to fewer platforms present in the Gulf of Mexico at the time of the present study compared to past studies. Abundance differences may also be attributed to sampling issues from previous studies that would result in overestimation or underestimation of actual Red Snapper abundance. Few quantitative abundance estimates of Red Snapper on platforms exist, and many previous quantitative abundance estimates suffer from inadequate sample sizes. The present study increased sampling efforts to directly estimate Red Snapper abundance at 17 platform sites off the Texas, Louisiana, and Alabama coasts and applied estimates of fisher non-reporting, emigration, tag shedding, and tagging artifacts to address the known difficulties in Petersen mark-recapture studies.

Fishing mortality rates reported in the present study were lower than previous estimates of Red Snapper fishing mortality on platforms, but were higher than fishing mortality estimates on small, submerged artificial reefs. Higher fishing mortality rates of Red Snapper on platforms was likely due to ease in locating the visible platform structure compared to locating submerged artificial reef habitats.

In the present study, the mean Red Snapper abundance of all platforms was extrapolated to estimate the total number of Red Snapper residing on platforms in the Gulf of Mexico. Based on this mean abundance estimate, Red Snapper on platforms only comprised 5% of the total Red Snapper biomass. Concern has previously been raised regarding the frequent practice of explosive removals to eliminate non-producing platforms and the effect it might have on the Red Snapper stock size. However, the present study does not support that explosive removals will significantly affect Red Snapper stock size since only a small proportion of the total Red Snapper biomass resides at platforms.

The present study showed Red Snapper abundance on platforms varied significantly with an interaction of region and year. These differences in abundance by year may be due to ease of fishing restrictions as new state management took effect. Additionally, platforms at greater distances from shore and in deeper waters likely had higher Red Snapper abundances due to the increasing difficulty of access to fishers.

In the present study, the Baily's modification of the Petersen estimator was adjusted for many of the difficulties that have been previously identified for mark-recapture studies. These adjustments included information on Red Snapper tagging effects, emigrations, tag retention, and fisher reporting rates from previous telemetry studies. The accuracy of the present abundance estimates was dependent on the accuracy of these telemetry estimates. Future Red Snapper mark-recapture studies would benefit by expanding mark-recapture efforts to repeated censuses with replacement to relax the assumption of closure. Extensive effort should be allocated to obtaining the highest recapture rate possible. Furthermore, it may also be beneficial for future mark-recapture studies to concentrate marking efforts at smaller spatial scales in order to more accurately inform local management agencies in light of the implementation of recent state-level fishery management.

References

Agar, J., J.A. Stephen, A. Strelcheck, and A. Diagne. 2014. The Gulf of Mexico red snapper IFQ program: the first five years. Marine Resource Economics 29: 177-198.

Avyle, M.J.V.D. and R.S. Hayward. 1999. Dynamics of exploited fish populations. Pages 134-137 in C.C. Kohler and W.A. Hubert, editors. Inland fisheries management in North America, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Bailey, N.T.J. 1951. On estimating the size of mobile populations from recapture data. Biometrika 38:293-306.

Baker, M.S. Jr., C.A. Wilson, and D.L. van Gent. 2001. Testing assumptions of otolith radiometric aging with two long-lived fishes from the Gulf of Mexico. Canadian Journal of Fisheries and Aquatic Sciences 52:1244-1252.

Bolker, B.M., M.E. Brooks, C.J. Clark, S.W. Geange, J.R. Poulsen, M.H. Stevens, and J.S. White. 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology and Evolution 24:127–135.

Brennan, N.P., K.M Leber, H.L. Blackenship, J.M. Ransier, and R. DeBruler Jr. 2005. An evaluation of coded wire and elastomer tag performance in juvenile common snook under field and laboratory conditions. North American Journal of Fisheries Management 25: 437-445.

Brown, T.L., and B.T. Wilkins. 1978. Clues to reasons for nonresponse, and its effect upon variable estimates. Journal of Leisure Research 10:226–231.

Bull, A.A., and J.J. Kendall, Jr. 1994. An indication of the process: offshore platforms asa rtificial reefs in the Gulf of Mexico. Bulletin of Marine Science 55:1086-1098.

Carlson, S.R., L.G. Coggins, Jr., and C.O. Swanton. 1998. A simple stratified design for mark-recapture estimation of salmon smolt abundance. Alaska Fishery Research Bulletin 5(2): 88-102.

Chapman, D.G. 1951. Some properties of hypergeometric distribution with applications to zoological sample censuses. University of California Publications in Statistics 1:131-160.

Collins, A.L., G.R Fitzhugh, L. Mourand, L.A. Lombardi, W.T. Walling, W.A. Fable, M.R. Burnett, and R.J. Allman. 2001. Preliminary results from a continuing study of spawning and fecundity in the red snapper (*Lutjanidae: Lutjanus campechanus*) from the Gulf of Mexico, 1998-1999. Gulf and Caribbean Fisheries Institute. 52:34-47.

Continental Shelf Associates, Inc. 2004. Explosive removal of offshore structures - information synthesis report. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2003-070, New Orleans, LA.

Cowan, J.H. Jr., and K.A. Rose. 2016. Oil and gas platforms in the Gulf of Mexico: their relationship to fish and fisheries. Chapter 5 *in* H. Mikkola, editor. Fisheries and aquaculture in the modern world. InTech.

Dauterive, L. 2000. Rigs-to-reefs policy, progress, and perspective. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Report MMS 2000-073, New Orleans, LA.

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

Everett, A. G., S. T. Szedlmayer, and B. J. Gallaway. In press. Red snapper (*Lutjanus campechanus*) movement patterns based on acoustic positioning around oil and gas platforms in the northern Gulf of Mexico.

Gallaway, B.J. 1980. Pelagic, reef and demersal fishes and macrocrustacean / biofouling communities. Volume 2 *in* W.B. Jackson and E.F. Wilkens, editors. Environmental assessment of Buccaneer gas and oil field in northwestern Gulf of Mexico, 1975-1980. NOAA Technical Memorandum NMFS-SEFC 48.

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

Gazey,W.J., and M.J. Staley. 1986. Population Estimation from mark recapture experiments using a sequential Bayes algorithm. Ecology 67:94.

Gitschlag, G.R., and M.L. Renaud. 1994. Field experiments on survival rates of caged and released red snapper. North American Journal of Fisheries Management 14:131-136.

Gitschlag, G.R., M.J. Schirripa, and J.E. Powers. 2000. Estimation of fisheries impacts due to underwater explosives used to sever and salvage oil and gas platforms in the U.S. Gulf of Mexico: Final report. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2000-087, New Orleans, LA

Gitschlag, G.R., M.J. Schirripa, J.E. Powers. 2003. Impacts of red snapper mortality associated with the explosive removal of oil and gas structures on stock assessments of red snapper in the Gulf of Mexico, Pages 83-94 *in* D.R. Stanley and A. Scarborough-Bull, editors. Fisheries, reefs, and offshore development. American Fisheries Society Symposium 36, Bethesda, Maryland.

Goodyear, C.P. 1995. Red snapper in U.S. waters of the Gulf of Mexico. National Oceanic Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Miami Laboratory, Miami, FL, 171 pp.

Gordon, Jr. W.R. 1993. Travel characteristics of marine anglers using oil and gas platforms in the central Gulf of Mexico. Mar Fish Rev 55(1):25-31.

Gresswell, R.E., W.J. Liss, G.A. Lomnicky, E.K. Deimling, R.L. Hoffman, and T. Tyler. 1997. Using Mark-Recapture Methods to Estimate Fish Abundance in Small Mountain Lakes. Northwest Science 71(1): 39-44.

Hood, P. B., A. J. Strelcheck, and P. Steele. 2007. A history of red snapper management in the Gulf of Mexico. In Red Snapper Ecology and Fisheries in the U.S. Gulf of Mexico: American Fisheries Society Symposium 60, ed. W. F. Patterson III, J. H. Cowan Jr., G. R. Fitzhugh, and D. L. Nieland, 267-284. Bethesda, MD: American Fisheries Society.

Huelsenbeck, J.P., and K.A. Crandall. 1997. Phylogeny estimation and hypothesis testing using maximum likelihood. Annual Reviews in Ecology and Systematics 28:437– 466.

Jackson, M.W., D.L. Nieland, and J.H. Cowan Jr. 2006. Diel spawning periodicity of red snapper *Lutjanus campechanus* in the northern Gulf of Mexico. Journal of Fish Biology, 68: 695-706.

Kaiser, M.J. 2006. The Louisiana artificial reef program. Marine Policy, 30:605-623.

Karnauskas, M., Walter II, J.F., Campbell, M.D., Pollack, A.D., Drymon, J.M., and S. Powers. 17. Red snapper distribution on natural habitats and artificial structures in the northern Gulf of Mexico. Marine and Coastal Fisheries 9: 50-67.

LGL Ecological Research Associates, Inc., Auburn University, and Marine Science Institute. 2018. Explosive removal of structures: fisheries impact assessment: field season 2 assemblage characterization report. Prepared for the Bureau of Ocean Energy Management, New Orleans, LA.

Mesa, M.G., and C.B. Schreck. 1989. Electrofishing mark-recapture and depletion methodologies evoke behavioral and physiological changes in cutthroat trout. Transactions of the American Fisheries Society 118:644-658.

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

Mudrak, P.A., and S. T. Szedlmayer. In press. Fishing mortality estimates for Red Snapper based on acoustic telemetry and conventional mark-recapture. In S.T. Szedlmayer and S.A. Bortone, editors. Red Snapper biology in a changing world, Boca Raton: CRC Press.

Nieland, D.L., and C.A. Wilson. 2003. Red snapper recruitment to and disappearance from oil and gas platforms in the northern Gulf of Mexico. Pages 73-81 *in* D.R. Stanberg and A. Scarborough-Bull, Editors. Fisheries, Reefs, and Offshore Development. American Fisheries Society, Symposium 36, Bethesda, Maryland.

Peabody, M.B. 2004. The fidelity of red snapper (*Lutjanus campechanus*) to petroleum platforms and artificial reefs in the northern Gulf of Mexico. Master's Thesis, Louisiana State University.

Patterson, W.F. III., J. C. Watterson, R.L. Ship, and J.H. Cowan, Jr. 2001. Movement of tagged red snapper in the northern Gulf of Mexico. Transactions of the American Fisheries Society 130: 533-545.

Petersen, C. G. J. 1896. The yearly immigration of young plaice into the Limfjord from the German Sea. Report of the Danish Biological Station (1895). 6: 5-84.

Peterson, D.L., M.B. Bain, and N. Haley. 2000. Evidence of declining recruitment of Atlantic sturgeon in the Hudson River. North American Journal of Fisheries Management 20: 231-238.

Peterson, J.T., R.F. Thurow, and J.W. Guzevich. 2004. An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462-475.

Peterson, N.P., and C.J. Cedereholm. 1984. A comparison of the removal and mark-recapture methods of population estimation for juvenile coho salmon in a small stream. North American Journal of Fisheries Management 4: 99-102.

Phelps, R.P., and D. Rodriguez. 2011. Effects of tag type on red snapper *Lutjanus campechanus* tag retention, growth, and survival under hatchery conditions. Journal of Applied Ichthyology 27: 1169-1172.

Piraino, M.N., and S.T. Szedlmayer. 2014. Fine-scale movements and home range of red snapper around artificial reefs in the northern Gulf of Mexico. Transactions of the American Fisheries Society 143: 988-998.

Polacheck, T., J.P. Eveson, G.M Laslett, K.H. Pollock, and W.S. Hearn. 2006. Integrating catch-at-age and multiyear tagging data: a combined Brownie and Petersen estimation approach in a fishery context. Canadian Journal of Fisheries and Aquatic Sciences 63(3): 534-548.

Pulsipher, A.G., O.O. Hedare, D.V. Mesyanzhinov, A. Dupont, and Q.L. Zhu. 2001. Forecasting the number of offshore platforms on the Gulf of Mexico OCS to the year 2023. Prepared by the Center for Energy Studies, Louisiana State University, Baton Rouge, LA. OCS Study MMS 2001-013. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA.

Render, J.H. 1995. The life history (age, growth, and reproduction) of red snapper (*Lutjanus campechanus*) and its affinity for oil and gas platforms. Ph.D. Dissertation, Louisiana State University, Baton Rouge, LA, 76 pp.

Reynolds, E.M., Cowan, J.H. Jr., Lewis, K.A., and K.A. Simonsen. 2018. Method for estimating relative abundance and species composition around oil and gas platforms in the northern Gulf of Mexico, U.S.A. Fisheries Research 201: 44-55.

Ricker W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada 191:1-382.

Rosenberger, A.E., and J.B. Dunham. 2005. Validation of abundance estimates from mark-recapture and removal techniques for rainbow trout captured by electrofishing in small streams. North American Journal of Fisheries Management 25:1395-1410.

Seaman, W.J., W.J. Lindberg, C.R. Gilbert, and T.K. Frazer. 1989. Fish habitat provided by obsolete petroleum platforms off southern Florida. Bulletin of Marine Science 44:1014-1022.

Seber, G.A.F. 1982. The Estimation of Animal Abundance and Related Parameters. 2nd edition. Griffin, London.

Seavy, N.E., S. Quader, J.D. Alexander, and C.J. Ralph. 2005. Generalized linear models and point count data: statistical considerations for the design and analysis of monitoring studies. Pages 744–753 *in* C. J. Ralph and T. D. Rich, editors. Bird conservation implementation and integration in the Americas: proceedings of the third international partners in flight conference, 2nd volume. U.S. Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-191, Albany, California.

Sedberry, G. R., J.C. McGovern, and C.A. Barans. 1999. A comparison of fish populations in Gray's Reef National Marine Sanctuary to similar habitats off the southeastern U.S.: implications for reef fish and sanctuary management. Proceedings of the Gulf and Caribbean Fisheries Institute 50:452–481.

Schirripa, M.J., and C.M. Legault. 1997. Status of the red snapper in U.S. waters of the Gulf of Mexico: updated through 1996. Southeast Fisheries Science Center, Miami Laboratory, NMFS, MIA-97/98-05.

Schirripa, M.J., and C.M. Legault. 1999. Status of the red snapper in U. S. waters of the Gulf of Mexico: updated through 1998. National Oceanographic and Atmospheric Administration, Sustainable Fisheries Division Contribution SFD-99/00-75. Miami, FL.

Schnabel, Z.E. 1938. The estimation of the total fish population of a lake. American Mathematical Monographs 45:348-368.

Schroepfer, R.L., and S.T. Szedlmayer. 2006. Estimates of residence and site fidelity for red snapper Lutjanus campechanus on artificial reefs in the northeastern Gulf of Mexico. Bulletin of Marine Science 78:93-101.

Seber, G.A.F. (1982). The Estimation of Animal Abundance and Related Parameters, 2nd ed. London: Griffin.

SEDAR. 2005. SEDAR7-Gulf of Mexico red snapper complete stock assessment report. SEDAR, Charleston, SC, 480 pp.

SEDAR. 2013. SEDAR 31 – Gulf of Mexico Red Snapper Stock Assessment Report. SEDAR, North Charleston SC. 1103 pp.

SEDAR. 2018. SEDAR 52-Gulf of Mexico Red Snapper Stock Assessment Report. SEDAR, North Charleston SC, 434 pp.

Sonnier, F., J. Teerling, and H.D. Hoese. 1976. Observations on the offshore reef and platform fish fauna of Louisiana. Copeia 1:105-111.

Stanley, T. R., and K. P. Bumham. 1999. A closure test for time-specific capture-recapture data. Environmental and Ecological Statistics 6:197-209.

Stanley, D.R., and C.A. Wilson. 1990. A fishery-dependent based study of fish species composition and associated catch rates around oil and gas structures off Louisiana. Fishery Bulletin 88:719-730.

Stanley, D.R., and C.A. Wilson. 1996. Abundance of fishes associated with a petroleum platform as measured with dual-beam hydroacoustics. ICES Journal of Marine Science 53:473-475.

Stanley, D.R., and C.A. Wilson. 1997. Seasonal and spatial variation in the abundance and size distribution of fishes associated with a petroleum platform in the northern Gulf of Mexico. Canadian Journal of Fisheries and Aquatic Sciences 54:1166-1176.

Stanley, D.R., and C.A. Wilson. 1998. Spatial variation in fish density at three petroleum platforms as measured with dual-beam hydroacoustics. Gulf of Mexico Science 16:73-82.

Stanley, D.R., and C.A. Wilson. 2000. Seasonal and spatial variation in the biomass and size frequency distribution of fish associated with oil and gas platforms in the northern Gulf Of Mexico. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2000-005, New Orleans, LA.

Stearns, S. 1883. Fluctuations in the fisheries of the U.S. Gulf of Mexico and the proposed investigation of them. Bulletin of the U.S. Fish Commission 3:467-468.

Syc, T.S., and S.T. Szedlmayer. 2012. A comparison of size and age of red snapper (*Lutjanus campechanus*) with the age of artificial reefs in the northern Gulf of Mexico. Fishery Bulletin-National Ocean and Atmospheric Administration 110(4):458-469.

Szedlmayer, S.T., and J.D. Lee. 2007. Diet shifts of juvenile red snapper (*Lutjanus campechanus*) with changes in habitat and fish size. Fisheries Bulletin 102: 366-375.

Szedlmayer, S.T., and R.L. Shipp. 1994. Movement and growth of red snapper, *Lutjanus campechanus*, from an artificial reef area in the northeastern Gulf of Mexico. Bulletin of Marine Science 55:887-896.

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

Topping, D.T., and S.T. Szedlmayer. 2013. Use of ultrasonic telemetry to estimate natural and fishing mortality of red snapper. Transactions of the American Fisheries Society 142:1090-1100.

Williams, L.J., J.L. Herbig, and S.T. Szedlmayer. 2015. A cage release method to improve fish tagging studies. Fisheries Research 172: 125-129.

Williams-Grove, L. J., and S. T. Szedlmayer. 2016. Acoustic positioning and movement patterns of red snapper *Lutjanus campechanus* around artificial reefs in the northern Gulf of Mexico. Marine Ecology Progress Series 553:233–251.

Wilson, C.A., and D.L. Nieland. 2004. The role of oil and gas platforms in providing habitat for northern Gulf of Mexico Red Snapper, *Lutjanus campechanus*. Proceedings of the Gulf and Caribbean Fisheries Institute 55:757-764.

Wilson, C.A., A. Pierce, and M.W. Miller. 2003. Rigs and Reefs: A comparison of the Fish Communities at to artificial reefs, a production platform, and a natural reef in the northern Gulf of Mexico. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2003-009, New Orleans, LA.

Wilson, C.A., V.R. Van Sickle, and D.L. Pope. 1987. Louisiana artificial reef plan. Louisiana Department of Wildlife and Fisheries, Technical Bulletin No. 41. Baton Rouge, LA: Louisiana State Sea Grant College Program. 143pp.

Table 1: Study platforms, tagging and recapture dates, and total number of Red Snapper tagged and recaptured at each platform. Factors for comparisons included regions (AL, TX, LA), depth, and distance-from-shore (Dist). Tags = number of Red Snapper initially tagged, Adj tags = number available for recapture after adjustments based on telemetry, Total = number of fish caught during recapture effort, Recap = number of recaptured marked Red Snapper.

Platform	Region	Depth (m)	Dist (km)	Tag date	Recap date	Tags	Adj tags	Total	Recap
30	AL	Shallow	20-60	17-Feb-17	28-Mar-17	33			0
30	AL	Shallow	20-60	10-Mar-17	28-Mar-17	31	51.4	105	2
2	TX	Shallow	20-60	10-Jul-17	31-Jul-17	47	38.3	158	3
4	TX	Deep	101-140	11-Jul-17	1-2-Aug-17	62	50.5	211	0
5.2	TX	Shallow	20-60	12-Jul-17	5-Sep-17	11	7.1	105	1
13	LA	Deep	61-100	17-Jul-17	14-Aug-17	101	80.2	95	18
9	LA	Shallow	61-100	18-Jul-17	15-Aug-17	30	34.4	55	2
11	LA	Shallow	61-100	20-Jul-17	5-Sep-17	63	38.8	77	11
7	LA	Shallow	61-100	21-Jul-17	16-Aug-17	30	24.3	57	2
18	LA	Deep	20-60	7-Aug-17	13-Sep-17	43	34.2	13	0
19	LA	Shallow	61-100	8-Aug-17	13-Sep-17	4	3.2	6	0
49	LA	Shallow	61-100	8-May-18	3-Aug-18	67	41.8	75	2
47	LA	Shallow	61-100	9-May-18	2-Aug-18	33	20.2	107	3
50	LA	Deep	61-100	9-May-18	4-Aug-18	21	12.1	76	0
31	TX	Shallow	20-60	22-May-18	31-Jul-18	49	30.8	5	0
36	TX	Shallow	20-60	23-May-18	1-Aug-18	26	16.5	24	0
2	TX	Shallow	20-60	24-May-18	5-Jun-18	47	38.3	158	3
56	AL	Deep	20-60	25-Jun-18	5-Aug-18	6	4.7	19	0
57	AL	Deep	20-60	25-Jun-18	6-Aug-18	1	0.79	20	0
58	AL	Deep	20-60	25-Jun-18	7-Aug-18	6	4.7	3	0
66	AL	Shallow	61-100	26-Jun-18	8-Aug-18	32	22.6	31	1
67	AL	Shallow	20-60	10-Sep-18	20-Oct-18	16	12.6	16	1

Table 2: Petersen abundance estimates with Bailey's modification of Red Snapper at platforms in the northern Gulf of Mexico.

Platform	Unadjusted abundance (95% CI)	Adjusted abundance (95% CI)
2	1868 (763 - 4,670)	1,521 (621 - 3,802)
4	13,144 (13,144 - 2,797)	10,698 (2,276 - 10,698)
5.2	583 (177 - 1,060)	377 (114 - 685)
7	580 (212 - 1,450)	469 (172 - 1,172)
9	1,385 (294 - 2,007)	642 (235 - 1,605)
11	410 (237 - 768)	252 (146 - 473)
13	510 (330 - 829)	405 (262 - 658)
18	602 (128 - 602)	478 (102 - 478)
30	4,032 (1,475 - 10,080)	1,818 (220 - 1,500)
2.1	732 (222 - 1,331)	602 (182 - 1,094)
31	294 (63 - 294)	185 (39 - 185)
36	650 (138 - 650)	412 (88 - 412)
47	891 (364 - 2,228)	558 (228 - 1,395)
49	1,697 (621 - 4,243)	1,059 (387 - 2,648)
50	1617 (344 - 1617)	928 (197 - 928)
66	512 (155 - 930)	512 (155 - 931)
67	136 (41 - 247)	136 (41 - 247)

Table 3: Summary of the total number of Red Snapper tagged and caught by fishers within one year of initial tagging at each platform with corresponding survival (*S*) and mortality (*F*) rates.

Platform	Tagged	Total Caught	*S* (95% CL)	*F* (95% CL)
2	47	13	0.66 (0.50 - 0.79)	0.41 (0.23 - 0.69)
4	62	2	0.96 (0.86 - 0.99)	0.04 (0.00 - 0.15)
5.2	11	3	0.77 (0.40 - 0.94)	0.26 (0.06 - 0.90)
7	30	5	0.87 (0.66 - 0.96)	0.14 (0.04 - 0.41)
9	47	0	1.0	0.00
11	63	14	0.80 (0.66 - 0.89)	0.22 (0.12 - 0.41)
13	101	22	0.69 (0.57 - 0.79)	0.37 (0.24 - 0.56)
18	43	0	1.00	0.00
19	4	0	1.00	0.00
30	64	22	0.62 (0.49 - 0.74)	0.47 (0.30 - 0.71)
2.1	61	12	0.78 (0.65 - 0.87)	0.25 (0.14 - 0.43)
31	49	4	0.91 (0.79 - 0.97)	0.09 (0.03 - 0.24)
36	26	2	0.92 (0.72 - 0.98)	0.09 (0.02 - 0.33)
47	33	4	0.85 (0.66 - 0.94)	0.16 (0.06 - 0.41)
49	67	18	0.67 (0.54 - 0.78)	0.40 (0.24 - 0.62)
50	21	5	0.73 (0.49 - 0.88)	0.32 (0.13 - 0.72)
56	6	1	0.80 (0.31 - 0.97)	0.22 (0.03 - 1.17)
57	1	0	1.0	0.00
58	6	0	1.0	0.00
66	32	12	0.62 (0.43 - 0.79)	0.47 (0.24 - 0.85)
67	16	2	0.85 (0.55 - 0.96)	0.17 (0.04 - 0.60)

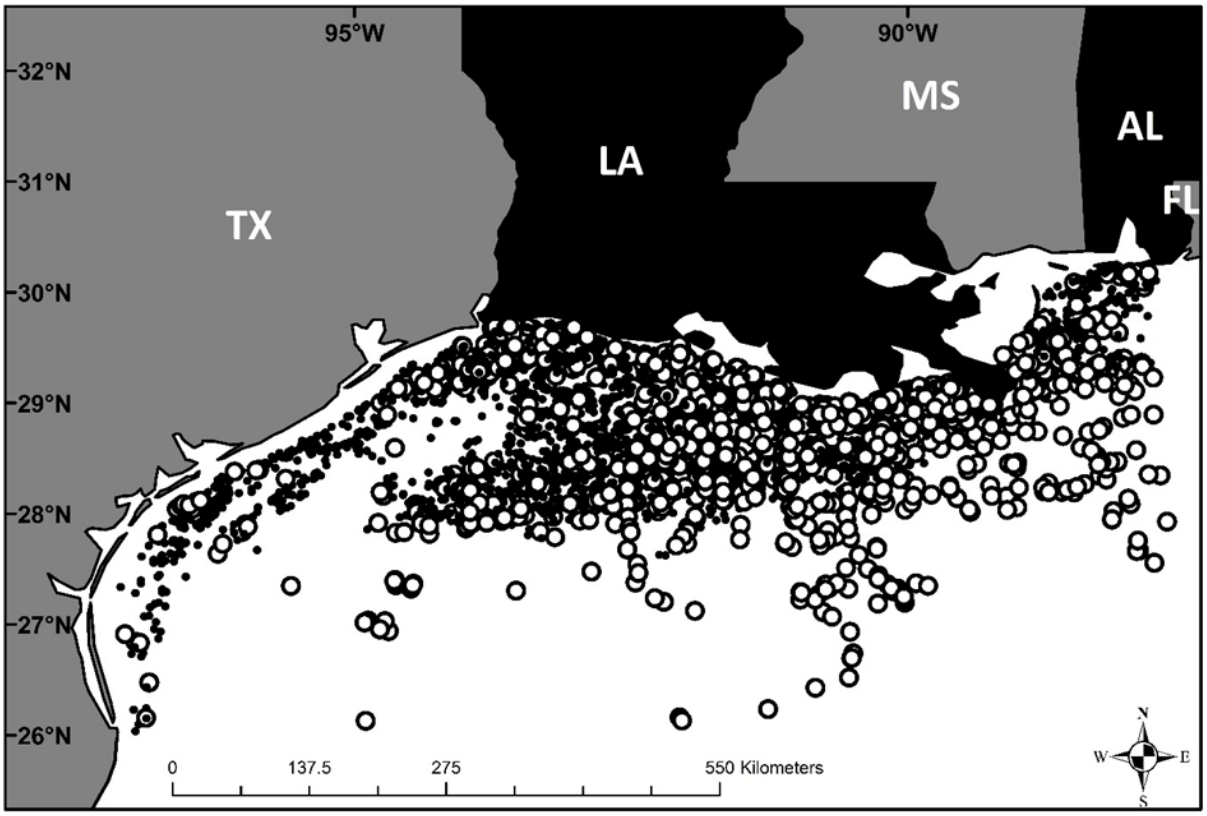

Figure 1: Osowski

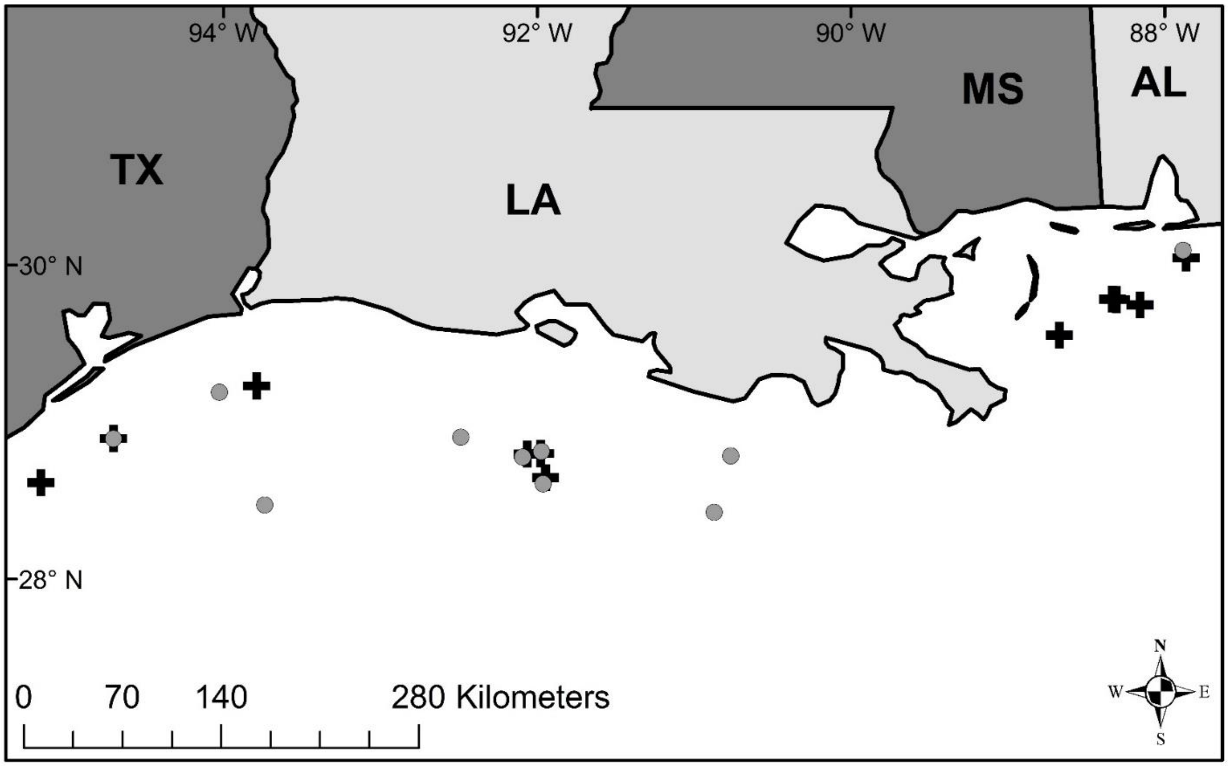

Figure 2: Osowski

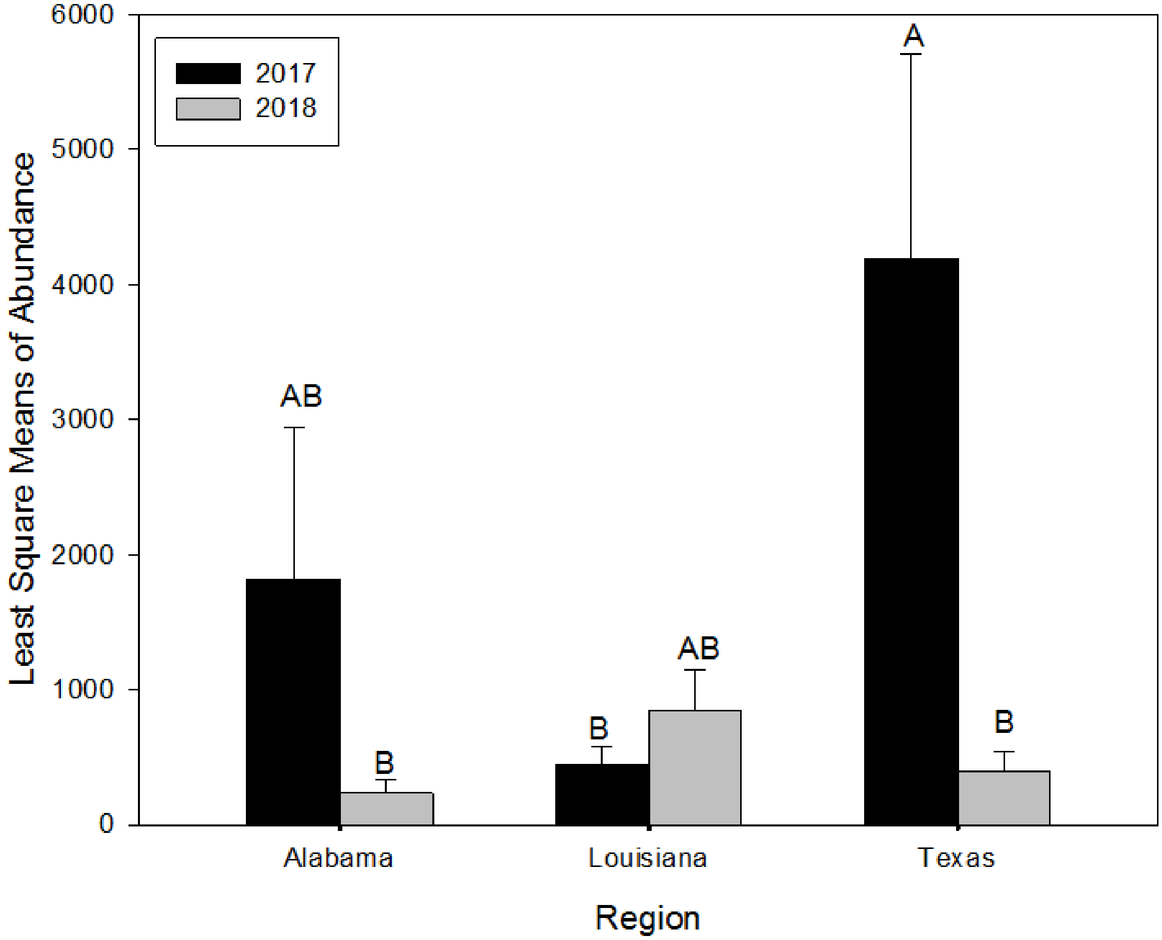

Figure 3: Osowski

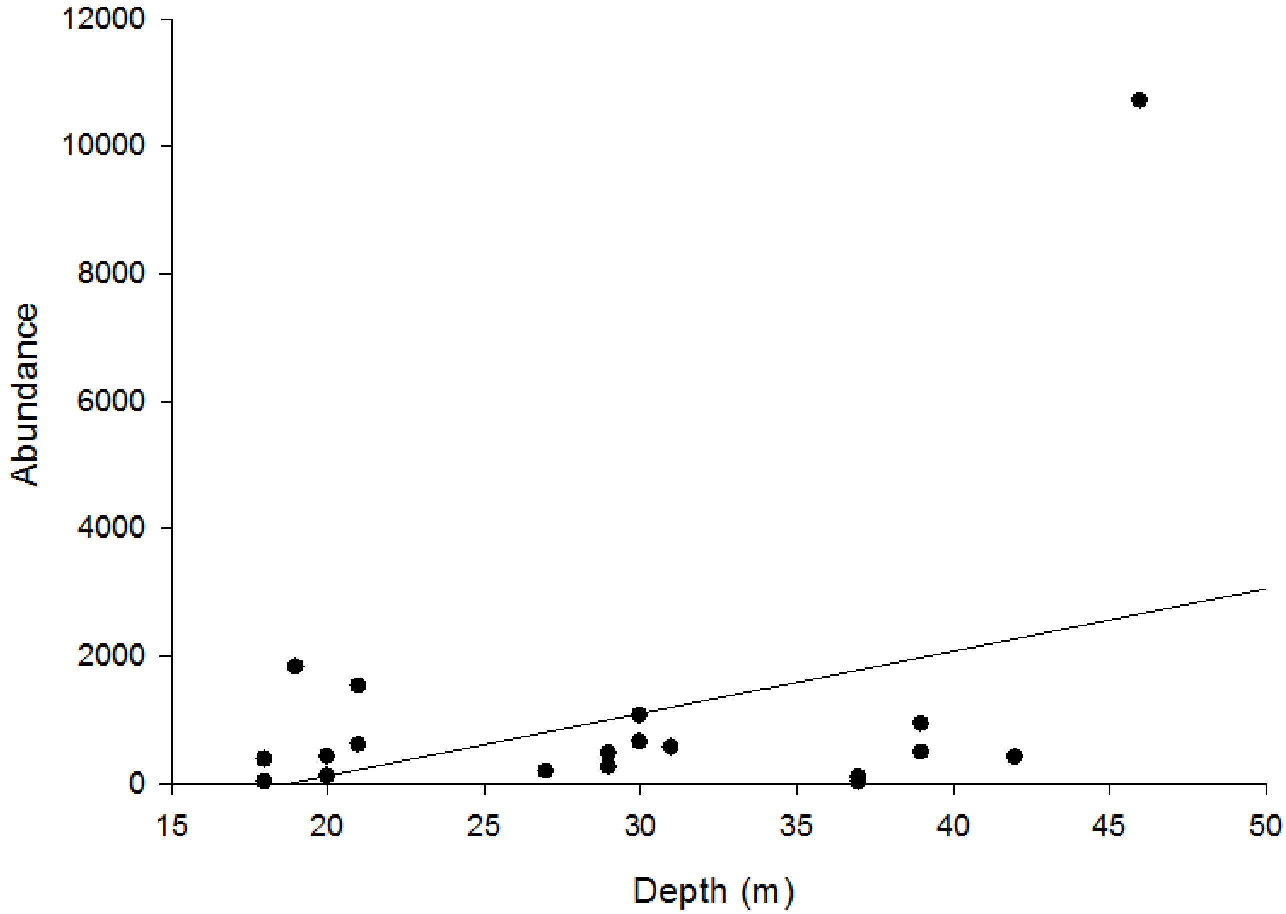

Figure 4: Osowski

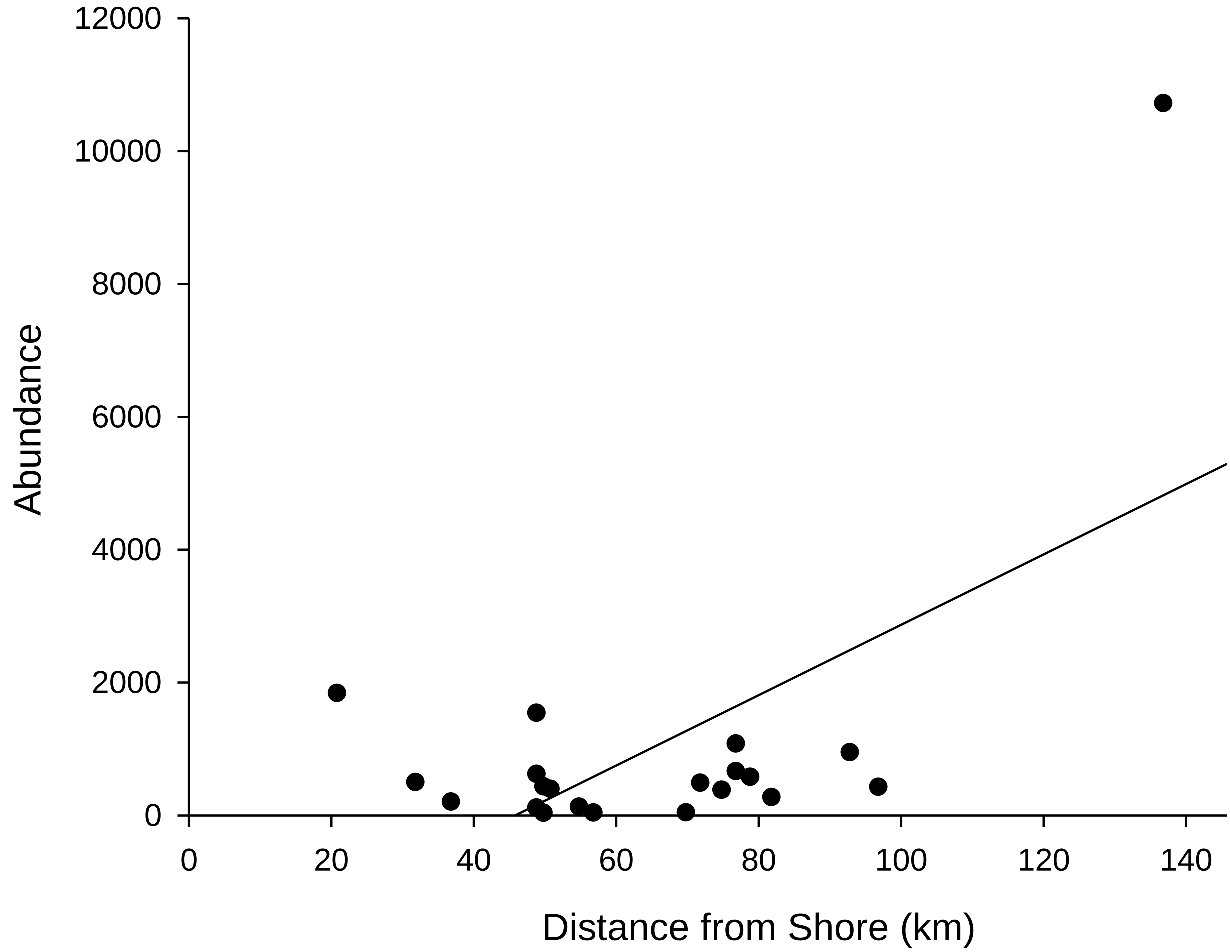

Figure 5: Osowski

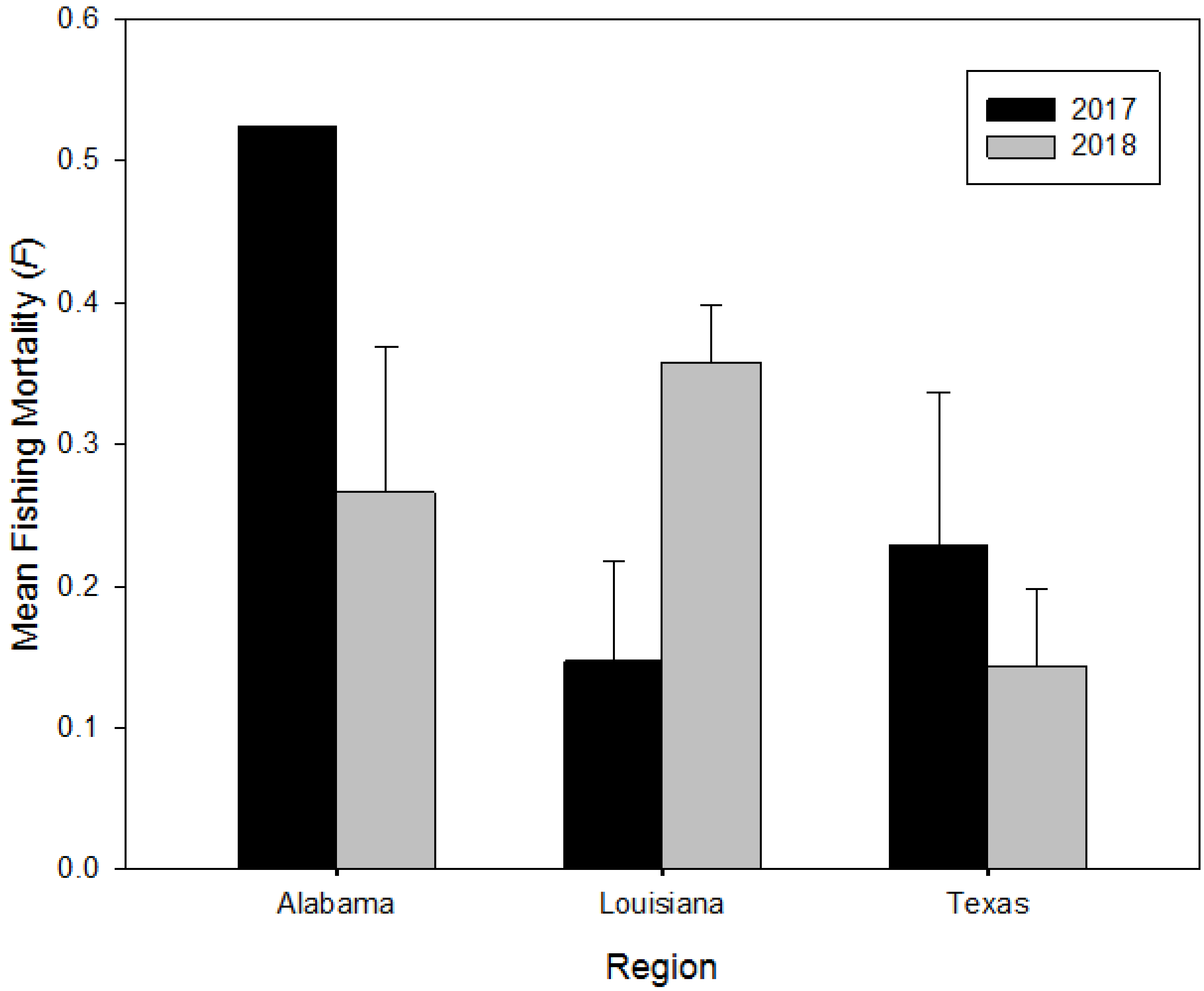

Figure 6: Osowski

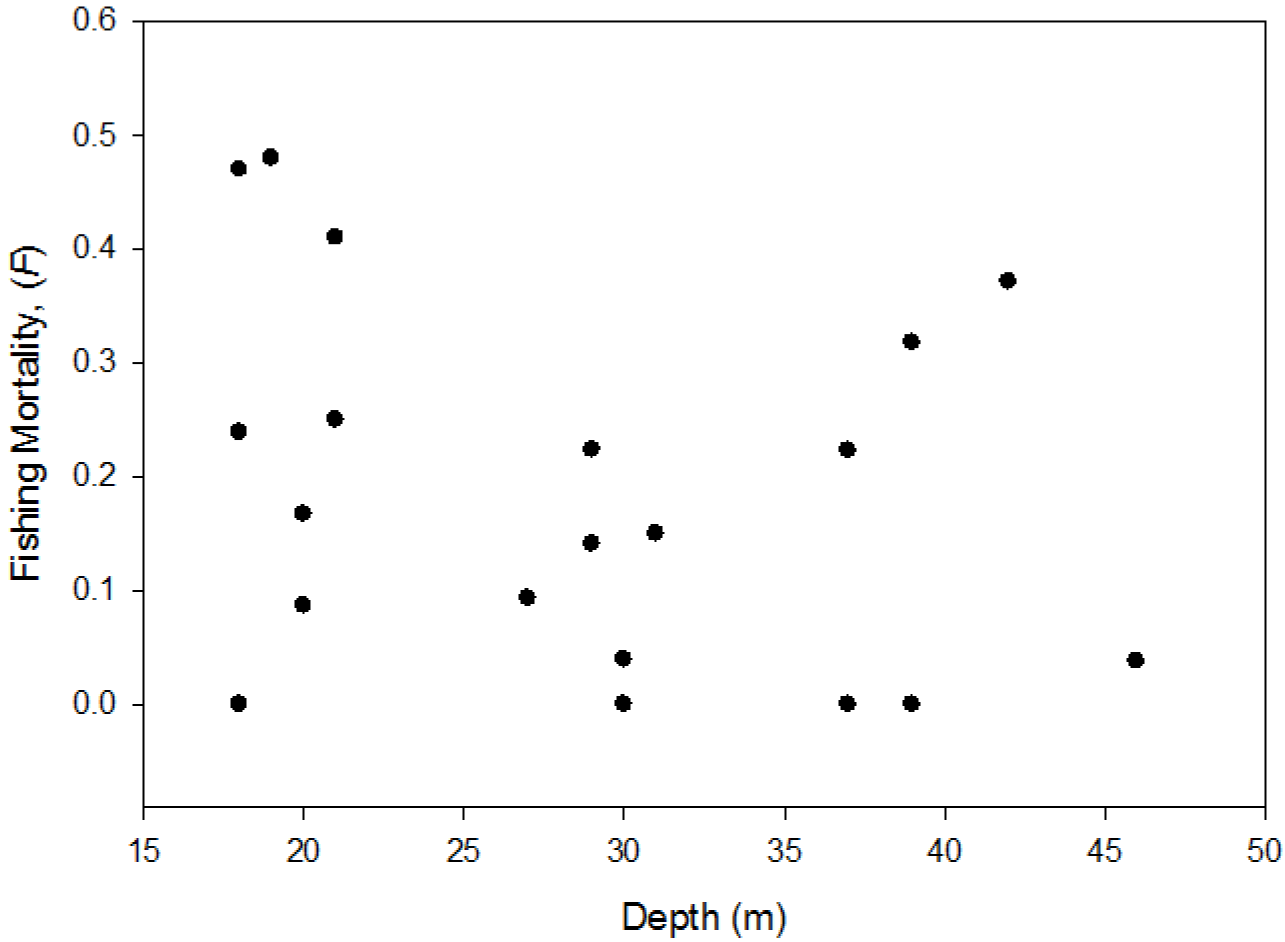

Figure 7: Osowski

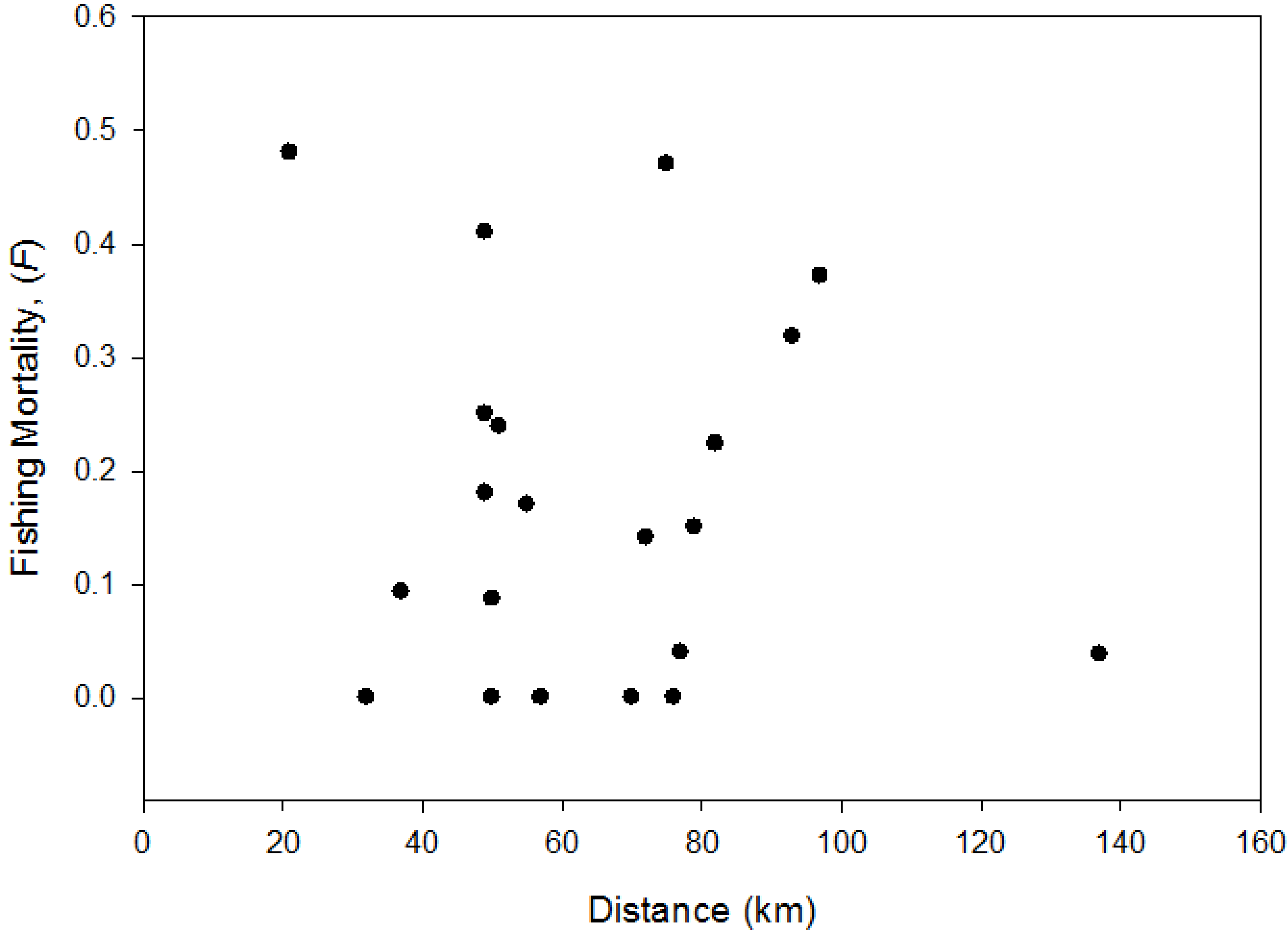

Figure 8: Osowski