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# Spatial Dynamics and Productivity of a Gulf of Mexico Commercial Reef Fish Fishery Following Large Scale Disturbance and Management Change

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# Spatial Dynamics and Productivity of a Gulf of Mexico Commercial Reef Fish Fishery Following Large Scale Disturbance and Management Change

by

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of the requirements for the degree of
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## DEDICATION

This work is dedicated to all the strong, intelligent, talented women that have inspired, supported, taught, and loved me through this process. Especially to my mother — the first person to show me what it truly means to work hard, persevere, and dream big. None of this would have been possible without you, Mom. I am eternally grateful, and I hope that I have made you proud.

*“Here’s to strong women – may we know them, may we be them, may we raise them.”*

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## ABSTRACT

The Gulf of Mexico commercial reef fish fishery has experienced significant management changes and disturbance in recent years, including transitioning two major fisheries from a traditional open access system into a limited entry individual fishing quota (IFQ) system in 2007 and 2010. Also in 2010, the *Deepwater Horizon* oil spill (DWH) released an estimated 4.9 million barrels of oil into the Gulf (~206 million U.S. gallons), and is still the largest U.S. environmental disaster to date. Emergency fishing closures initiated shortly after the oil spill began were successful in keeping tainted seafood from reaching markets. However, effects of DWH closures on fisher decision making, fishery productivity, and distribution of fishing effort all remain poorly understood. Understanding the range and magnitude of fishers' responses to perturbations — including regulatory change and human-induced environmental disasters — is critical for designing effective management and disaster response policies that can meet biological, ecological, economic, social, and sustainability objectives.

This work characterized the spatial and temporal patterns of productivity and fishing effort for the Gulf of Mexico (GoM) commercial reef fish fishery. Patterns of productivity and effort distribution were used to examine the response of fishers to management change and large-scale disturbance, namely the DWH fishing closures. Fisheries-dependent logbook trip reports were used to quantify revenue and catch-per-unit-effort (CPUE) patterns from 2000-2014. Novel to fisheries work in the GoM, complementary vessel monitoring systems (VMS) satellite tracking data were used to quantify high-resolution spatial distribution patterns over time,

relative to the DWH fishing closures. A general linear modeling (GLM) approach was also used to examine which variables may have contributed to resilience of fishers after DWH closures.

Results suggested that this fishery was largely resilient to the DWH fishing closures in 2010, although exact outcomes varied by region. Overall fleet-level productivity steadily increased over time, but regional patterns were based on major species in catch. Productivity in the western GoM was consistently highest over time, and trips in the west and central GoM were dominated by Red snapper (*Lutjanus campechanus*) and Vermilion snapper (*Rhomboplites aurorubens*). Trips in the east were dominated by Red grouper (*Epinephelus morio*) and Gag grouper (*Mycteroperca microlepis*). Shifts in spatial distribution to new productive fishing grounds or reduced competition *via* fewer vessels or trips may explain the increases in productivity observed over the study period.

Consolidation in the fleet was apparent, with fewer individual vessels and fewer total trips over time. However, the rate of vessel drop out after DWH (5%) was far below the annual background attrition rate of ~14-20%. Relative productivity patterns inside *vs.* outside the boundaries of fishing closures did not change over time, and there were even some increases in productivity observed during and after DWH in the eastern GoM. Yet, vessels that dropped out after DWH were concentrated in the north-central and eastern GoM. Distribution of fishing grounds before and after DWH were highly similar, and there were increases in effort along the outer West Florida Shelf. Variability in revenue and CPUE, CPUE magnitude, and magnitude of grouper landings were significant predictors of dropping out of the fishery in the GLMs. Synergies with the Red snapper or Grouper-Tilefish IFQs may have "primed" the fishery for resilience by eliminating inconsistent or marginal fishers before the oil spill, and may further explain some of the spatially varying patterns of productivity and attrition after 2010. Resilience was likely also enhanced by the more than $2 billion in emergency compensation payments made to captains, crew, and vessel owners for lost fishing income and assistance with oil remediation efforts.

This work stands to make a significant contribution to our understanding of how the DWH oil spill impacted fisheries and communities in the GoM. The results add to a growing body of literature suggesting that the acute population- and ecosystem-level impacts of the DWH oil spill were not as strong or severe as initially anticipated. This work also stands to make contributions to the broader understanding of how this fishery has performed in the wake of recent management change and major environmental disturbance.

# CHAPTER 1. INTRODUCTION

## Rationale

Changing management policies or other sudden disturbances can have significant impacts on fisheries and the fishers that operate within them. Understanding the potential range and magnitude of these impacts is critical for designing effective and adaptive management strategies that can meet diverse biological, ecological, social, and economic objectives. This is especially true as fisheries management increasingly moves toward multispecies and ecosystem-based approaches. Understanding vulnerability and resilience of fishing communities is equally as important for managers in order to identify communities that might be adversely affected by management decisions, and ensure that economic and social disruptions are minimal to allow for long-term sustainability (Jacob et al. 2013).

Over the past decade, there have been significant shifts in fisheries management in the Gulf of Mexico (GoM) as well as other large-scale and sudden disturbances, with implications for fisheries and coastal communities across the region. Several regulatory amendments have been added and modified in the Gulf of Mexico Fishery Management Council (GMFMC) Reef Fish Fishery Management Plan (FMP)[1] to better manage and rebuild stocks. Gear restrictions, closed seasons, spatial closures, size limits, and per-trip catch limits have all been used extensively. Two major commercial fisheries have moved from a traditional open access system — where resources are considered to be in a common pool without any designated access or

[1] FMP available online at: http://gulfcouncil.org/fishery-management/implemented-plans/reef-fish/.

property rights for users — to a limited access rights-based management system known as individual fishing quotas (IFQs), with defined annual allocation rights to total catch for each fisher. An IFQ system is often put in place to reduce the number of vessels in a fleet (i.e., reduce overcapacity), to in turn reduce harvest effort and competition, increase economic efficiency, and reduce overexploitation of a stock (Branch et al. 2006). Red snapper management shifted to an IFQ in 2007 to reduce overcapacity, overexploitation, and dangerous derby fishing (where fishers race to catch as much as they can as fast as they can; Hood et al. 2007, Agar et al. 2014). Groupers and tilefish have been managed together under a single IFQ since 2010 to similarly reduce derby fishing, lengthen fishing seasons, improve market conditions and profitability, and reduce bycatch and discard mortality.

Additionally, in 2010 the GoM was struck by the largest accidental oil spill in U.S. waters to date. Starting on April 20, 2010, an estimated 4.9 million barrels of oil (approximately 206 million US gallons) spilled from the *Deepwater Horizon* (DWH) oil well into the GoM, until the wellhead was finally capped 87 days later on July 15. To ensure that oil-contaminated seafood did not reach market, the National Oceanic and Atmospheric Administration (NOAA) in conjunction with affected states instituted a series of emergency fishing closures, from May 2 through November 15, 2010.[2] The closures were substantial in size, reaching a maximum of just over 229,000 $km^2$ (or 37% of the U.S. portion of the GoM) on June 2, 2010 (Figure 1.1). By November 15, 2010 all closures were removed except the area immediately around the wellhead, and by April 19, 2011 all closed areas had been reopened. These closures were successful in that no tainted seafood was reported to have entered the supply chain (Lubchenco et al. 2012).

[2] All DWH fishing closure information is from the NOAA Fisheries Southeast Regional Office *Deepwater Horizon/BP Oil Spill Information* page, available online at: http://sero.nmfs.noaa.gov/deepwater_horizon/size_percent_closure/index.html.

Surely, one would expect regulatory change and environmental disasters of these duration and magnitude to have impacts on the livelihoods of fishermen and the broader coastal communities that depend on income from fishing and related industries. While the impacts of IFQ implementation have been studied (with mandated five-year review programs as part of their design), the effects of DWH closures on fisher decision making, fishery productivity, the spatiotemporal distribution of fishing effort, and relative differences between displaced and non-displaced fishers remain poorly understood. There are few studies that explicitly report on changes in landings, catch composition, or revenue after DWH (McCrea-Strub et al. 2011, Sumaila et al. 2012, Murawski et al. 2016). However, these studies used coarse-resolution spatial and temporal data sets, thus limiting the interpretation and conclusions about how fisheries actively responded during and after the oil spill and ensuing closures. It is more likely that there were regionally-specific and temporally-varying responses to the oil spill, depending on the fishing community, proximity to the surface expression of oil, and location of fishing grounds relative to emergency fishing closures. It is therefore critical to understand how fishers and fishing communities responded to these significant perturbations in order to anticipate and mitigate potential negative impacts in the future.

**Overview of Dissertation**

This dissertation characterized the spatiotemporal patterns of productivity (i.e., ex-vessel revenue and catch-per-unit-effort, CPUE) in the GoM commercial reef fish fishery from 2000-2014, and quantified impacts from large-scale disturbance that occurred over the same period. The 2010 DWH fishing closures were used as an embedded experiment to study fishery response and resilience to disturbance. More specifically, using data from onboard observers, trip logbook reports, and satellite tracking vessel monitoring systems (VMS), this dissertation encompassed analyses of:

(1) Gulf-wide distribution of fishing effort and changes in fishery productivity from 2000-2014 (*Chapter 2*);

(2) Differences in effort distribution and productivity among displaced *vs.* non-displaced trips before, during, and after DWH fishing closures (*Chapter 3)*; and

(3) Potential factors contributing to resilience of fishers after the DWH fishing closures, with a quantification of shifting effort distribution after closures (*Chapter 4*).

The reef fish complex for the purposes of this work included snappers, groupers, tilefish, jacks, and triggerfish (with species as designated by NOAA Fisheries; see Tables 1.1 and 1.2). These groups were chosen due to their high recreational and commercial importance, the availability and consistency of data, and the large percentage of all commercial trips reporting landings for these groups (79% of trips from 2000-2014 reported one of the groups as the top revenue-earning group, and 82% of trips reported landings in general). All the species are also managed under the GMFMC as part of the Reef Fish FMP.[1] Analyses focused on trips/vessels that reported longline or vertical line (i.e., bandit-reel or handline) as the top revenue-earning gear, as these are the main gears used in this fishery (Scott-Denton et al. 2011) and represented 89% of reported trip data from 2000-2014. The results of this work are discussed in the context of a changing management landscape for this fishery — including implementation of IFQs in 2007 and 2010 — and potential impacts of environmental and oceanographic variability.

Unlike previous studies of DWH impacts, and unique to fisheries studies in the GoM in general, this work used high-resolution VMS data to examine fishing activity. VMS technology provides a geospatial reference point for a vessel approximately every hour for the duration of every trip, and has been required on all vessels with a commercial reef fish permit since 2007. Currently, there are ~800 valid commercial permits for GoM reef fish,[3] making this a very unique and robust data set. VMS data have been gaining in use and utility as a tool to characterize fishing activity and understand fishers' response to policy change, including the distribution of fishing pressure on various spatial and temporal scales (Witt and Godley 2007, Jennings and Lee 2012), the distribution of fishing pressure relative to regional habitat heterogeneity (Stelzenmüeller et al. 2008), and fishing behavior after implementation of large closed areas (Murawski et al. 2005). Complementary trip logbook (or, vessel trip report) and onboard observer data were also used to define "rules" for discriminating fishing activity and to quantify fishery productivity, thus enhancing the overall explanatory power of the VMS data. Integrating the VMS and logbook data in this way can help better identify areas where stocks are being targeted, both individually and in a multispecies context, and to assess drivers of behavioral change (Gerritsen and Lordan 2010). Given the importance of commercial fishing to the economic, cultural, and social well-being of many coastal communities in the GoM, this research stands to be of broad interest to fishing and tourism sectors, fisheries managers, researchers, government agencies, oil spill response agencies, and policy makers.

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**Table 1.1.** Top species group landed as a percentage of all logbook trip data from 2000-2014.

| Top group landed | % of total logbook trips (n=162,697) | % of final logbook trips used in productivity analyses (n=96,665 trips) |
|---|---:|---:|
| Shallow water groupers | 35.72 | 46.26 |
| Mid-depth snappers | 29.14 | 38.61 |
| Shallow water snappers | 9.09 | 6.25 |
| Deep water groupers | 3.03 | 4.07 |
| Jacks | 1.31 | 1.48 |
| Coastal pelagics | 16.04 | 1.04 |
| Tilefish | 0.55 | 0.71 |
| Grunts and Porgies | 1.03 | 0.68 |
| Sharks | 3.4 | 0.57 |
| Triggerfish | 0.15 | 0.19 |
| Other species | 0.28 | 0.07 |
| Tunas | 0.15 | 0.05 |

**Note:** Values are frequency of occurrence. The middle column contains percentages for all logbook trip data, and the right-most column contains percentages for only trips used in productivity analyses (see methods in Chapter 2). Groups are listed in descending order of percentage in the final logbook data set.

**Table 1.2.** Species included in each group used for logbook analyses of GoM longline and vertical line fishers.

| Top group | Species included | Scientific name |
|---|---|---|
| Shallow water snappers | Hogfish | *Lachnolaimus maximus* |
| | Lane snapper | *Lutjanus synagris* |
| | Mangrove snapper | *Lutjanus griseus* |
| | Mutton snapper | *Lutjanus analis* |
| | Yellowtail snapper | *Ocyurus chrysurus* |
| | Other snappers (general) | |
| Shallow water groupers | Black grouper | *Mycteroperca bonaci* |
| | Gag | *Mycteroperca microlepis* |
| | Red grouper | *Epinephelus morio* |
| | Red hind | *Epinephelus guttatus* |
| | Rock hind | *Epinephelus adscensionis* |
| | Scamp | *Mycteroperca phenax* |
| | Yellowfin grouper | *Mycteroperca venenosa* |
| | Yellowmouth grouper | *Mycteroperca interstitialis* |
| | Other groupers (general) | |
| Mid-depth snappers | Black snapper | *Apsilus dentatus* |
| | Dog snapper | *Lutjanus jocu* |
| | Mahogany snapper | *Lutjanus mahogoni* |
| | Queen snapper | *Etelis oculatus* |
| | Red snapper | *Lutjanus campechanus* |
| | Schoolmaster | *Lutjanus apodus* |
| | Silk snapper | *Lutjanus vivanus* |
| | Vermilion snapper | *Rhomboplites aurorubens* |
| | Other mid-depth snappers (general) | |
| Deep water groupers | Misty grouper | *Hyporthodus mystacinus* |
| | Snowy grouper | *Epinephelus niveatus* |
| | Speckled hind | *Epinephelus drummondhayi* |
| | Warsaw | *Epinephelus nigritus* |
| | Yellowedge grouper | *Epinephelus flavolimbatus* |
| Jacks | Greater amberjack | *Seriola dumerili* |
| | Lesser amberjack | *Seriola fasciata* |
| | Other jacks (general) | *Seriola sp.* |
| Tilefish | Blackline tilefish | *Caulolatilus cyanops* |
| | Golden tilefish | *Lopholatilus chamaeleonticeps* |
| | Goldface tilefish | *Caulolatilus chrysops* |
| | Grey tilefish | *Caulolatilus microps* |
| | Other tilefish | |
| Triggerfish | Spadefish | *Chaetodipterus faber* |
| | Triggerfish (general) | |

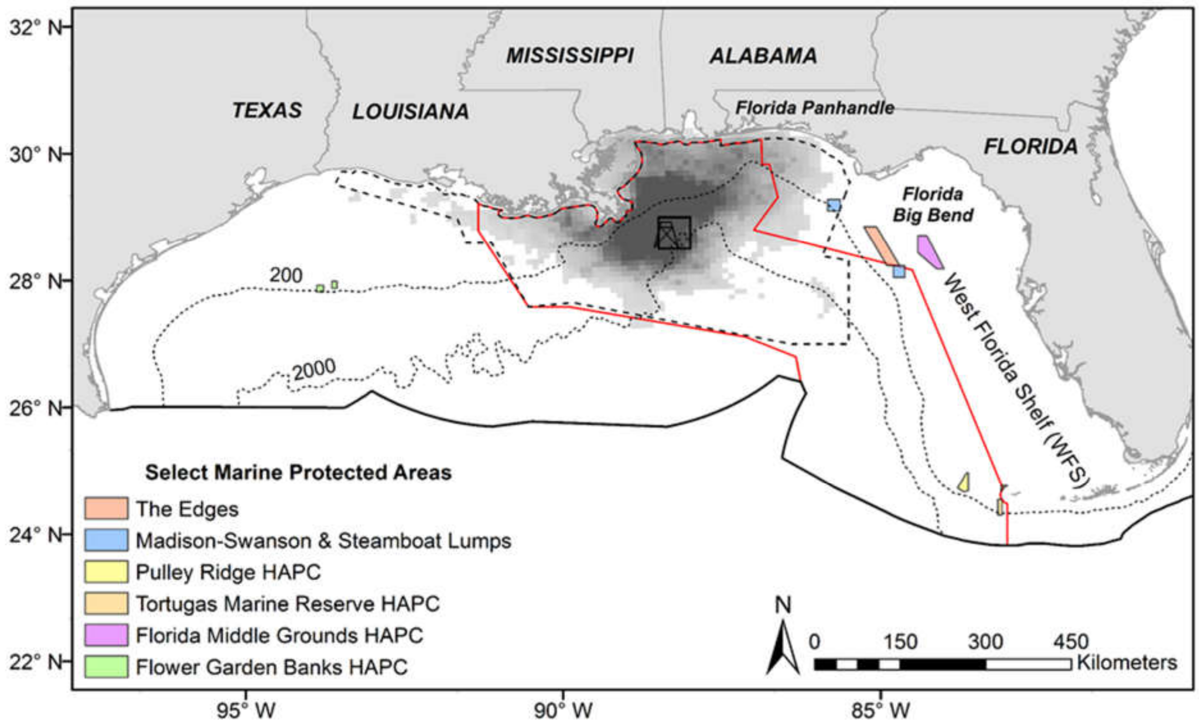


**Figure 1.1.** Study region and *Deepwater Horizon* fishing closures extent. The location of the DWH wellhead is marked with an oil derrick symbol and the surface expression of the DWH oil spill is shown in gray tones, with the darker colors representing longer exposure to oil. The maximum extent of the DWH emergency fishing closures (229,270 $km^2$ on June 2, 2010) is delineated with the solid red polygon, the dotted black polygon is the extent of the fishing closures on July 22, 2010 (149,026 $km^2$), and the solid black square is the closure immediately around the wellhead (2,697 $km^2$) that was closed until April 19, 2011. Marine protected areas shown include The Edges, Madison-Swanson, and Steamboat Lumps [designed to protect GoM reef fish; Code of Federal Regulations, Title 50, §622.34(a)] and Pulley Ridge, Tortugas Marine Reserve, Florida Middle Grounds, and Flower Garden Banks [designated as Habitat Areas of Particular Concern (HAPC) to protect GoM corals; Code of Federal Regulations, Title 50, §622.74(a-d)]. The 200-m and 2000-m isobaths are labeled and marked with dotted lines and the U.S. Exclusive Economic Zone (EEZ) is marked with a solid black line. Note that the southern end of the June 2 fishing closure intersects the EEZ. All fishery closure data and marine protected area polygons were downloaded from NOAA Fisheries Southeast Regional Office (available online at: http://sero.nmfs.noaa.gov/deepwater_horizon/closure_info/index.html and http://sero.nmfs.noaa.gov/maps_gis_data/fisheries/gom/GOM_index.html).

# CHAPTER 2. CHARACTERIZING SPATIOTEMPORAL PATTERNS OF EFFORT AND PRODUCTIVITY

## Introduction

The fluctuating nature of fisheries resources leads to high interannual variation in landings and income for fishers, necessitating risk-coping strategies and fishing behaviors that can ameliorate the inherent financial risk of depending on natural resources for income (Sethi 2010, Kasperski and Holland 2013). Variability in productivity may stem from changes in population size (e.g., after implementation of a marine reserve, increased fishing pressure), short and long-term environmental fluctuations (e.g., natural disasters, El Niño climate forcing, upwelling cycles), or external economic conditions (e.g., global fuel prices, consumer demand). Fishing pressure is also likely to differ by gear type, vessel size, geography, and patchiness of the environment. The demographics, socioeconomic conditions, politics, social networks, governance structures, culture, and geography of individual coastal communities will additionally determine how they interact with and influence the fishery system in which they operate (e.g., Himes 2003, Cinner et al. 2009, Jones 2013, Powell et al. 2018). Of course, none of these mechanisms occur in isolation, and are constantly interacting and evolving as part of a dynamic coupled social-ecological system. That is, fishing activity both drives and is driven by changes in management, the community, the environment, and overall stock condition.

Spatial distribution of fishing effort is one critical component of fisher behavior that can change in response to management and other externalities. Understanding the distribution and redistribution of fishing effort is critical for continued environmental and resource sustainability, as changes in fishing location choice and targeting behavior can have direct and indirect consequences. For example, displacement from a closed area may lead to congestion, increased competition, secondary displacement, overexploitation, and habitat degradation in areas outside the closure. More fishermen are competing for the same resource in a smaller space, meaning there is lower overall fishery resource available than before (Valcic 2009). Increased bycatch or discarding of non-target species can also occur when effort is redistributed (e.g., Abbott and Haynie 2012), although the ability of fishers to change gear or targeting behavior to avoid particular species has been well documented (e.g., Branch and Hilborn 2008).

Redistribution of effort is also likely to be heterogeneous over space and time, especially in an environment with patchy resources, and will be driven primarily by costs and expectations of profitability. Realistic models of fishing effort reallocation (that include a heterogeneous system and adaptive fisher behavior that responds to economic incentives) demonstrate an optimal utilization on the part of fishermen that maximizes return on investment (i.e., concentrating effort in areas and during times that yield maximum profit; Sanchirico and Wilen 2001, Smith and Wilen 2003, Dowling et al. 2012). At the same time, the ability of fishers to capitalize on new fishing locations or targeting behaviors as part of an optimal effort strategy (e.g., by engaging in "fishing the line" behavior around marine reserves) has the potential to increase landings and revenue for trips after displacement (see for instance Murawski et al. 2005 and Kellner et al. 2007). By understanding patterns of effort displacement and redistribution, regulations and incentive structures can be better designed to anticipate and address these issues.

This chapter used a combination of VMS and trip logbook (or, vessel trip report) records to characterize the distribution of fishing effort and productivity of the commercial reef fish fishery on an annual and Gulf-wide scale from 2000-2014. Analyses were conducted to understand changes in effort distribution, revenue, and catch-per-unit-effort (CPUE) over time and to identify significant fishing and productivity "hot spots". Specific attention was paid to changes in productivity for snappers and groupers, two particularly important, and contentious, groups for fishers and managers in the GoM. Results are discussed in the context of current regulations for this fishery and emergency fishing closures during DWH.

**Materials and Methods**

Trip logbook and VMS data were selected based on consistency and quality of reporting. Logbook revenue was inflation adjusted to 2008 US dollars ($2008) and all productivity variables were standardized to account for differences by vessel size (see detailed methods in Appendices A-C). VMS data were validated through a series of quality control steps to isolate a subset of data representing only fishing activity (e.g., eliminated travel to and from fishing grounds; see Appendices D-F). Logbook records from 2000-2014 were used to describe productivity patterns (ex-vessel revenue and catch-per-unit-effort, CPUE) and quantify any significant changes either over time or by region (western, central, and eastern GoM; Figure 2.1). Productivity patterns for snappers and groupers were additionally quantified, because they were the dominant top landed groups reported in the fishery (95% of trips; see Table 1.1), and continue to be important fishery and management targets. The fleet-level spatial distributions of fishing effort overall, by year, and by gear type were characterized and compared, and spatial statistics were used to quantify significant "hot spots" of fishing activity, revenue, and CPUE from 2008-2012. Methods for each of these analyses are described in more detail below.

### *Fishery Productivity*

Regions were defined as west, central, and east (Figure 2.1) based on initial examination of effort distributions across the GoM. The region designations were also consistent with delineations in previous studies on the GoM reef fish fishery (e.g., Weninger and Perruso 2013). Each trip was allocated to a region based on the logbook-reported top area fished (i.e., the logbook grid area producing a plurality of revenue for the trip) and its location relative to region boundaries (see Appendix G for NOAA Fisheries maps of logbook reporting grids). Three trips were eliminated because the reported top area could not be matched to a valid logbook reporting grid (final n=96,665 trips). Trip-level total revenue ($2008) and CPUE (gutted lbs. landed per number of hooks fished) were used as response variables in a series of analysis of variance (ANOVA) tests to examine annual (2000-2014) and regional differences in productivity. Landings were not used, since landings and revenue were shown to be highly correlated (Pearson correlation = 0.98, $p<0.001$) and patterns were similar between them. Snapper and grouper revenue and CPUE were similarly used as response variables in an additional series of ANOVAs to examine changes in these groups by region and over time. Tests used year and region as main effects and year × region as the interaction effect. All response variables were natural log ($\log_e$) transformed prior to analysis to approximate a normal distribution and equalize variances among years and regions. All tests were conducted at an alpha of 0.05, and significant results from the ANOVA were explored further using a Tukey-HSD *post-hoc* test for group means.

### *Spatial Distribution of Fishing Effort*

All effort densities were first calculated as the number of VMS pings per 0.15° × 0.15° grid cell (~15 km × 15 km), using the *raster* package in R (Hijmans et al. 2016). The density values thus represented the number of hourly VMS points identified as active fishing with a grid cell. Grid cells with less than three VMS pings were reassigned as "NA" and not mapped, to ensure confidentiality of the data. Fishing effort densities were calculated for the entire GoM aggregated over 2008-2012 (overall fleet distribution; n=22,427 trips), for each individual year, for each gear type (longline or vertical line), and for each gear × year combination.

A spatial difference index was calculated (Lee et al. 2010) to quantify the difference in effort distribution among all combinations of years (interannual fleet variability), between gears in each year, and among years within each gear type (interannual gear variability). Using the *raster* package, each raster layer was normalized so that the sum of all cell values was equal to 1. The per-cell absolute difference between two layers was then calculated, summed over the entire study region, and divided by two. This provided an index of difference that varied from zero, such that an index of 0 represented identical spatial distribution of fishing activity, and 1 represented maximum difference, or no overlap, in spatial use.

For comparison of densities between years or gears, density values were standardized (ranging from 0 to 1) relative to the maximum for a given time period or gear grouping. Relative differences in effort distribution (based on standardized effort density) were then calculated as the difference between individual density layers. A difference of 0 indicated no change in relative density, or complete overlap in distribution, while a value of 1 or -1 indicated maximum difference, or no overlap in spatial distribution.

*Significant Clustering of Effort, Revenue, and CPUE*

Significant spatial clustering of fishing effort, revenue, and CPUE were determined using the optimized hot spot analysis tool in ArcGIS 10.3 (ESRI, Redlands, CA, USA). The optimized hot spot analysis uses a Getis-Ord (*Gi**) spatial statistic (Ord and Getis 1995) to determine statistically significant clustering of incident points (e.g., fishing trip locations), or values associated with incident points (e.g., revenue and CPUE for each trip). The local sum for a given feature and its neighbors are compared proportionally to the global sum of all features; when the difference between the observed local sum and expected local sum is larger than would be expected by random chance, the region is considered to be a significant cluster. Significant clustering of high values is called a "hot spot" while significant clustering of low values is a called a "cold spot."

The optimized hot spot analysis tool first interrogates the data to choose the optimal spatial scale over which to conduct the analysis. Because of the large number of trip locations in the data set, the average distance that yielded 30 nearest neighbors was used to determine the analysis scale in most cases (Table 2.1). As a best practice, 30 nearest neighbors is the minimum that will yield reliable results when conducting spatial autocorrelation analyses.[4] For longline trips, the analysis scale was determined with an incremental spatial autocorrelation test. Distances were selected in increasing increments and the degree of clustering for each distance was measured with a global Moran's I statistic.[4] The distance that produced the peak degree of spatial clustering was subsequently used for the hot spot analysis (Table 2.1). The *Gi** statistic was calculated based on the pre-determined spatial scale, and a resultant z-score and p-value indicated whether significant clustering of trip locations or values were more pronounced than would be expected under a null hypothesis of a random distribution. Statistical significance was corrected for multiple testing and spatial dependence of neighboring features to reduce the

[4] ESRI. 2016. *How Spatial Autocorrelation (Global Moran's I) works*. Available online at: http://desktop.arcgis.com/en/arcmap/10.3/tools/spatial-statistics-toolbox/h-how-spatial-autocorrelation-moran-s-i-spatial-st.htm.

potential of false positives (Type I error) using a False Discovery Rate (FDR) correction (de Castro and Singer 2006).

Separate analyses were run to evaluate clustering of overall effort, revenue, and CPUE (aggregated across 2008-2012 and both gears) as well as effort, revenue, and CPUE by gear type (aggregated across 2008-2012). Since there is only a single revenue and CPUE value reported for each logbook trip, but potentially multiple VMS points for a given trip, the geographic midpoint of each individual trip (i.e., the mean fishing location) was calculated from VMS data and matched to the associated trip revenue and CPUE. This approach does not require the assumption that all trip values be distributed evenly across all VMS points within a trip, but rather assigns the entirety of a trip's effort to a single location. Thus, the hot spot analysis used trip midpoints rather than all locations identified as fishing (n=21,680 trips from 2008-2012). Values for revenue and CPUE were back transformed before being used in the analysis, and all tests were conducted at an alpha of 0.05. Clusters for effort returned with fewer than three data points were removed from the final results to ensure confidentiality of the data.

## Results

*Fishery Productivity*

*Total fleet over time.* The total number of unique individual vessels and logbook trips declined over time, with a greater number of trips consistently in the eastern region (Figure 2.2). There was a seasonal pattern to the number of logbook trips, with peak number of trips in the spring and summer months (Figure 2.3), but there were more trips in the eastern region irrespective of this seasonal fluctuation. All effects of year, region, and the interaction of year × region for total revenue and total CPUE were significant at the 0.05 level (Table 2.2). Mean trip revenue and CPUE were significantly different between all three regions, with the highest overall values in the western region, followed by the central and eastern regions (Figure 2.4; Tukey-HSD $p<0.001$ for all comparisons). When looking over time, both productivity measures increased gradually over the study period (Figure 2.5), with significantly greater values in 2014 compared with 2000 (Tukey-HSD, $p<0.001$). There were several significant shifts, or clusters of similar years, apparent as well (note clustering of similar letters in Figure 2.5). Mean trip revenue increased significantly from 2000 to 2001, 2006 to 2007, and 2011 to 2012 (Tukey-HSD, $p<0.001$ for all comparisons), and was significantly greater in the cluster of 2012-2014 than any of the preceding years. CPUE, on the other hand, increased significantly only from 2000 to 2001 and 2008 to 2009 (Tukey-HSD, $p<0.001$ for both comparisons), with no significant differences among the other consecutive years. After 2009, CPUE remained stable.

*Interaction of year and region.* The western region had higher revenues and CPUEs than the others, but only after 2006 (Figure 2.6). In 2013, the western region had the greatest revenue of the entire year × region series. The annual trend for each variable and the significant clustering of years appeared to be driven by the western region (compare Figures 2.5 and 2.6), which had significant increases in revenue from 2006-2007 and 2012-2013, increases in CPUE in 2006-2007, and a sharp decline in CPUE in 2014. The central region had a gradual increase in CPUE over the time series, but no significant increase among consecutive years. The eastern region had a significant increase in CPUE in 2009 and remained steady thereafter. In the central region, revenue was significantly greater in 2013 and 2014 than the preceding years, while in the eastern region revenue was greater in 2012-2014 than in preceding years (Figure 2.6).

*Snappers and groupers*. The effects of year, region, and the year × region interaction were significant for all tests of snapper and grouper productivity at the 0.05 level (Table 2.3).

Snapper revenue, snapper CPUE, and grouper CPUE were significantly higher in 2014 than in 2000 (Tukey-HSD, $p<0.001$), but grouper revenue was not (Tukey-HSD, $p=0.06$). There were also significant fluctuations in the intervening years (Figure 2.7). In particular, snapper productivity increased in 2009, 2010, and 2011 after especially low values in 2007-2008. Snapper revenue increased by 91, 135, and 64% in 2009, 2010, and 2011, respectively, and CPUE increased by 167, 85, and 40%, respectively. Grouper productivity was especially high in 2007-2009, with an average revenue and CPUE value that was 96-131% and 90-152% higher, respectively, than the average value for years either before or after.

Snapper revenue and CPUE were greatest overall in the western region while grouper CPUE and revenue were greatest in the eastern region (Figure 2.8). Landings in the west and central regions were dominated by mid-depth snappers (91% and 86%, respectively; Table 2.4), with Red snapper (*Lutjanus campechanus*) and Vermilion snapper (*Rhomboplites aurorubens*) making up a collective 85-90% of the top landed species (Table 2.5). Shallow water groupers were the top landed group in the east (67% of trips; Table 2.4), with Red grouper (*Epinephelus morio*) and Gag (*Mycteroperca microlepis*) constituting the top species for a collective 60% of trips (Table 2.5). This pattern held over time as well; snapper revenue and CPUE were significantly lowest in the eastern region over all years, and grouper revenue and CPUE were significantly greatest in the eastern region over all years (Figure 2.9). Values were not significantly different between the west and central regions in a majority of years for any of the snapper or grouper variables. While snapper values in the east were lowest of all regions, there were significant increases in snapper revenue and CPUE in the east in 2009, 2010, and 2011, with a significant decrease in 2013 (Figure 2.9, panels B and F). Grouper had an especially strong fluctuation in the eastern region around 2010: CPUE was at its highest point in 2009, followed by sharp declines in 2010 and 2011, and a return to pre-2009 levels in 2012-2014 (Figure 2.9G). Grouper revenue also declined sharply in the east in 2010 and 2011, with a return to pre-2010 levels in 2012-2014 (Figure 2.9C). The central region similarly experienced a significant decline in grouper revenue and CPUE in 2011 (Figure 2.9, panels D and H), but levels rebounded in 2012-2014.

*Spatial Distribution of Fishing Effort*

For the entire fleet, the greatest concentration of fishing effort was in the eastern GoM on the outer West Florida Shelf (WFS) and offshore of the Alabama coast along the 200-m isobath (Figure 2.10). The distribution of total effort was generally consistent from year to year, with an interannual fleet variability index ranging from 0.22-0.34 (i.e., 66-78% similarity in fishing grounds for all combinations of years; Table 2.6). However, there was a clear delineation of effort by gear type (Figure 2.11). Longline fishers (n=122 individual vessels) were heavily concentrated on the outer margins of the continental shelf near the 200-m isobath, with entire regions of the inner shelf empty of any significant effort. This distribution pattern closely followed the boundaries of existing longline gear restricted areas: the year-round bag/weight limit restricted area[5] around the entire GoM (dotted black line in Figure 2.11) and the seasonal (June-August) bottom longline closure[6] in the eastern GoM (solid black line in Figure 2.11).

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[5] Established in 1990 as part of Amendment 1 to the GMFMC Reef Fish FMP. Limits catch for vessels using longline or buoy gear to catch Gulf reef fish to either established bag limits or 5% of total weight onboard for species without bag limits. Includes all federal waters inside of 50 fathoms west of Cape San Blas, Florida (85°30'W longitude) and all federal waters inside of 20 fathoms east of Cape San Blas. See Code of Federal Regulations §622.35(c) and §622.38(b) for more details.

[6] Established in 2010 as part of Amendment 31 to the GMFMC Reef Fish FMP. Prohibits bottom longlining for Gulf reef fish in the area from June-August. The boundary includes all federal waters east of Cape San Blas, Florida along the 35-fathom contour. See Code of Federal Regulations §622.35(b) for more details.

Vertical line fishers (n=666 individual vessels), on the other hand, were distributed throughout the study region and the maximum effort density was concentrated along the Florida Big Bend and north-central GoM regions. The spatial difference index between the two gears overall was 0.70 (30% similarity in fishing grounds from 2008-2012) and similarity was not greater than 32% for any single year (Table 2.6). Interannual similarity within gears was high, with similarity in longline effort distribution ranging from 63-73% and vertical line effort distribution ranging from 65-79% (Table 2.6).

*Significant Clustering of Effort, Revenue, and CPUE*

There were 604 significant effort clusters for the entire fleet, with 122 clusters removed due to low point counts (less than 3), for a final total of 482 significant effort clusters (Table 2.1; Figure 2.12). Of those, 255 were significant hot spots (i.e., greater effort than would be expected by a random distribution of effort) and 227 were significant cold spots (i.e., lower effort than would be expected by a random distribution). Trips with vertical line had 364 significant clusters (with 210 hot spots and 154 cold spots) and trips with longline effort had 208 significant clusters (with 155 hot spots and 53 cold spots; Table 2.1; Figure 2.12). Similar to the total fleet effort density distribution, significant trip hot spots were located in the eastern GoM, most prominently in the Florida Big Bend and along the Florida panhandle and Alabama coasts (Figure 2.12A). The gear-specific trip clusters were separated similarly to the gear-specific densities, with a clear contribution from each gear to the overall pattern. Vertical line trip hot spots were concentrated in the eastern GoM and north-central in the Florida Big Bend and Alabama coast (Figure 2.12B). Longline trip hot spots were located further south on the mid-WFS (Figure 2.12C).

Significant clustering of revenue and CPUE did not strictly follow the patterns of trip clustering. Revenue and CPUE hot spots were distributed throughout the GoM, including areas that were significant trip effort *cold* spots (Figures 2.13 and 2.14). Revenue hot spots were located on the mid- to outer-continental shelf in the east and followed the contour of the 200-m isobath in the north-central and western GoM. Significant revenue cold spots were located from southern Florida to eastern Louisiana, and were further inshore than the revenue hot spots. The north-central region (Florida panhandle and Alabama coast) had a greater concentration of cold spots than hot spots. Gear-specific analyses revealed that vertical line trips were largely driving the overall pattern (Figure 12.3B and C). CPUE hotspots were patchier/more dispersed than revenue hot spots (Figure 2.14) and while there were a greater total number of significant clusters for vertical line revenue and CPUE, a greater proportion of the longline clusters were categorized as hot spots (Table 2.1).

## Discussion

The results from this chapter demonstrated annually increasing and recently stable productivity for the fishery as a whole, as well as regionally structured productivity and gear-dependent effort patterns. Overall annual productivity, as measured by ex-vessel revenue and CPUE, increased over the study period. Notably, mean per-trip revenue was significantly greater in the last three years (2012-2014) than in the period surrounding the DWH oil spill (2007-2011). While mean per-trip CPUE was not significantly greater after DWH, it was not significantly lower either. However, the magnitude of productivity and composition of top landed species varied by region. Productivity was highest in the western GoM overall and annually after 2006 (with especially high revenue in 2013). Red snapper and Vermilion snapper dominated trips in the west and central regions, and the aggregate and annual productivity for all snappers was significantly higher in the west and central regions. Trips in the eastern region were composed primarily of Red grouper and Gag, with significantly higher aggregate and annual grouper productivity than in the west or central regions. Importantly, the magnitude of revenue and CPUE for snappers was much greater than groupers (5-6 times greater overall and 10-12 times greater annually), therefore likely driving the overall regional pattern of greater productivity in the west. Given the large proportion of snappers and groupers reported for trips from 2000 through 2014 (Tables 2.4 and 2.5), it is highly unlikely that the regional patterns were attributable to other species groups.

Over the entire time range, the only productivity metric that did not significantly increase was grouper revenue. Grouper productivity did have three strong years in 2007-2009, followed by a sharp decline in 2010 and 2011 in the east and central GoM, and a significant increase in productivity in the regions in 2012-2014. The increase in grouper productivity from 2007-2009 was most likely the result of strong Red grouper year classes in 2006 and 2007 (Lombardi-Carlson 2014) and strong Gag year classes in 2006 and 2007 (Lombardi et al. 2013).

The increase in grouper productivity from 2012-2014, especially in the eastern GoM, may be attributable to enhanced Gag recruitment *via* deep upwelling and larval transport into coastal seagrass beds. The WFS bottom circulation is upwelling-favorable during Gag spawning months (i.e., late winter through early spring) and currents can transport demersal larvae eastward near to coastal seagrass habitats (Weisberg et al. 2014). In December 2010, an early onset of strong weather fronts led to anomalously strong winds and induced an upwelling event (Hu et al. 2011), which may have enhanced bottom transport of Gag larvae across the shelf and into settlement habitat. At the same time, the upwelling event triggered a month-long phytoplankton bloom from Mobile Bay to the Florida Keys, which could have provided a food source for newly hatched Gag larvae at the surface, thus enhancing larval survival and subsequent recruitment success.

The partitioning of snapper and grouper productivity into these regions is consistent with results from previous studies, documented fishing pressure on the groups, and known distributions of the stocks (Koenig et al. 1996, Weninger and Waters 2003, Scott-Denton et al. 2011, Zhang and Smith 2011). In particular, the dominance of Red snapper in the west and central GoM has been attributed to the proliferation of ~20,000 artificial reefs and ~4,000 oil and gas platforms in a region that is otherwise muddy and habitat-limited for juvenile recruitment (Gallaway et al. 2009, Shipp and Bortone 2009). A major portion of the Gag population in the GoM spawn on deep hard-bottom reefs of the WFS (Koenig and Coleman 1998) from late winter through early spring, and subsequent larval settlement occurs in eastern coastal seagrass beds. The upwelling-favorable bottom currents on the WFS during Gag spawning season allow for successful across-shelf transport of larvae near to coastal seagrass beds (Weisberg et al. 2014). In addition, Red grouper rely on the karst topography and carbonate-derived sands in the eastern GoM for excavating pits on the seafloor that serve as home territories and spawning sites (Coleman et al. 2010, Wall et al. 2011, Harter et al. 2017).

The three regions that were assigned to the logbook trips were somewhat arbitrary, although based on initial data screening and *a priori* knowledge of fishing pressure and species distributions. A finer-scale regional structure could be determined and assigned to logbook trips by instead using multivariate methods designed to detect structure in ecological communities. For example, similarity profile analysis (SIMPROF) objectively identifies members of groups based on species composition and abundance data, and provides a profile of underlying structure that can be visualized with a simple line plot (Clarke et al. 2008). Modeling the spatial variation of trips' species composition and abundance across different scales could additionally be performed using a technique known as multi-scale pattern analysis (MSPA; Jombart et al. 2009). MSPA is an ordination method that decomposes ecological variability into several spatial scales and presents results graphically. With habitat or environmental variables, a canonical (i.e., constrained) form of MSPA could be used to assess the potential spatial scales of species-environment relationships. These types of multivariate analyses could be performed using logbook trip landings and revenue (or, when available, discards data) to better define regional structure in this fishery.

The greatest concentration of fleet-level fishing effort was in the eastern GoM on the outer WFS and in the central GoM offshore of the Alabama coast along the 200-m isobath. Interannual effort distribution (quantified with a spatial difference index) was relatively consistent from 2008 through 2012, with 66-78% similarity in effort density among years. Significant effort hot spots were located from mid-Florida to Alabama, while significant revenue hot spots were distributed throughout the GoM along the seaward extent of trip locations and CPUE hot spots were in smaller more diffuse patches along the WFS and offshore of Louisiana. The significant hot spots of revenue and CPUE were likely attributable to the snapper and grouper productivity patterns discussed above. In particular, the western hot spots cannot be explained alone by number of trips taken or fishing effort intensity, as there were consistently fewer trips and lower effort density in the west than the central or eastern regions.

Still, spatial restrictions for the use of longline gear clearly had an impact on the effort distribution of the fleet. Effort density for longline fishers was concentrated outside the boundaries of two major areal closures: the longline/buoy gear restricted area[5] and the longline seasonal closed area.[6] Effort for vertical line fishers on the other hand was distributed throughout the GoM. These gear-specific effort distribution patterns were consistent with previous analyses of effort from observer data for this fishery (Scott-Denton et al. 2011). The locations of significant effort hot spots were similarly separated, with vertical line hot spots in the Florida panhandle and Big Bend regions, and longline hot spots further south and offshore on the mid-WFS. Consequently, the similarity in effort distribution between gears was only 24-32% annually. Effort distribution was much more similar within a gear, with 63-73% annual overlap for longline fishers and 65-79% annual overlap for vertical line fishers. Similar gear-dependent patterns with consistent interannual distributions have been reported elsewhere (Lee et al. 2010).

The patterns of effort density, effort hot spots, and revenue hot spots were all similar (compare Figures 2.11, 2.12, and 2.13), suggesting that ex-vessel revenue was contributing to location choice in this fishery in addition to gear regulations. Profitability, or the expectation of profitability, is the strongest determinant of fishing location choice in commercial fisheries (Smith 2000), with greater revenues or profit increasing probability of fishing in a particular location (Zhang and Smith 2011, Weninger and Perruso 2013). Fishers can exert strong control over the species that are caught, even in a multispecies fishery, and have the ability to adjust targeting behavior based on imposed regulations (Branch and Hilborn 2008, Weninger and Perruso 2013). Yet, the cost of fishing may increase when regulations necessitate a change in targeting behavior (e.g., longer search time, longer travel time to fishing grounds, modification of gear or bait). While trip costs were not included in this analysis (to allow for an examination of profitability rather than revenue), it would be possible to include some measure of cost for select trips in future work. Available cost data in logbooks include fuel, ice, oil, bait, groceries, packing fees, and purchase of IFQ allocation starting in 2007.

The dominance of Red snapper, Red grouper, and Gag in the trip catch, together with the overall increasing revenue and CPUE over the time series, additionally suggest that the IFQ systems for Red snapper and Grouper-Tilefish were playing a role in the patterns described. Both IFQ systems have been shown to be successful in reducing overcapacity with reductions in the number of active vessels, reducing or eliminating quota overages, reducing discard rates, lengthening open seasons (now both at 365 days), increasing the average price fishermen receive for catch, and increasing the overall economic efficiency and productivity of the fisheries (Hood et al. 2007, Agar et al. 2014, Brinson and Thunberg 2016). The observed consolidation of the fleet (i.e., fewer vessels and trips over time) together with increased revenue and CPUE for snappers since 2008 are in line with these previous assessments. Explicitly teasing apart the effects of the IFQs from the DWH closures will require data on which fishers were in the IFQs upon initiation, initial allocation amounts, and productivity before and after implementation. Those data are beyond the scope of what was available for this work.

Fisheries management aims to reduce excess fishing pressure to maintain a stock into the future, both on the short and long term, while minimizing negative social and economic impacts on communities. Fisheries management is therefore necessarily responsive to changing environmental, biological, and social factors. To be most effective, management schemes should be designed to either anticipate adaptations in fishers' behavior across multiple spatial scales and for multiple species, or be robust to changes in fisher behaviors (Abbott and Haynie 2012). Failure to include fisher decision-making and, perhaps more importantly, the potential for adaptive fisher behavior in management policies can severely undermine the biological, ecological, and economic objectives of fisheries management.

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**Table 2.1.** Optimized hot spot analysis distance band used and results for each test.

| Analysis | Fixed distance band (km) | # of significant clusters/points in clusters | % hot spots | % cold spots |
|---|---|---|---|---|
| All trip effort | 61.8 | 482 | 53 | 47 |
| All trip revenue | 8.1 | 7,971 | 40 | 60 |
| All trip CPUE | 8.1 | 3,836 | 57 | 43 |
| Vertical line trip effort | 63.7 | 364 | 58 | 42 |
| Vertical line trip revenue | 8.1 | 5,363 | 41 | 59 |
| Vertical line trip CPUE | 8.1 | 5,597 | 36 | 64 |
| Longline trip effort | 75.8 | 208 | 75 | 25 |
| Longline trip revenue | 19.9 | 620 | 67 | 33 |
| Longline trip CPUE | 25.0 | 124 | 56 | 44 |

**Note:** The distance bands for longline trips were determined using an incremental spatial autocorrelation test. All others were selected based on the average distance for 30 nearest neighbors.

**Table 2.2.** Results from ANOVA on overall fishery productivity by region and year from 2000-2014.

| Response | Effect | d.f. | SS | MS | F | P |
|---|---|---|---|---|---|---|
| ln(revenue) | Region | 2 | 895 | 447.6 | 341.6 | < 0.001 |
| | Year | 14 | 2,968 | 212 | 161.8 | < 0.001 |
| | Region × Year | 28 | 1,043 | 37.2 | 28.4 | < 0.001 |
| | Residual | 96,620 | 126,625 | 1.3 | | |
| | | d.f. | SS | MS | F | P |
| ln(CPUE) | Region | 2 | 2,786 | 1393.1 | 264.4 | < 0.001 |
| | Year | 14 | 6,735 | 481.1 | 91.3 | < 0.001 |
| | Region × Year | 28 | 1,842 | 65.8 | 12.5 | < 0.001 |
| | Residual | 96,620 | 509,011 | 5.3 | | |

**Table 2.3.** Results from ANOVA on snapper and grouper productivity by region and year from 2000-2014.

| Response | Effect | d.f. | SS | MS | F | P |
|---|---|---|---|---|---|---|
| ln(Snapper revenue) | Region | 2 | 291,549 | 145,775 | 13,027.2 | < 0.001 |
| | Year | 14 | 57,186 | 4,085 | 365.03 | < 0.001 |
| | Region × Year | 28 | 13,241 | 473 | 42.3 | < 0.001 |
| | Residual | 96,620 | 1,081,179 | 11 | | |
| ln(Grouper revenue) | | d.f. | SS | MS | F | P |
| | Region | 2 | 389,449 | 194,725 | 16,819.4 | < 0.001 |
| | Year | 14 | 8,188 | 585 | 50.5 | < 0.001 |
| | Region × Year | 28 | 5,648 | 202 | 17.4 | < 0.001 |
| | Residual | 96,620 | 1,118,606 | 12 | | |
| | | d.f. | SS | MS | F | P |
| ln(Snapper CPUE) | Region | 2 | 259,704 | 129,852 | 7,534.7 | < 0.001 |
| | Year | 14 | 59,457 | 4,247 | 246.4 | < 0.001 |
| | Region × Year | 28 | 16,194 | 578 | 33.6 | < 0.001 |
| | Residual | 96,620 | 1,665,134 | 17 | | |
| | | d.f. | SS | MS | F | P |
| ln(Grouper CPUE) | Region | 2 | 314,548 | 157,274 | 14,314.8 | < 0.001 |
| | Year | 14 | 6,278 | 448 | 40.8 | < 0.001 |
| | Region × Year | 28 | 5,862 | 209 | 19.1 | < 0.001 |
| | Residual | 96,620 | 1,061,546 | 11 | | |

**Table 2.4.** Top species group landed in each region from 2000-2014.

| Region | Top group | Total regional logbook trips | Percentage of regional trips |
|---|---|---|---|
| West | Mid-depth snappers | 8,851 | 91.1 |
| | Deep water groupers | 440 | 4.5 |
| | Shallow water snappers | 143 | 1.5 |
| | All others | 278 | 2.9 |
| Central | Mid-depth snappers | 18,020 | 85.7 |
| | Shallow water snappers | 944 | 4.5 |
| | Deep water groupers | 612 | 2.9 |
| | Jacks | 460 | 2.2 |
| | All others | 1,000 | 4.8 |
| East | Shallow water groupers | 44,361 | 67.3 |
| | Mid-depth snappers | 10,447 | 15.8 |
| | Shallow water snappers | 4,958 | 7.5 |
| | Deep water groupers | 2,886 | 4.4 |
| | All others | 3,265 | 5.0 |

**Table 2.5.** Top species landed in each region from 2000-2014.

| Region | Top species | Total regional logbook trips | Percentage of regional trips |
|---|---|---|---|
| West | Red snapper | 7,652 | 78.8 |
| | Vermilion snapper | 1,198 | 12.3 |
| | Yellowedge grouper | 306 | 3.2 |
| | Lane snapper | 134 | 1.4 |
| | Golden tilefish | 108 | 1.1 |
| Central | Red snapper | 12,686 | 60.3 |
| | Vermilion snapper | 5,324 | 25.3 |
| | Mangrove snapper | 900 | 4.3 |
| | Yellowedge grouper | 442 | 2.1 |
| | Greater amberjack | 387 | 1.8 |
| | King mackerel | 275 | 1.3 |
| East | Red grouper | 29,311 | 44.5 |
| | Gag | 11,055 | 16.8 |
| | Red snapper | 6,488 | 9.8 |
| | Vermilion snapper | 4,215 | 6.4 |
| | Yellowtail snapper | 3,605 | 5.5 |
| | Black grouper | 2,827 | 4.3 |
| | Yellowedge grouper | 2,264 | 3.4 |
| | Mangrove snapper | 965 | 1.5 |
| | Greater amberjack | 928 | 1.4 |

**Note:** Only species constituting greater than or equal to 1% of trips are shown.

**Table 2.6.** Spatial difference index and percent similarity of effort density distributions among years and gear types.

| Comparison | Year | Spatial difference index | % similarity in distribution |
|---|---|---|---|
| Between year (gears aggregated) | 2008-2009 | 0.25 | 75 |
| | 2008-2011 | 0.30 | 71 |
| | 2009-2011 | 0.32 | 68 |
| | 2010-2008 | 0.26 | 74 |
| | 2010-2009 | 0.26 | 74 |
| | 2010-2011 | 0.22 | 78 |
| | 2012-2008 | 0.26 | 74 |
| | 2012-2009 | 0.34 | 66 |
| | 2012-2010 | 0.29 | 71 |
| | 2012-2011 | 0.27 | 73 |
| Between gear, within year | 2008 | 0.76 | 24 |
| | 2009 | 0.72 | 28 |
| | 2010 | 0.72 | 28 |
| | 2011 | 0.71 | 30 |
| | 2012 | 0.68 | 32 |
| Within gear: Vertical line trips | 2008-2009 | 0.21 | 79 |
| | 2008-2011 | 0.33 | 67 |
| | 2009-2011 | 0.31 | 69 |
| | 2010-2008 | 0.28 | 72 |
| | 2010-2009 | 0.27 | 73 |
| | 2010-2011 | 0.23 | 77 |
| | 2012-2008 | 0.35 | 65 |
| | 2012-2009 | 0.32 | 68 |
| | 2012-2010 | 0.30 | 70 |
| | 2012-2011 | 0.26 | 74 |
| Within gear: Longline trips | 2008-2009 | 0.32 | 68 |
| | 2008-2011 | 0.34 | 66 |
| | 2009-2011 | 0.37 | 63 |
| | 2010-2008 | 0.32 | 68 |
| | 2010-2009 | 0.34 | 66 |
| | 2010-2011 | 0.28 | 72 |
| | 2012-2008 | 0.28 | 73 |
| | 2012-2009 | 0.37 | 63 |
| | 2012-2010 | 0.32 | 68 |
| | 2012-2011 | 0.34 | 66 |

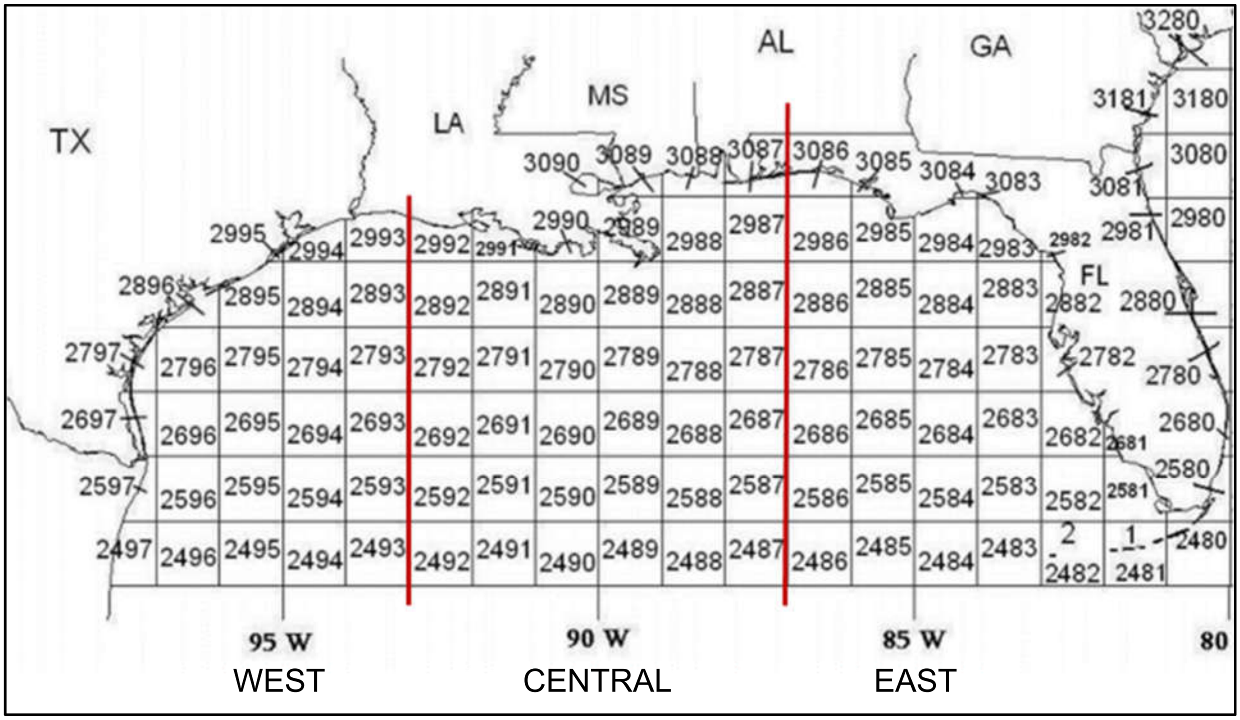


**Figure 2.1.** Geographic zone delineation for regional analyses, based on trip logbook statistical reporting grids. Map is from the 2013 reef fish logbook reporting form and was obtained online through the NOAA Southeast Fisheries Science Center *Fishermen and seafood dealers forms archive* (www.sefsc.noaa.gov/fisheries/reporting_archive.htm). Vertical red lines denote breaks for west, central, and east regions.

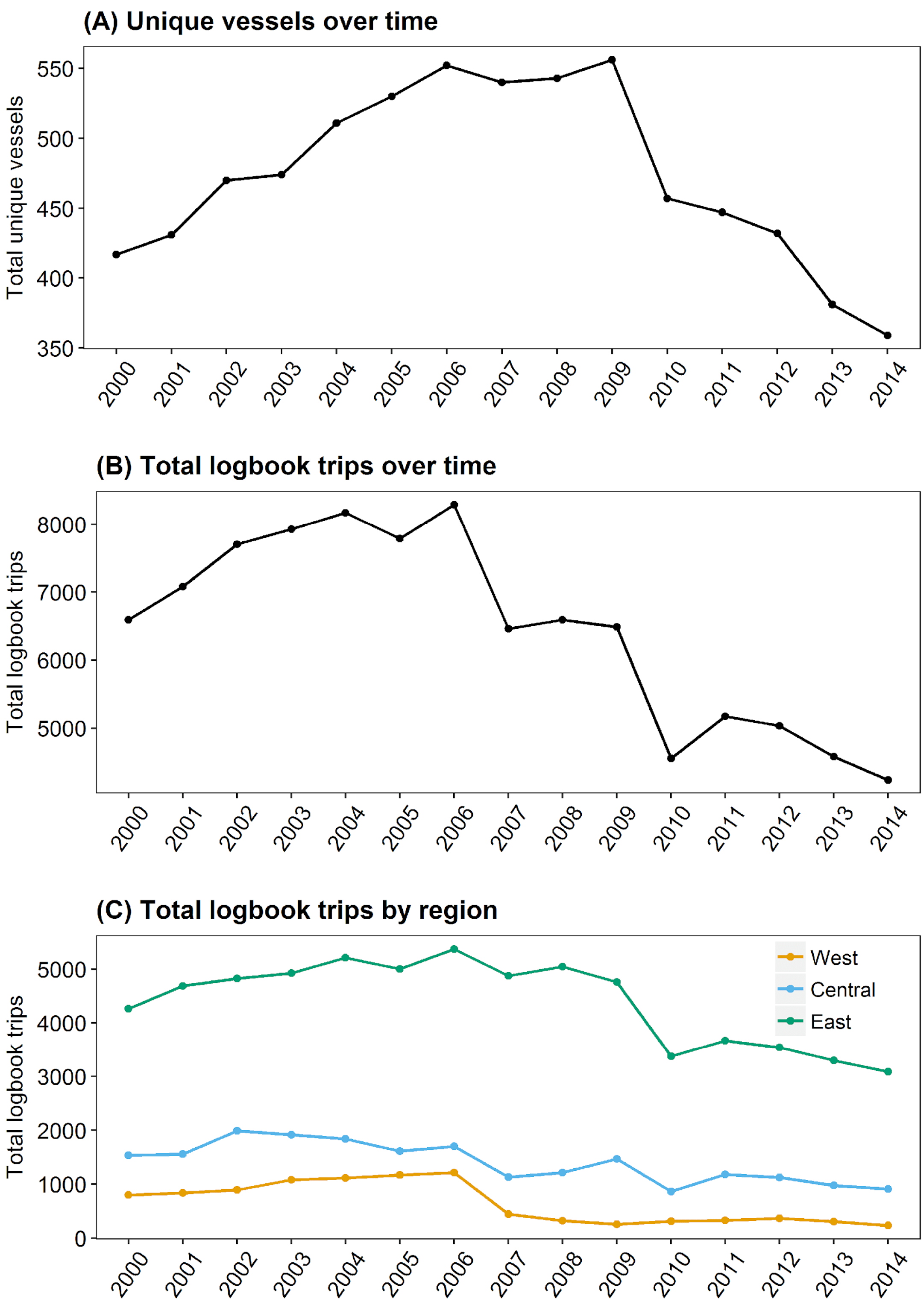


**Figure 2.2.** Number of total individual vessels and logbook trips over time. (A) Unique individual vessels in each year, (B) total logbook trips in each year of the final data set (n=96,665 trips) and (C) total logbook trips in each year of the final data set by region.

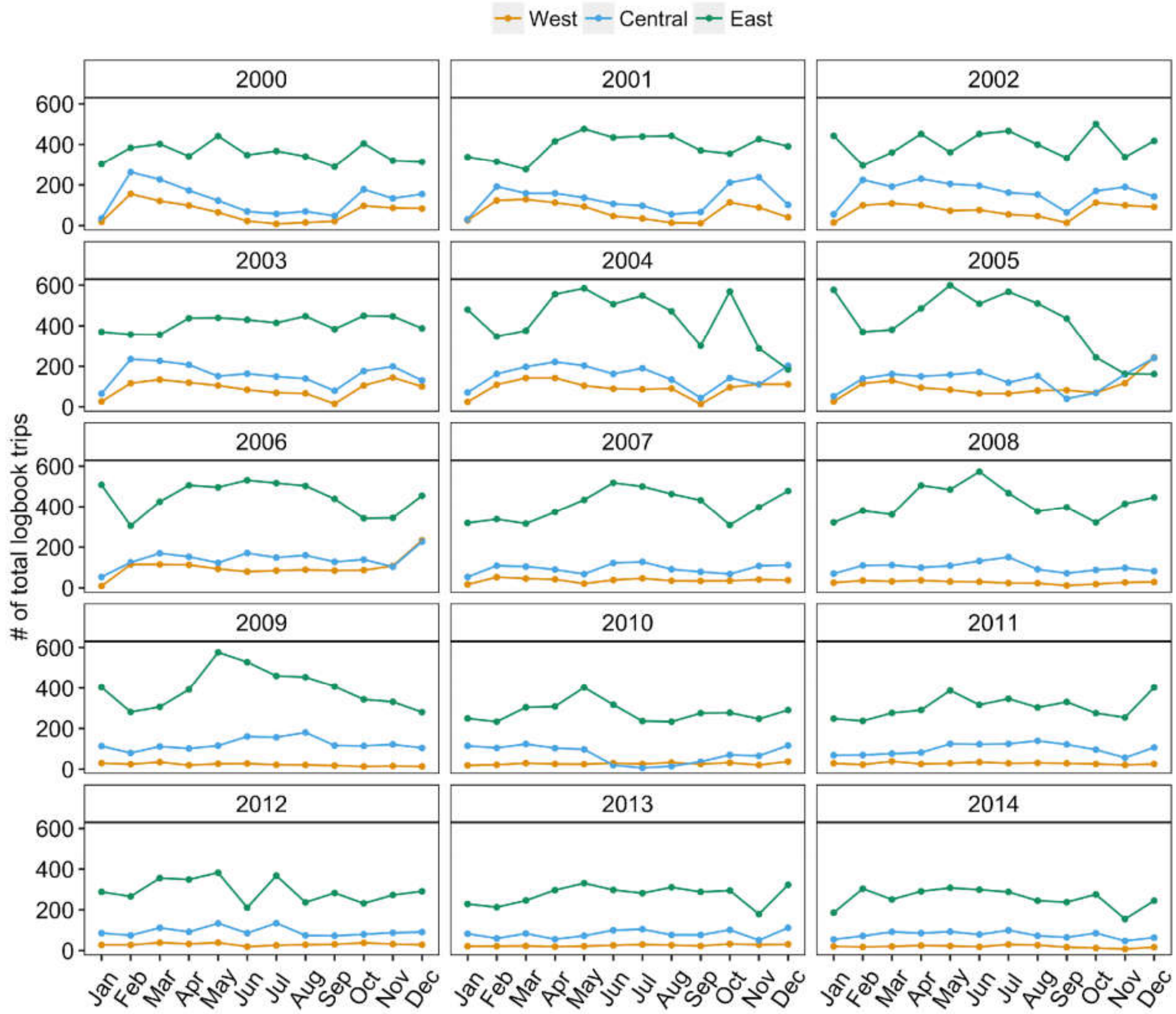


**Figure 2.3.** Monthly number of logbook trips per year in each GoM region. Data are from the final logbook trips used in analyses.

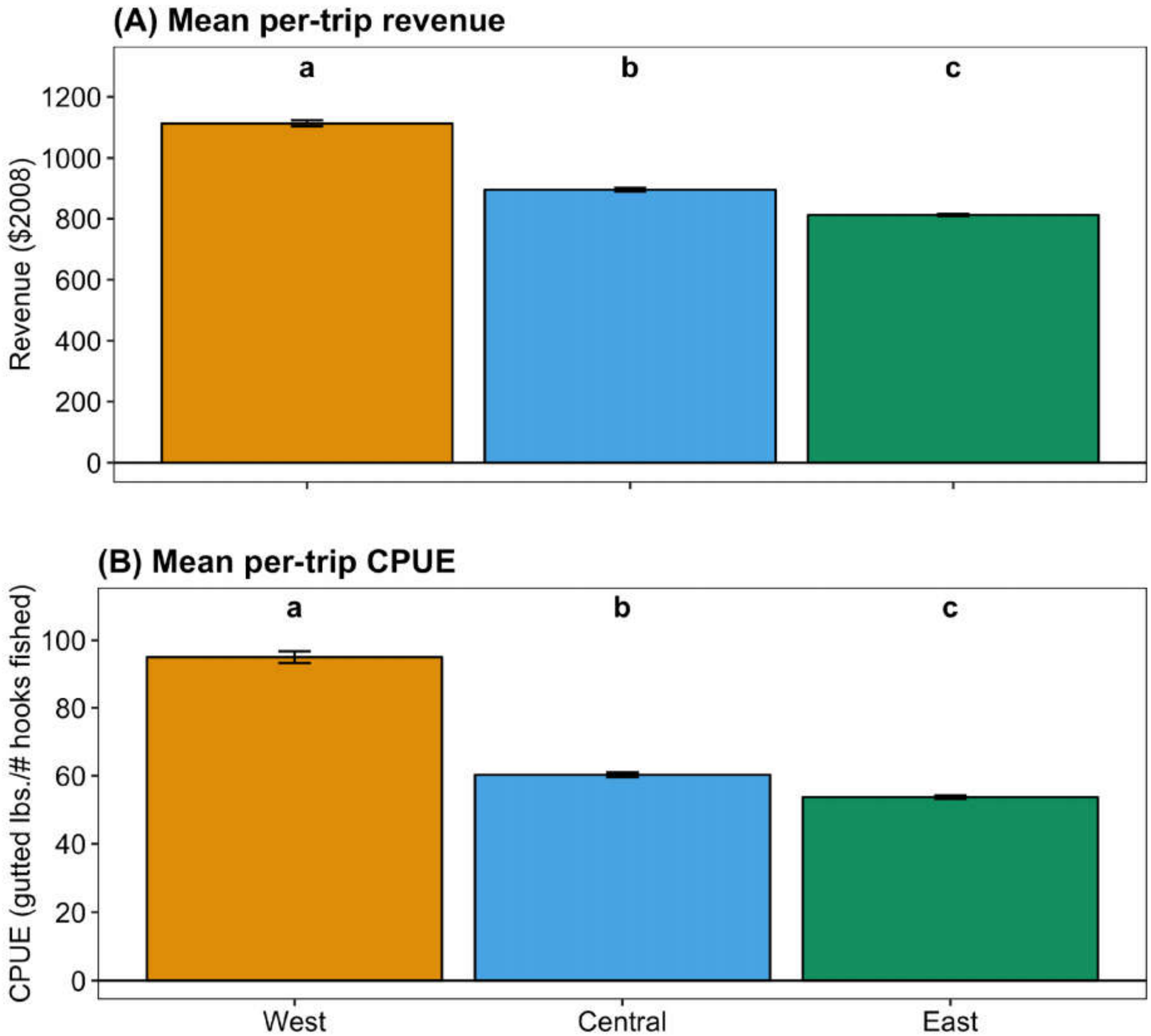


**Figure 2.4.** Mean per-trip productivity from 2000-2014 for west, central, and east regions of the GoM. Values are back-transformed means ± SEM (standard error of the mean). Different letters above the bars denote significant differences as detected from a Tukey-HSD *post-hoc* test after ANOVA.

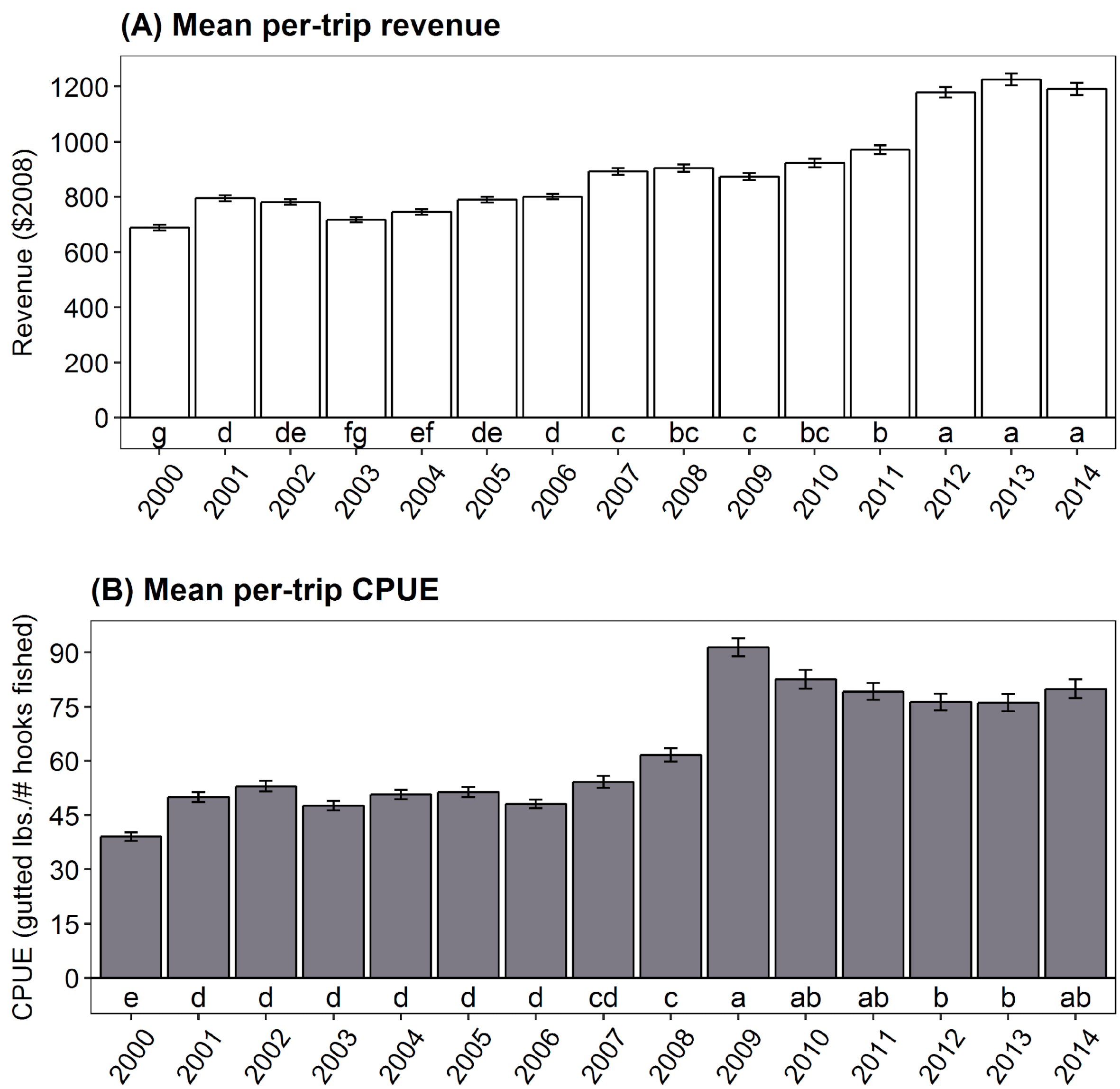


**Figure 2.5.** Mean annual per-trip productivity for the entire fleet from 2000-2014. Values are back-transformed means ± SEM. Letters below bars denote significant differences among years as detected from a Tukey-HSD *post-hoc* test after ANOVA; years with the same letter are not significantly different at the 0.05 level.

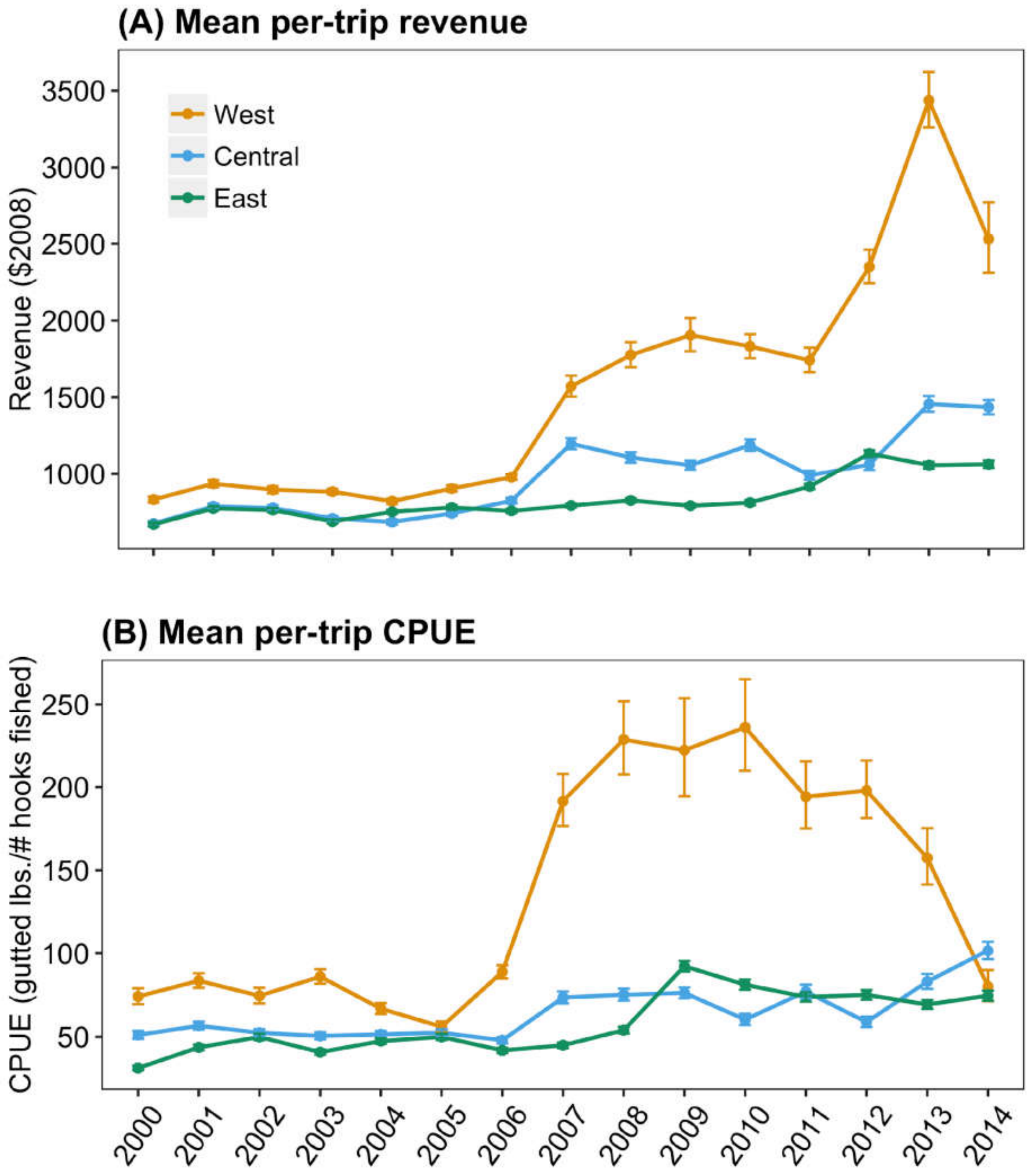


**Figure 2.6.** Mean annual per-trip productivity by GoM region from 2000-2014. Values are back-transformed means ± SEM.

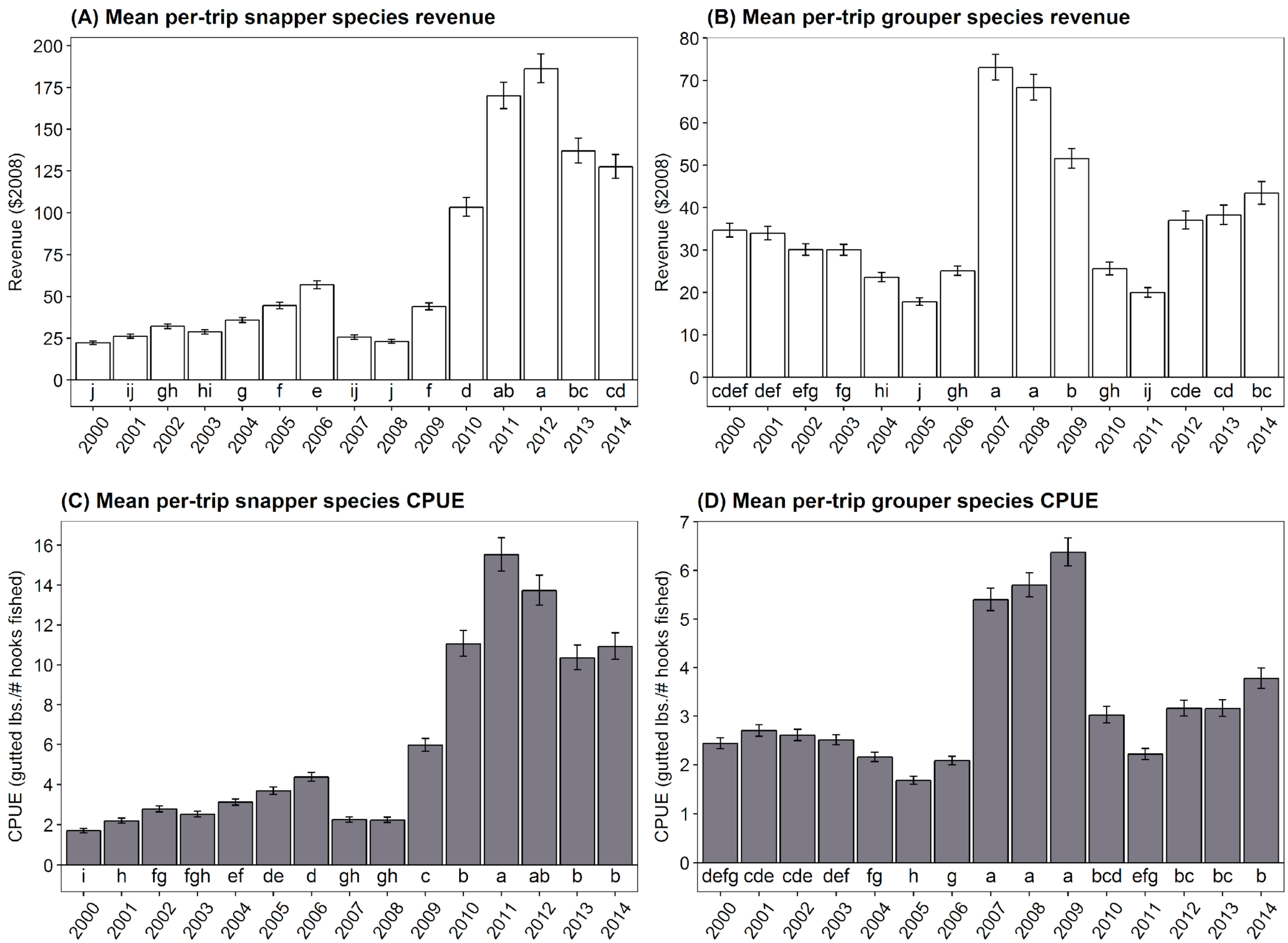


**Figure 2.7.** Mean annual per-trip CPUE and revenue of snappers (panel A, C) and groupers (panel B, D) from 2000-2014. Values are back-transformed means ± SEM. Letters below bars denote significant differences among years as detected from a Tukey-HSD *post-hoc* test after ANOVA; years with the same letter are not significantly different at the 0.05 level.

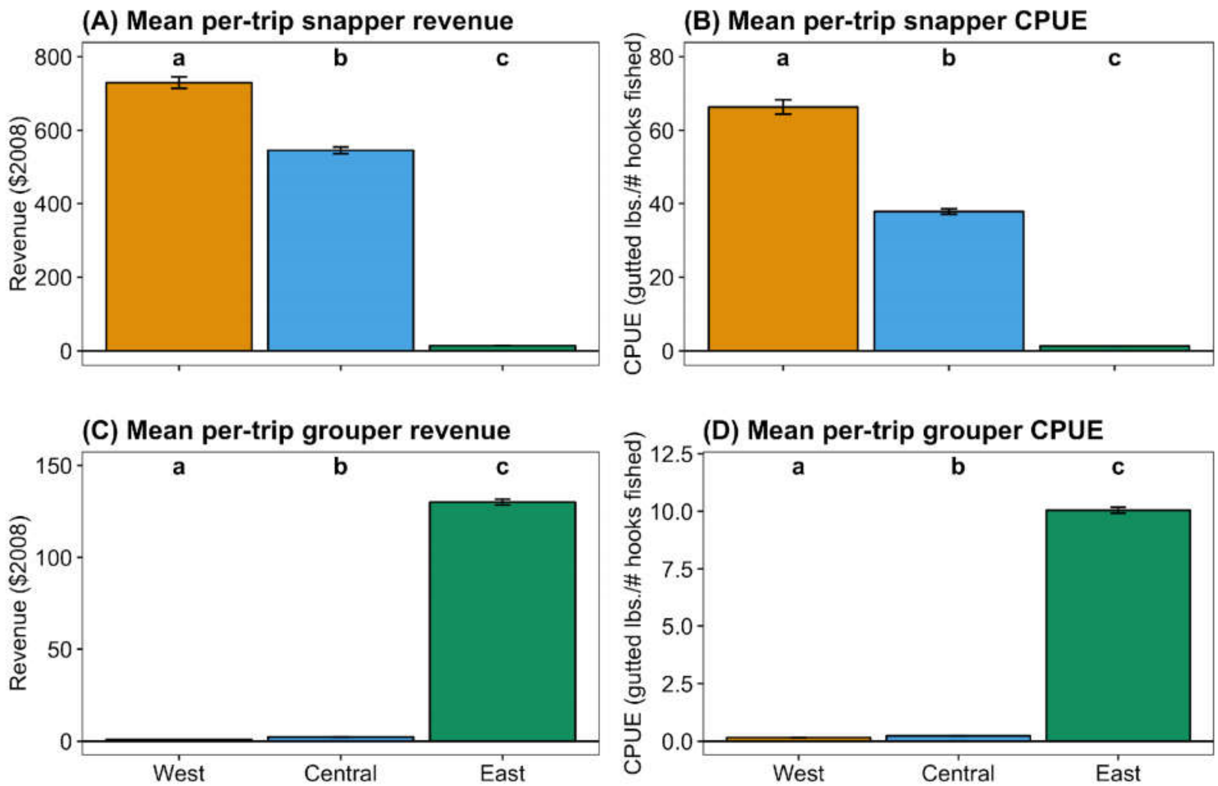


**Figure 2.8.** Mean per-trip snapper and grouper productivity from 2000-2014 for west, central, and east regions of the GoM. Values are back-transformed means ± SEM. Different letters above the bars denote significant differences as detected from a Tukey-HSD *post-hoc* test after ANOVA.

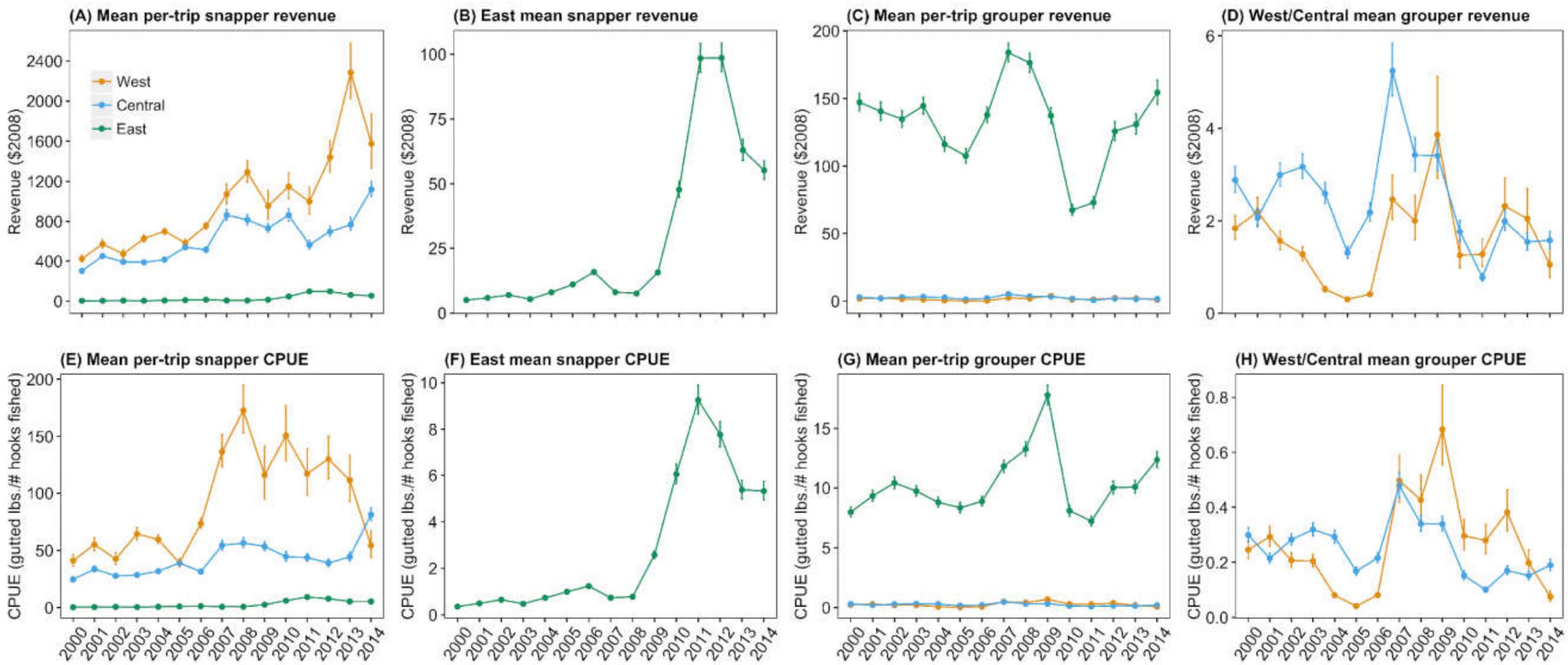


**Figure 2.9.** Mean annual per-trip snapper and grouper productivity by GoM region from 2000-2014. Values are back-transformed means ± SEM. Panels B and F show only the east region to enhance the pattern that is smoothed in panels A and B. Panels D and H show only the west and central regions to enhance the pattern that is smoothed in panels C and G. Note the change in scales across panels.

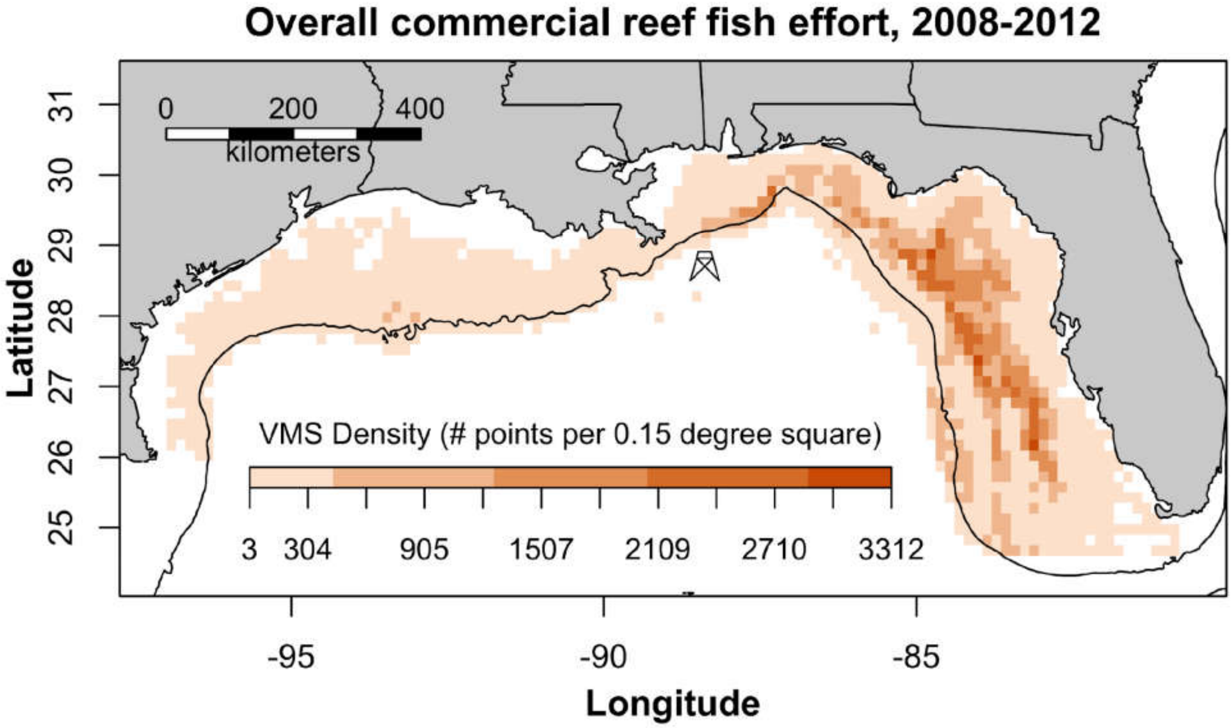


**Figure 2.10.** Distribution of fishing effort as identified by VMS points, aggregated over 2008-2012 for the entire fleet. Density was calculated as the number of VMS pings per 0.15° × 0.15° grid cell. The location of the DWH wellhead is marked with an oil derrick, and the 200-m isobath is shown with a solid black line.

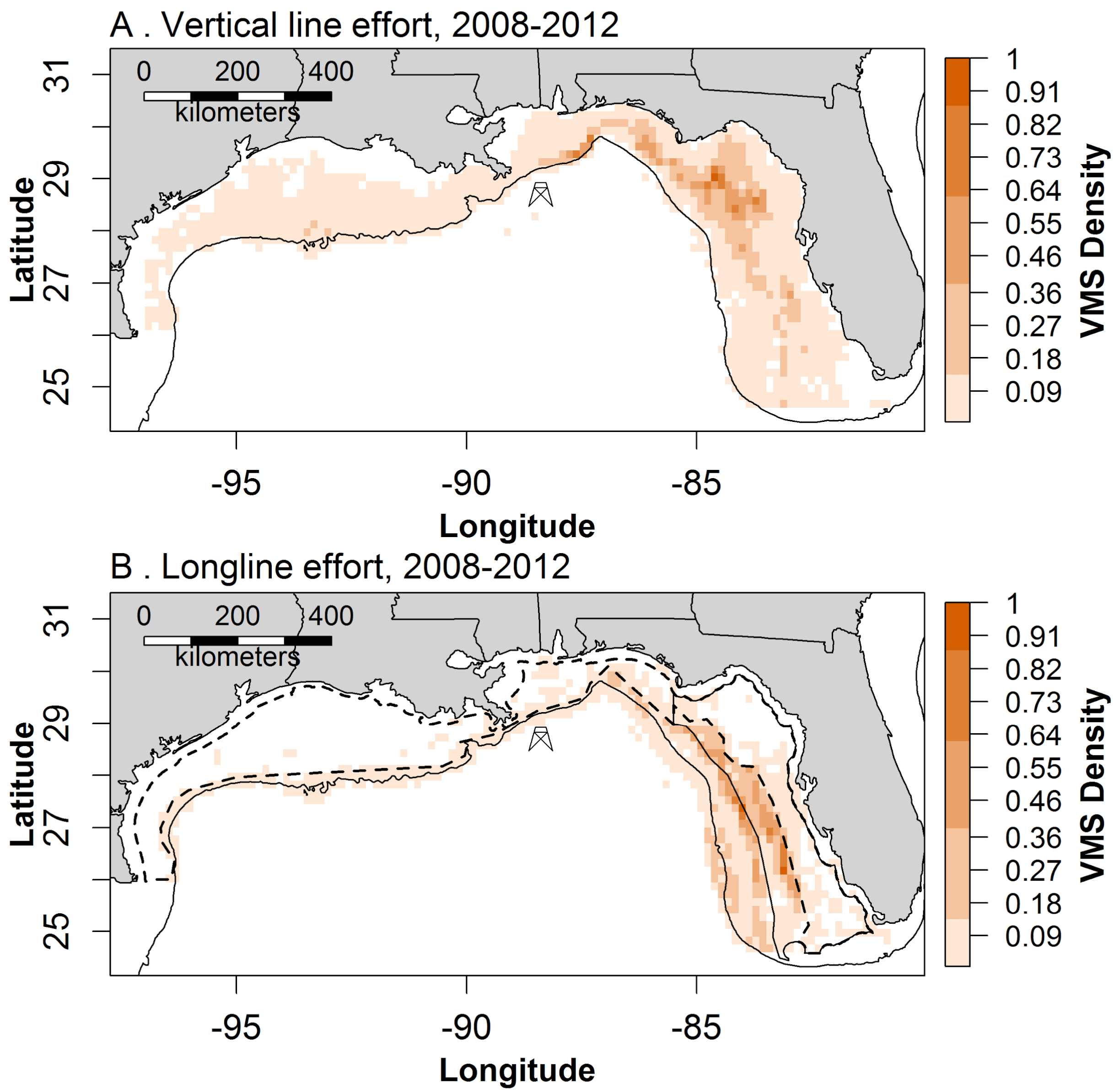


**Figure 2.11.** Distribution of fishing effort as identified by VMS points, aggregated over 2008-2012 for (A) vertical line and (B) longline gears. Density was calculated as the number of VMS pings per 0.15° × 0.15° grid cell and has been standardized to the maximum value to range from 0 (no effort) to 1 (maximum density of effort). In panel B, the dotted black line marks the year-round longline/buoy gear restricted area and the solid black line in the eastern GoM marks the seasonal bottom longline closure. Note that the restricted area and seasonal closure overlap and share an inner boundary in the eastern GoM. All other symbols are as in Figure 2.10.

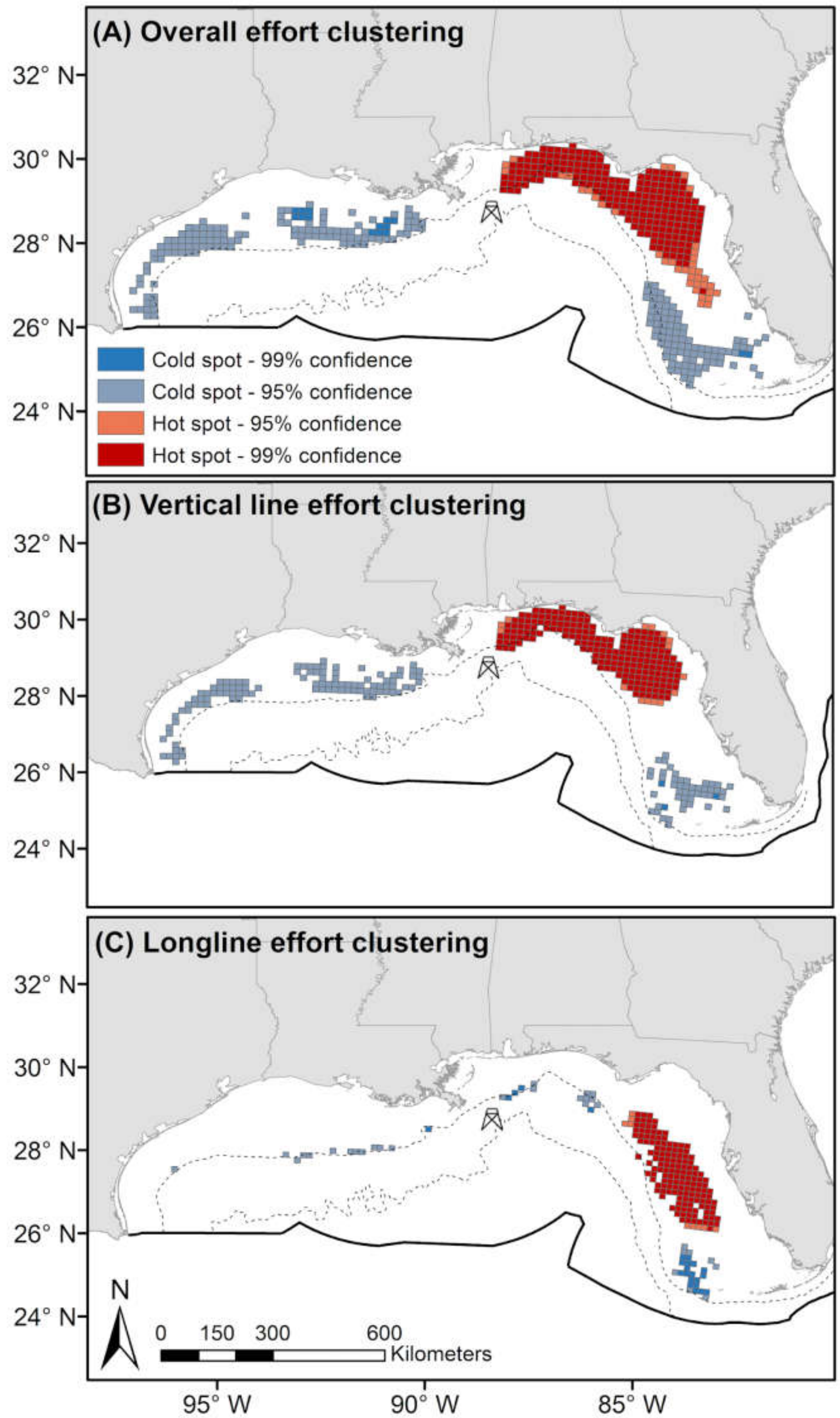


**Figure 2.12.** Significant clustering of fishing trips from optimized hot spot analysis. (A) All fleet effort aggregated over 2008-2012, (B) trips reporting vertical line as the top gear from 2008-2012, (C) trips reporting longline as the top gear from 2008-2012. Warm colors represent hot spots (regions of higher than expected clustering) and cool colors represent cold spots (regions of lower than expected clustering). All clusters shown are significant at an alpha of 0.05 (95% confidence). All symbols are as in Figure 1.1.

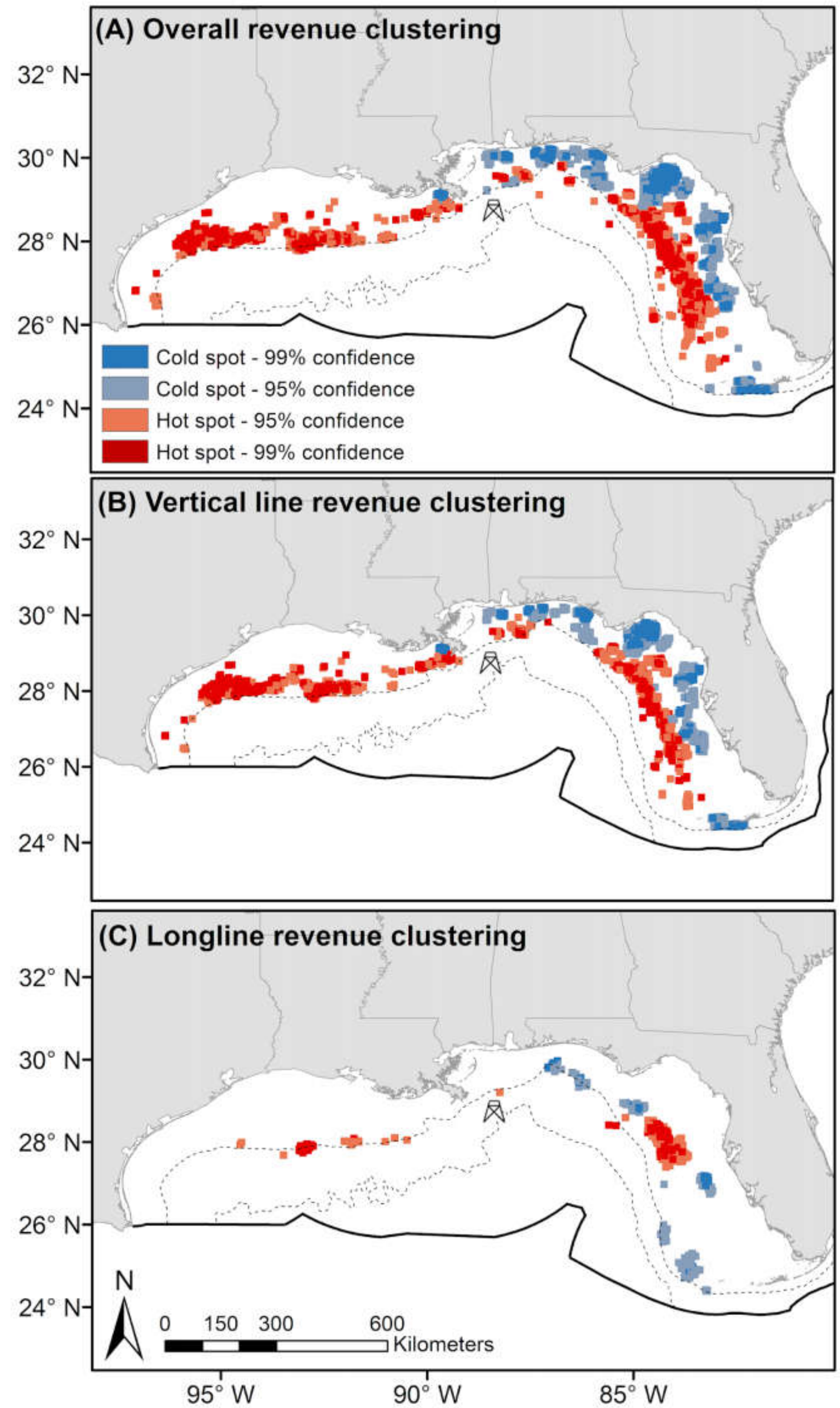


**Figure 2.13.** Significant clustering of logbook-reported revenue from optimized hot spot analysis. (A) All fleet revenue aggregated over 2008-2012, (B) trips reporting vertical line as the top gear from 2008-2012, (C) trips reporting longline as the top gear from 2008-2012. Warm colors represent hot spots (regions of higher than expected clustering) and cool colors represent cold spots (regions of lower than expected clustering). All clusters shown are significant at an alpha of 0.05 (95% confidence). All symbols are as in Figure 1.1.

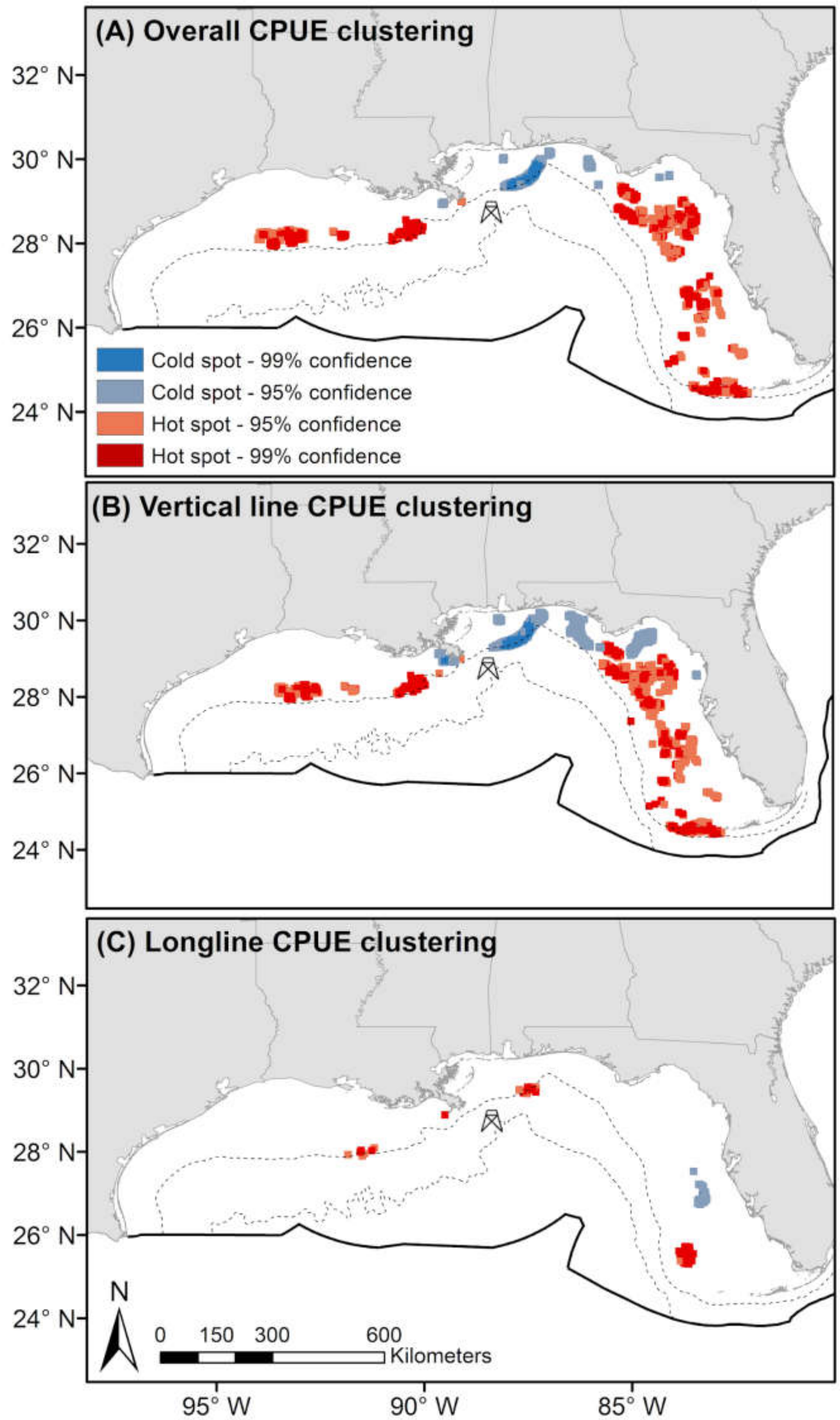


**Figure 2.14.** Significant clustering of logbook-reported CPUE from optimized hot spot analysis. (A) All fleet CPUE aggregated over 2008-2012, (B) trips reporting vertical line as the top gear from 2008-2012, (C) trips reporting longline as the top gear from 2008-2012. Warm colors represent hot spots (regions of higher than expected clustering) and cool colors represent cold spots (regions of lower than expected clustering). All clusters shown are significant at an alpha of 0.05 (95% confidence). All symbols are as in Figure 1.1.

# CHAPTER 3. IMPACTS OF CLOSURES ON FISHERS BEFORE, DURING, AND AFTER THE *DEEPWATER HORIZON* OIL SPILL

## Introduction

Significant perturbations in the availability of fisheries resources can have far reaching consequences for fishers, their families, and the communities that depend on fishing for economic sustainability (Pollnac et al. 2006, Olson 2011, Colburn and Jepson 2012). Changes in available biomass, landings, or revenue as the result of perturbations (e.g., new regulations, economic recessions, oil spills, hurricanes, stock collapse) can in turn lead to changes in income, economic efficiency or security, targeting behavior, space use, and continued sustainability of the resource. These impacts can distill into communities *via* changes in revenue streams and influence fisher job satisfaction, overall community well-being, and sustainability of fishing communities (Pollnac et al. 2006, Mascia and Claus 2009, Himes-Cornell and Kasperski 2016). The impacts from perturbations can linger over time, with ultimate outcomes depending on the ability of a community to cope with change (i.e., community resilience, as defined in Jepson and Colburn 2013). The recognition that people are integral parts of fisheries systems makes the concepts of adaptation and resilience central for designing effective restoration and ecosystem-based management (Peterson et al. 2011, Jepson and Colburn 2013). To that end, there is a growing body of research that integrates qualitative social science methods with more quantitative approaches to develop social impact assessment frameworks (Pollnac et al. 2006) and indicators of community vulnerability and resilience to disturbance (Jacob et al. 2010, Jepson and Colburn 2013).[7]

Communities can maintain or improve resilience if they can anticipate, absorb, and diffuse change over time. To ameliorate some of the effects from large perturbations, fishers can take advantage of diverse opportunities for fishing and advanced knowledge of fishery productivity when faced with environmental or regulatory change (Hackett et al. 2015, Richerson and Holland 2017). For example, Brandt (2005) noted that advanced knowledge of IFQ implementation and allocation mechanisms for Atlantic surf clams incentivized fishers to increase fishing pressure to establish a historical record of harvest. However, it is much more difficult to adapt to and absorb change when a disturbance is large and sudden. Response to sudden change may therefore be disproportionate throughout a fishery, especially if a disturbance is not equitable over space or time. What's more, if some fishers drop out (either temporally or permanently) after a disturbance, there may be indirect benefits for remaining fishers in the form of reduced competition.

This chapter quantified differences in total productivity and distribution of effort for trips inside *vs.* outside the boundaries of DWH emergency fishing closures before (January 1, 2008 through May 1, 2010), during (May 2, 2010 through November 14, 2010), and after (November 15, 2010 through December 29, 2012) the closures were in place. The location of a trip relative to the spatial extent and temporal duration of DWH closures was used as a proxy for impact and displacement due to the closures. It was expected that trips closer to the wellhead or closures would have a higher level of impact (i.e., displacement), and would therefore have reduced revenue or CPUE during the active closure period.

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[7] See also the *Gulf of Mexico and South Atlantic Fishing Community Snapshots* project from NOAA Fisheries: http://sero.nmfs.noaa.gov/sustainable_fisheries/social/community_snapshot/.

## Materials and Methods

Trip logbook and VMS data from 2008-2012 were selected as described previously (see Chapter 2 and Appendices A-D), and matched based on unique vessel identification number and a trip identifier assigned by NOAA NMFS Southeast Fisheries Science Center. The location of fishing for each trip relative to the boundaries and duration of individual DWH fishing closures was used to examine differences among trips that were inside closure boundaries ("displaced") and those that were not ("intact"). Productivity and effort distribution were quantified for trips before, during, and after closures to compare patterns across space and time relative to the potential impact of displacement from fishing closures. Each of these analyses are described in more detail below.

### *Quantifying Spatial Impact of Closures*

In order to quantify potential spatial displacement from DWH fishing closures, a metric of fishing effort relative to DWH fishing closures was calculated for each trip. A heat map of "closure proportion" for the DWH fishing closures was created in ArcGIS 10.3 by: (1) importing and overlaying the polygons of all emergency closures,[8] (2) calculating a cumulative number of days closed in each region based on the total duration of all overlapping closure polygons, and (3) dividing the cumulative days closed in each region by the total number of days closures were enforced (352 days from May 2, 2010 to April 19, 2011; Figure 3.1). Values for closure proportion therefore ranged from 0 to 1 (e.g., an area that was never closed was assigned a value of 0 and the area closed for 352 days was assigned a value of 1). VMS points were overlain onto

[8] Downloaded from the NOAA Fisheries Southeast Regional Office *Deepwater Horizon/BP Oil Spill Information* page. Available online at: http://sero.nmfs.noaa.gov/deepwater_horizon/closure_info/index.html.

the heat map, and the underlying closure proportion values were assigned to the points. A trip's mean spatial impact metric was calculated as the mean closure proportion value for all VMS points comprising the trip.

*Categorizing Trip Displacement*

The determination of displacement ("displaced" or "intact") for each trip (n=17,240) was based on the location of VMS points for the trip relative to fishing closure boundaries. If the mean spatial impact metric for a trip was greater than 0, then the location of the trip was on average within a region that was closed during DWH, and was considered to be "displaced." Individual trips were used for this determination rather than an aggregate record of individual vessels/fishers, because there were individual vessels that had records both within and outside the closed areas. Significant differences in the spatial impact metric between displaced and intact trips, trips among regions (west/central or east), and trips over time (before, during, or after closures) were determined with an analysis of variance (ANOVA). Region, time, and displacement were used as the main effects, and time × region, region × displacement, time × displacement, and time × region × displacement were used as interaction effects. Tukey-HSD *post-hoc* tests were used to examine differences among levels for significant effects. The west and central regions (Figure 2.1) were combined because the west/displaced trip replication was too low to properly perform statistical analysis (n=2).

Trips identified as displaced during closures (May 2, 2010 through November 14, 2010) were further categorized by displacement magnitude (Table 3.1). A Fisher's Natural Breaks Classification was applied to the mean spatial impact metric for displaced trips only using the *classInt* package in R (Bivand et al. 2015) to obtain three displacement magnitude categories (1 being the lowest and 3 being the greatest). Trips that were not displaced during closures were assigned a displacement magnitude of 0.

*Differences in Trip-Level Productivity*

Differences in trip-level total revenue ($2008) and CPUE (gutted lbs. landed per number of hooks fished) among displaced and intact trips over time and among regions were evaluated with a series of ANOVA tests. Displacement status (displaced or intact), region (west/central or east), time (before, during, or after closures), and the interactions of time × region, time × displacement, region × displacement, and time × region × displacement were used as model effects. Trips were assigned to a time category based on the trip start date, either as before (January 1, 2008 through May 1, 2010), during (May 2, 2010 through November 14, 2010), or after (November 15, 2010 through December 29, 2012) DWH closures. For the pre- and post-closure subsets, only vessels that also had logbook records during closures were considered. Running tests with the three time groups enabled comparisons among productivity patterns before, during, and after the DWH closures.

A series of ANOVA tests were also performed on logbook trips displaced during closures (n=544 trips). These tests examined potential differences in trip revenue and CPUE among trips with fishing locations inside closure boundaries based on region (west/central or east) and displacement magnitude (1, 2, or 3 as described above). Region, displacement magnitude, and region × displacement magnitude were used as model effects. All ANOVA tests were performed on $\log_e$-transformed data, at an alpha of 0.05 and Tukey-HSD *post-hoc* tests were used to examine significant differences among levels.

*Fishing Activity Relative to Displacement*

The spatial distribution of fishing activity for displaced and intact trips before, during, and after closures was quantified by mapping the density of VMS pings per 0.15° × 0.15° grid cell, using the *raster* package in R (Hijmans et al. 2016). All VMS points identified as active fishing (see Appendices D-F) were used. Grid cells with less than three VMS pings were reassigned as "NA" and not mapped, to ensure confidentiality of the data.

Shifts in distribution of effort before *vs.* after closures for displaced and intact trips were examined by comparing standardized relative density for each group pre- and post-closure. Effort density values were standardized (ranging from 0 to 1) relative to the maximum. Differences in effort distribution (based on standardized effort density) were then calculated as the difference between individual density layers. A difference of 0 indicated no change in relative density, or complete overlap in distribution, while a value of 1 or -1 indicated maximum difference, or no overlap in spatial distribution. A spatial difference index was additionally calculated (Lee et al. 2010; see Chapter 2) to quantify differences (and percent similarity) among effort distributions for displaced and intact trips before, during, and after DWH fishing closures.

## Results

*Categorizing Trip Displacement*

There were more logbook trips and more individual vessels in the east before, during, and after closures than in the west/central region (Table 3.2). During closures, 72% of logbook trips were identified as outside the closure boundaries (intact) and 28% were identified as inside closure boundaries (displaced). The trips were made by 344 individual vessels, of which 186 (54%) did not fish within a closure boundary at all, while 158 (46%) had at least some portion of their record during closures within a closure boundary (i.e., mean spatial impact metric over all trips was > 0). Before and after closures, a greater percentage of all trips were *within* closure boundaries (43% and 42%, respectively) and more individual vessels made trips to locations *inside* a closure boundary (63% and 60%, respectively; Table 3.2). That is, a greater proportion of fishing trips were outside the closure boundaries during closures than either before or after. This pattern was observed in both regions, although the difference in proportion of displaced trips was greater for the west/central region (20-24% decrease in trips within closure boundaries during closures) than for the east region (7-9% decrease in trips). The mean spatial impact metric in the west/central region was significantly lower for trips during closures than either before or after closures (Tukey-HSD, $p<0.001$ for all comparisons), but there were no significant differences in spatial impact metric among times in the east (Tukey-HSD, $p>0.06$ for all comparisons; Figure 3.2).

Moreover, there was a greater percentage of trips located within closure boundaries and a greater percentage of vessels fishing within closure boundaries for a portion of their record in the west/central region across all time periods (Table 3.2). The mean spatial impact metric was also significantly greater for trips in the west/central region than in the east during all time periods (Tukey-HSD, $p<0.001$ for all comparisons), indicating more total fishing activity within closure boundaries in the west/central region (Table 3.3, Figure 3.2). For trips within closure boundaries during closures (i.e., excluding intact trips), the mean spatial impact (± standard error) was significantly lower in the east region ($0.10 \pm 0.004$ compared to $0.27 \pm 0.007$; ANOVA, $p<0.001$).

*Differences in Trip-Level Productivity*

The effects of time, region, time × region, and region × displacement were significant for trip revenue while the effects of region, displacement, time × displacement, region × displacement, and the three-way interaction of time × region × displacement were significant for trip CPUE (Table 3.4). Tukey-HSD tests revealed that revenue was higher overall after closures than either before or during closures ($p<0.001$), CPUE and revenue were higher overall in the west/central region ($p<0.001$), and CPUE was significantly higher for intact trips, both overall ($p<0.001$) and within regions ($p<0.001$ for all comparisons). Revenue was significantly different between displaced and intact trips but the pattern was reversed between the two regions; in the west/central, revenue was significantly higher for intact trips ($p<0.001$) while in the east revenue was significantly higher for displaced trips ($p<0.001$; Figure 3.3).

Intra-regional patterns of revenue and CPUE between displaced and intact trips were consistent over time (Figure 3.3). In the west/central region, revenue and CPUE were both significantly lower for displaced trips in every time period ($p<0.01$ for all comparisons). Additionally, revenue and CPUE were not significantly different between time periods for displaced or intact fishers in the west/central ($p>0.97$ for all comparisons). In the east region, CPUE was significantly lower for displaced trips in every time period ($p<0.001$ for all comparisons) while revenue was significantly higher for displaced trips before and after closures ($p<0.001$ for both comparisons). Revenue was higher for displaced trips in the east during closures, but the difference was not statistically significant ($p=0.12$). Both displaced and intact trips had significantly greater revenue after closures than either before or during ($p<0.001$ for all comparisons), and displaced trips had significantly lower CPUE before closures than either during ($p=0.03$) or after ($p<0.001$).

Considering only displaced trips during DWH closures, only the effect of region was significant for revenue while region and region × displacement magnitude were significant for CPUE (Table 3.5). West/central revenue was significantly higher overall than east revenue, regardless of displacement magnitude (Tukey-HSD, $p=0.02$; Figure 3.4). West/central CPUE was also higher overall than in the east ($p<0.001$), although regional differences were only significant between the magnitude '2' groups ($p<0.001$).

*Fishing Activity Relative to Displacement*

The distribution of effort (measured as the density of VMS points) was different for displaced and intact trips. Before, during, and after closures, intact trips (i.e., those not fishing inside closure boundaries), were distributed around the periphery of the GoM (Texas and Florida), with the highest effort densities located in the Florida Big Bend and near- to mid-shore on the WFS (Figures 3.5-3.7). Displaced trip effort (i.e., those fishing inside closure boundaries) was very nearly opposite of this pattern. Displaced trip effort was located primarily in the central GoM, offshore of Louisiana, Alabama, and Mississippi, as well as the panhandle of Florida and mid- to far-offshore on the WFS. Given the definition of intact and displaced trips for this analysis (i.e., inside or outside closure boundaries, respectively), and the spatial extent of the fishing closures, these effort distributions follows what would be expected.

The density of effort overall was lowest during closures and lower after closures as compared to before for both displaced and intact trips (compare the range of density values in Figures 3.5-3.7). On the whole, effort throughout the GoM was more contracted and patchier during closures than either before or after. The greatest density of effort for intact trips was in the Big Bend before closures (Figure 3.5), shifted southward and inshore during closures (Figure

3.6), and then shifted back offshore after closures, with higher relative densities on the southern WFS (Figure 3.7). The greatest effort density for displaced trips was offshore of the Alabama coast (slightly northeast of the DWH wellhead) and mid-peninsular Florida before and after closures (Figures 3.5 and 3.7), but was shifted into the Florida panhandle and southern WFS during closures with no appreciable effort near the wellhead (Figure 3.6).

The most prominent shifts in effort distribution pre- to post-closures were seen in the eastern GoM for intact trips and in the central and eastern GoM for displaced trips (Figure 3.8). For intact trips, effort generally shifted south and west from the Big Bend and inshore WFS to more offshore locations along peninsular Florida. The greatest decrease in relative effort density outside of closure boundaries (-0.42) was along the Florida panhandle (black circle in Figure 3.8A) and the greatest increase (0.47) was on the WFS, offshore of Tampa Bay (black diamond in Figure 3.8A). For displaced trips, effort shifted southward along the mid-WFS and north-northeast closer to the Florida coast in the central GoM. The greatest decrease in relative effort density (-0.47) was offshore of Alabama (black circle in Figure 3.8B), just northeast of the wellhead, where the greatest density of effort was located before closures. The greatest increase (0.54) was on the mid-WFS, only slightly south of where the greatest increase was for intact trips (black diamond in Figure 3.8B).

The similarity in effort distribution for displaced *vs.* intact fishers was low before, during, and after closures (22-26%; Table 3.6), but there were slightly fewer vessels overall in the fleet after closures (Table 3.2). On the other hand, the inter-group similarity in effort distribution among times was moderately high. Comparing effort densities before and after closures, trips within closure boundaries were 76% similar and trips outside of closure boundaries were 78%

similar. Comparisons with trips that occurred during closures were slightly lower: 58-63% for trips inside closure boundaries and 70-75% for trips outside closure boundaries.

**Discussion**

Results indicate that fishers adapted and were largely resilient to DWH fishing closures, although there were some regional differences in the impact and response. Fishing grounds for trips in the west/central region were more heavily impacted by closures (as quantified by the spatial impact metric) than trips in the east. There was also a greater percentage of trips and vessels inside closure boundaries in the western GoM over all time periods. This result is not unexpected, given the location of the DWH wellhead and the known extent of the various closures. The total number and proportion of trips and vessels inside closure boundaries decreased during DWH in both regions, with a greater magnitude of change in the west (i.e., where impact from displacement was greater). Yet, revenue and CPUE were consistently higher in the west/central GoM than in the east, regardless of impact from closures. As was discussed in Chapter 2, this is likely due in part to the magnitude of snapper landings that made up a majority of catch in western GoM trips. Among trips displaced during DWH, displacement magnitude was not a significant factor affecting productivity.

Contrary to expectations, productivity was not adversely affected during DWH closures for trips either inside or outside closures. Productivity remained the same across time periods for both intact and displaced trips in the west and increases were seen for productivity during and after closures in the east. The pattern of productivity between trips inside and outside closure boundaries remained the same over time: generally higher for trips outside boundaries, with the exception of higher revenue for trips inside boundaries in the east. It is possible that the constant and even increased productivity for trips was the result of reduced competition during and after closures. That is, there were fewer vessels in the fleet during and after closures, especially inside closure boundaries, and overall lower density after closures (compare the maximum density values in Figures 3.5 and 3.7). The fleet conditions during and after closures thus could have resulted in reduced competition and increased landings and CPUE.

Similarity in effort distribution was greatest within displacement category before *vs.* after closures. This implies that fishers were generally returning to pre-DWH fishing grounds, regardless of whether trips were first inside or outside closure boundaries. On the other hand, there was evidence of shifting fishing grounds beyond DWH. For intact trips, maximum density during closures shifted southward along the WFS, and remained there even after closures were removed. Although there was lower density overall during closures, the shift in effort distribution for these fishers (who were not fishing directly inside closures, and may have moved simply to avoid being near the DWH disaster as it was occurring) suggests that there may have been a more productive or economically efficient fishing ground along the WFS than was being used previously. Displaced fishers similarly displayed a shift in effort away from pre-closure fishing grounds northeast of the wellhead, although the pattern was less distinct than for intact trips. Profitability and economic incentives are a primary determinant of fishery participation and location choice (Smith and Wilen 2003), making it likely that fishers were concentrating effort pre- and post-DWH in such a way that optimized profitability, perhaps in this new fishing ground on the WFS.

However, profitability varies based on species movement patterns, level of competition, and vessel capacity (Dowling et al. 2012). Therefore, the ability of fishers to be successful during and after a large-scale disturbance will also depend on the diversity of the fishing portfolio (i.e., participation in a variety of fisheries) and other underlying economic factors for individual fishers. For example, Hackett et al. (2015) found that greater diversity of fishing income and lower interannual variability in fishing income were consistent predictors of remaining in a California commercial fishery after a series of regulatory changes that reduced access to fishery resources. There is also likely variation among captains in the decision-making process (e.g., based on level of experience or familiarity with alternative fishing grounds) or variable decisions depending upon environmental, management, and economic conditions (e.g., weather, remaining quota for the target species, in-season species, market conditions, and perceived profitability of fishing location; Sanchirico and Wilen 2001, Smith and Wilen 2003). Further investigation of the relationship between species composition, the makeup of vessels in the fleet, and profitability in this region would help elucidate the mechanisms driving increased effort in this region during and after DWH.

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**Table 3.1.** Intervals of mean spatial impact metric used to categorize trips during DWH closures into displacement magnitude categories.

| Mean trip spatial impact metric | Displacement magnitude category | # of trips during DWH closures |
|---:|---:|---:|
| 0 | 0 | 1,379 |
| [0.0002,0.09] | 1 | 161 |
| (0.09,0.22] | 2 | 222 |
| (0.22,0.44] | 3 | 161 |

**Note:** All intervals were closed on the right.

**Table 3.2.** Logbook trips and vessels identified as displaced or intact overall and by region before, during, and after DWH fishing closures.

| Region | Trip start relative to closures | Total logbook trips | Total vessels | Trips inside a closure boundary (displaced) | | Trips outside a closure boundary (intact) | | Vessels with no fishing records inside a closure boundary | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | N | % | N | % | N | % |
| Combined | Before | 8,852 | 325 | 3,812 | 43 | 5,040 | 57 | 121 | 37 |
| | During | 1,923 | 344 | 544 | 28 | 1,379 | 72 | 186 | 54 |
| | After | 6,465 | 309 | 2,683 | 42 | 3,782 | 58 | 124 | 40 |
| West/Central | Before | 2,484 | 96 | 1,877 | 76 | 607 | 24 | 20 | 21 |
| | During | 354 | 91 | 184 | 52 | 170 | 48 | 29 | 32 |
| | After | 1,529 | 92 | 1,096 | 72 | 433 | 28 | 22 | 24 |
| East | Before | 6,368 | 253 | 1,935 | 30 | 4,433 | 70 | 102 | 40 |
| | During | 1,569 | 262 | 360 | 23 | 1,209 | 77 | 159 | 61 |
| | After | 4,936 | 236 | 1,587 | 32 | 3,349 | 68 | 103 | 44 |

**Table 3.3.** Results from ANOVA on differences in spatial impact metric among GoM regions and time periods relative to DWH fishing closures.

| Effect | d.f. | SS | MS | F | P |
|---|---|---|---|---|---|
| Time | 2 | 4.6 | 2.3 | 181.5 | < 0.001 |
| Region | 1 | 160.3 | 160.3 | 12,621.7 | < 0.001 |
| Time × Region | 2 | 4.2 | 2.1 | 165.4 | < 0.001 |
| Residual | 17,234 | 218.9 | 0.01 | | |

**Table 3.4.** Results from ANOVA tests on per-trip productivity among GoM regions, time periods relative to DWH fishing closures, and displacement category relative to fishing closure boundaries.

| Response | Effect | d.f. | SS | MS | F | P |
|---|---|---|---|---|---|---|
| ln(revenue) | Time | 2 | 216 | 108.1 | 110.9 | < 0.001 |
| | Region | 1 | 271 | 270.6 | 277.4 | < 0.001 |
| | Displacement | 1 | 0.4 | 0.4 | 0.4 | 0.6 |
| | Time × Region | 2 | 58 | 29.1 | 29.8 | < 0.001 |
| | Time × Displacement | 2 | 0.4 | 0.2 | 0.2 | 0.8 |
| | Region × Displacement | 1 | 383 | 382.6 | 392.2 | < 0.001 |
| | Time × Region × Displacement | 2 | 2 | 1.1 | 1.1 | 0.3 |
| | Residual | 17,228 | 16,803 | 1 | | |
| | | d.f. | SS | MS | F | P |
| ln(CPUE) | Time | 2 | 16 | 8 | 1.7 | 0.2 |
| | Region | 1 | 57 | 57 | 12.0 | < 0.001 |
| | Displacement | 1 | 6,659 | 6,659 | 1,406 | < 0.001 |
| | Time × Region | 2 | 9 | 4 | 0.9 | 0.4 |
| | Time × Displacement | 2 | 215 | 107 | 22.6 | < 0.001 |
| | Region × Displacement | 1 | 34 | 34 | 7.2 | < 0.01 |
| | Time × Region × Displacement | 2 | 80 | 40 | 8.4 | < 0.001 |
| | Residual | 17,228 | 81,593 | 5 | | |

**Table 3.5.** Results from ANOVA tests on per-trip productivity among GoM regions and magnitude of displacement for trips displaced during DWH closures.

| Response | Effect | d.f. | SS | MS | F | P |
|---|---|---|---|---|---|---|
| ln(revenue) | Displacement magnitude | 2 | 4.8 | 2.4 | 2.8 | 0.06 |
| | Region | 1 | 9.6 | 9.6 | 11.1 | 0.001 |
| | Region × Displacement magnitude | 2 | 1.6 | 0.8 | 0.9 | 0.4 |
| | Residual | 538 | 462.7 | 0.9 | | |
| | | d.f. | SS | MS | F | P |
| ln(CPUE) | Displacement magnitude | 2 | 19.5 | 9.76 | 2.3 | 0.10 |
| | Region | 1 | 62.2 | 62.2 | 14.7 | < 0.001 |
| | Region × Displacement magnitude | 2 | 28.4 | 14.2 | 3.4 | 0.04 |
| | Residual | 538 | 2,271.1 | 4.2 | | |

**Table 3.6.** Spatial difference metric and percent similarity in effort distribution between displaced and intact trips before, during, and after DWH closures.

| Comparison | Spatial difference metric | % similarity |
|---|---|---|
| Displaced-intact before | 0.76 | 24 |
| Displaced-intact during | 0.78 | 22 |
| Displaced-intact after | 0.74 | 26 |
| Displaced before-after | 0.24 | 76 |
| Intact before-after | 0.22 | 78 |
| Displaced during-after | 0.38 | 63 |
| Intact during-after | 0.31 | 70 |
| Displaced during-before | 0.42 | 58 |
| Intact during-before | 0.25 | 75 |

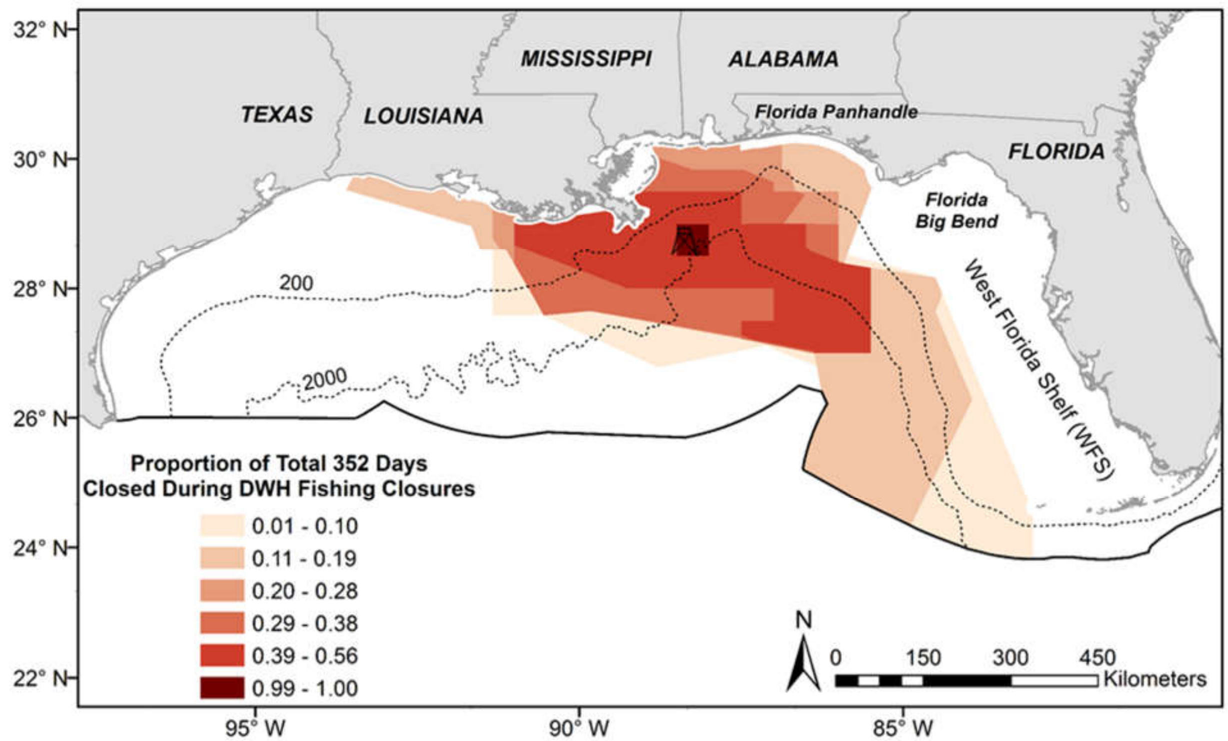


**Figure 3.1.** Cumulative days closed, expressed as a proportion of total days closed for each emergency closure region during the *Deepwater Horizon* oil spill. Darker colors represent a larger proportion of cumulative days closed. Symbols are as in Figure 1.1.

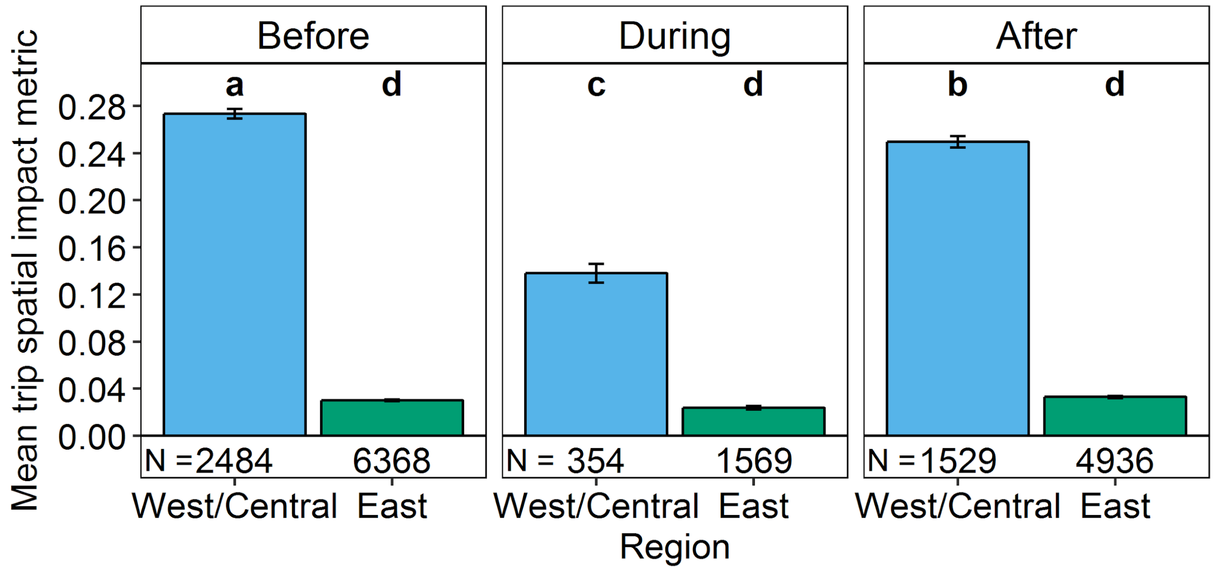


**Figure 3.2.** Mean trip-level spatial impact metric by GoM region and time relative to DWH fishing closures. Values were calculated based on location of VMS points relative to closure boundaries for trips before, during, and after DWH closures. The number of trips (N) in each region/time combination are given below the bars. Different letters above the bars denote significant differences as detected from a Tukey-HSD *post-hoc* test after ANOVA.

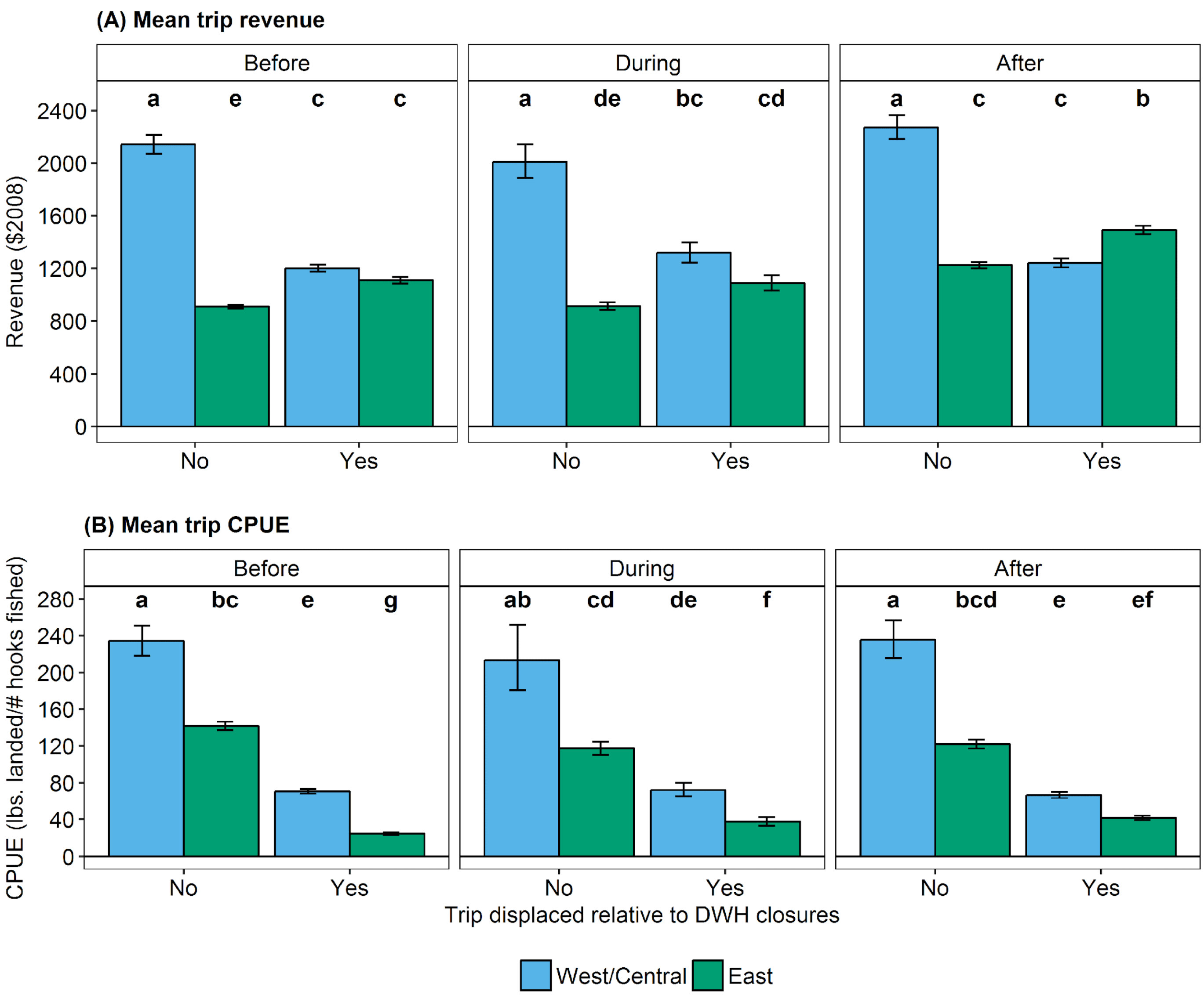


**Figure 3.3.** Mean trip-level productivity before, during, and after DWH closures for displaced and intact trips by GoM region. Values are back-transformed means ± SEM. Different letters above the bars denote significant differences as detected from a Tukey-HSD *post-hoc* test after ANOVA.

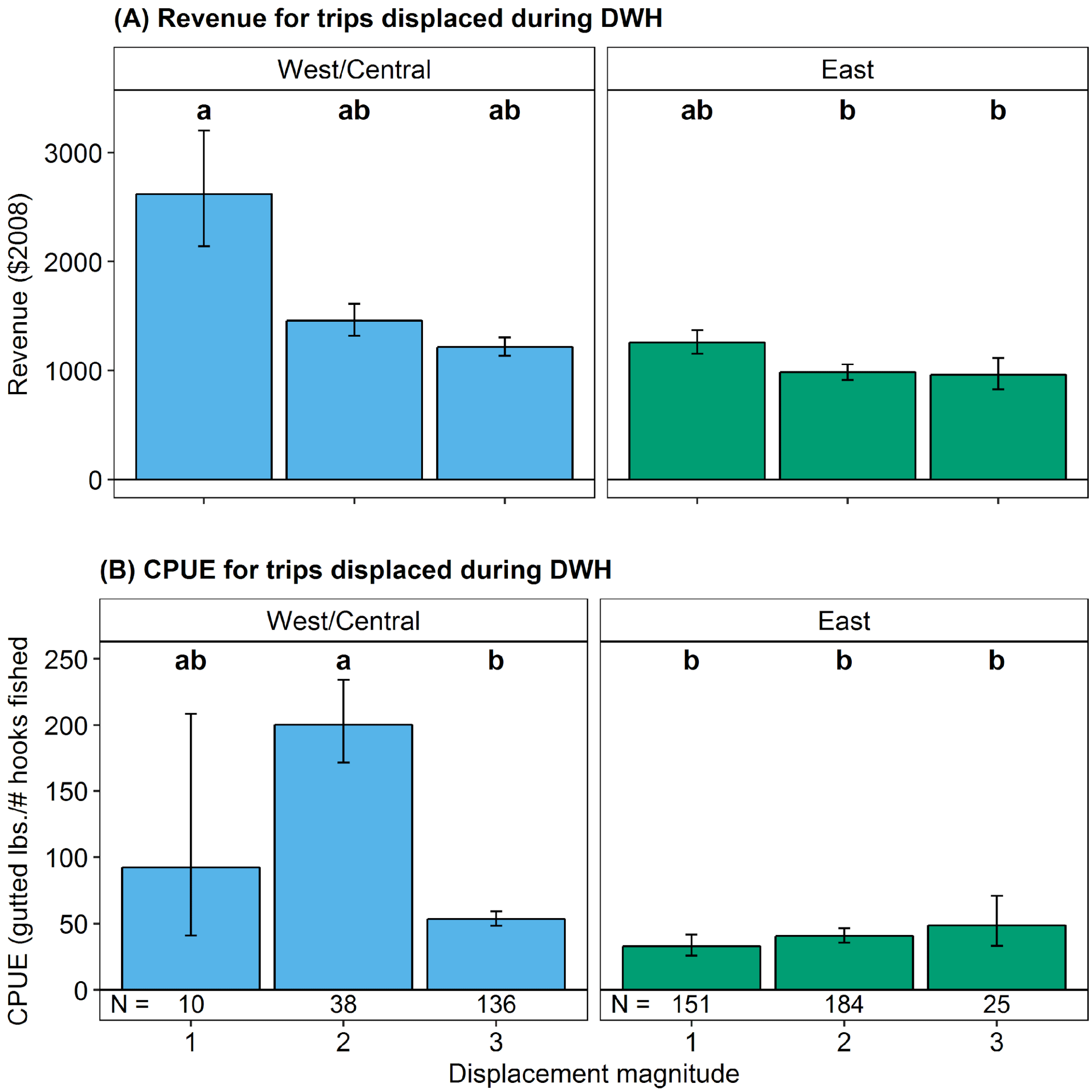


**Figure 3.4.** Mean trip-level productivity for trips displaced during DWH fishing closures. Values are back-transformed means ± SEM. Different letters above the bars denote significant differences as detected from a Tukey-HSD *post-hoc* test after ANOVA. Numbers below the bars indicate the number of replicates (N).

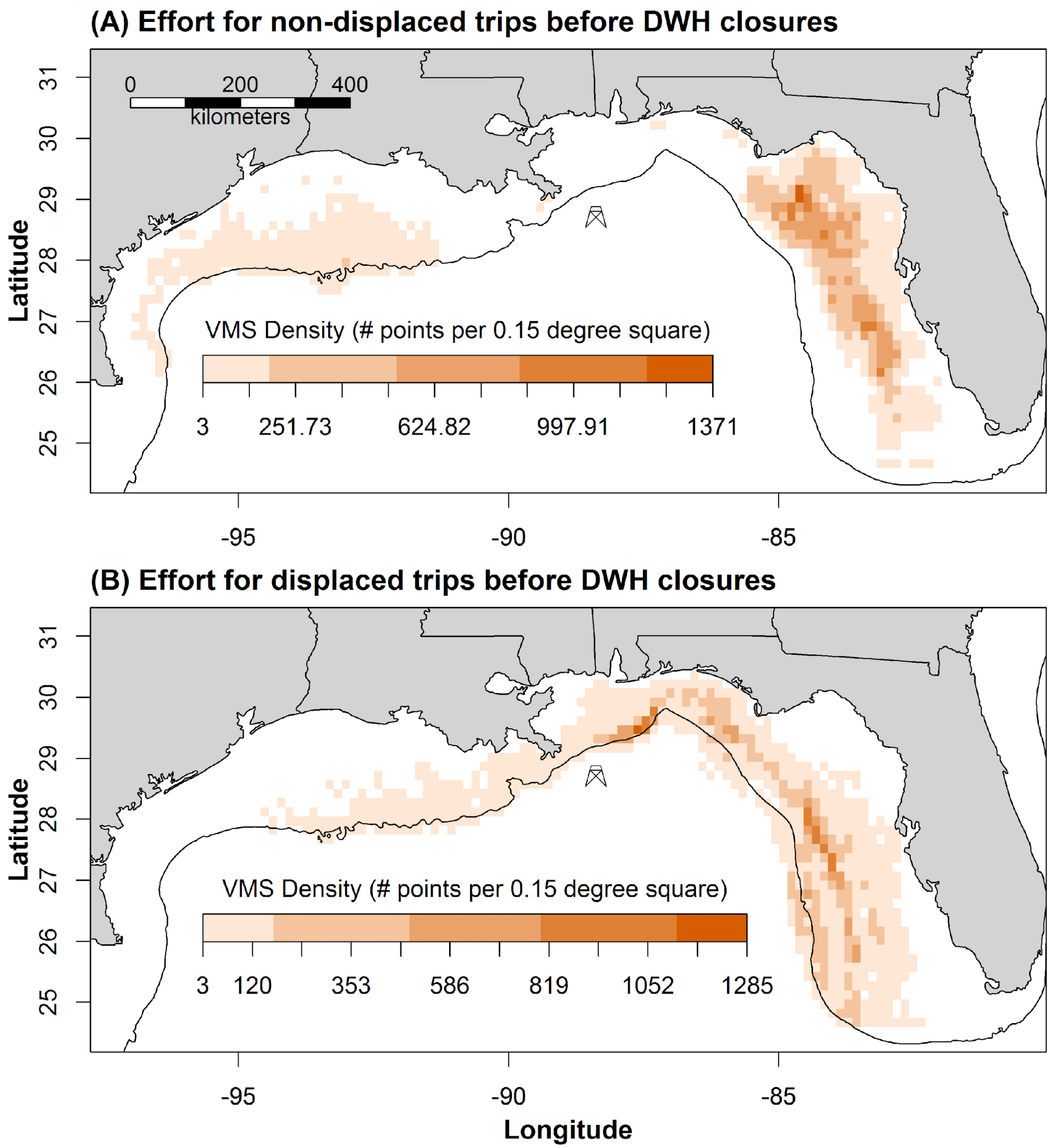


**Figure 3.5.** Density of fishing effort for 2008-2010 trips aggregated before DWH emergency fishing closures. (A) Trips outside of closure boundaries ("intact") and (B) trips inside closures boundaries ("displaced"). Density was calculated as the number of VMS pings per 0.15° × 0.15° grid cell.

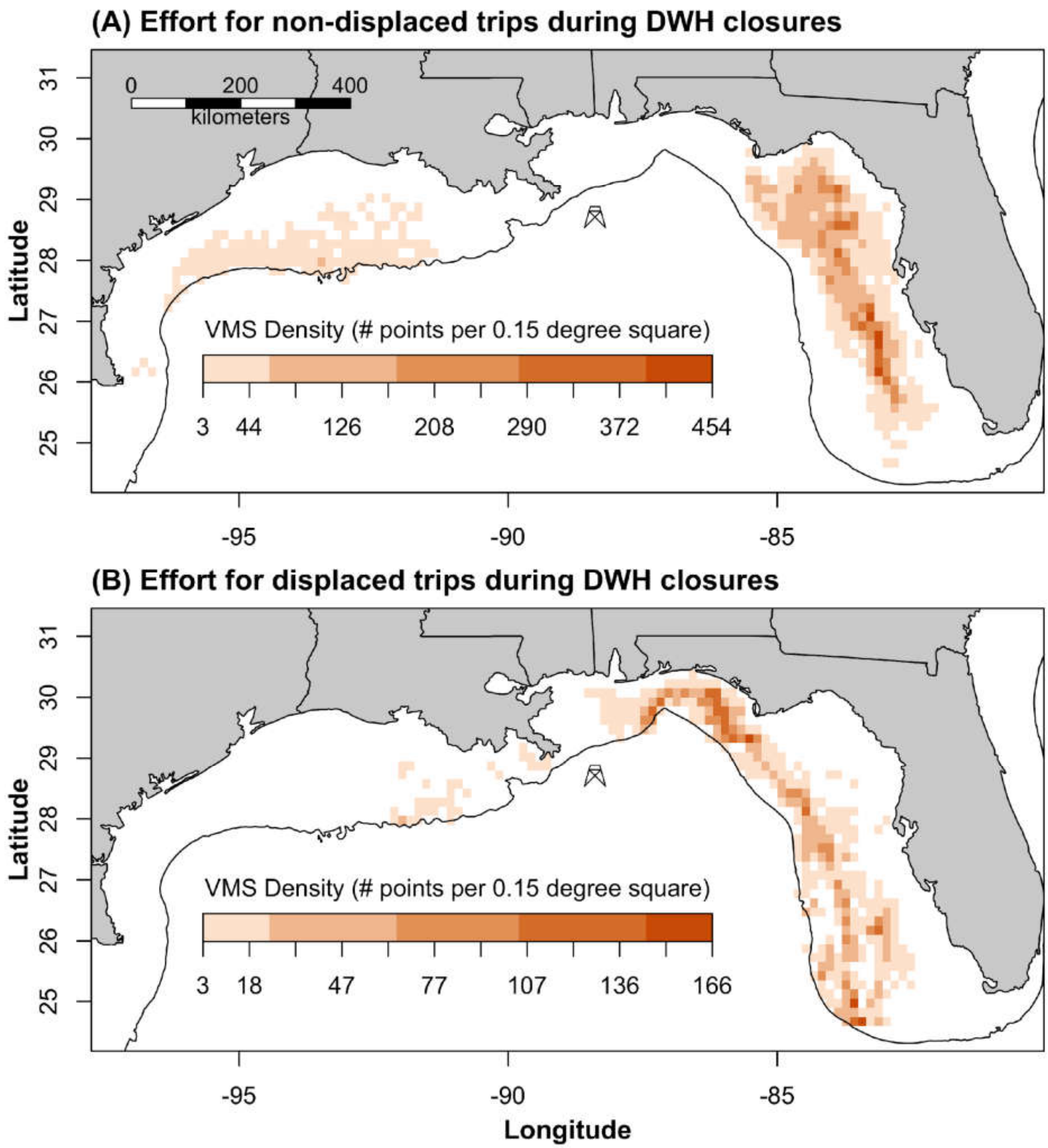


**Figure 3.6.** Density of fishing effort for trips aggregated during DWH emergency fishing closures in 2010. (A) Trips outside of closure boundaries ("intact") and (B) trips inside closures boundaries ("displaced"). Density was calculated as the number of VMS pings per 0.15° × 0.15° grid cell.

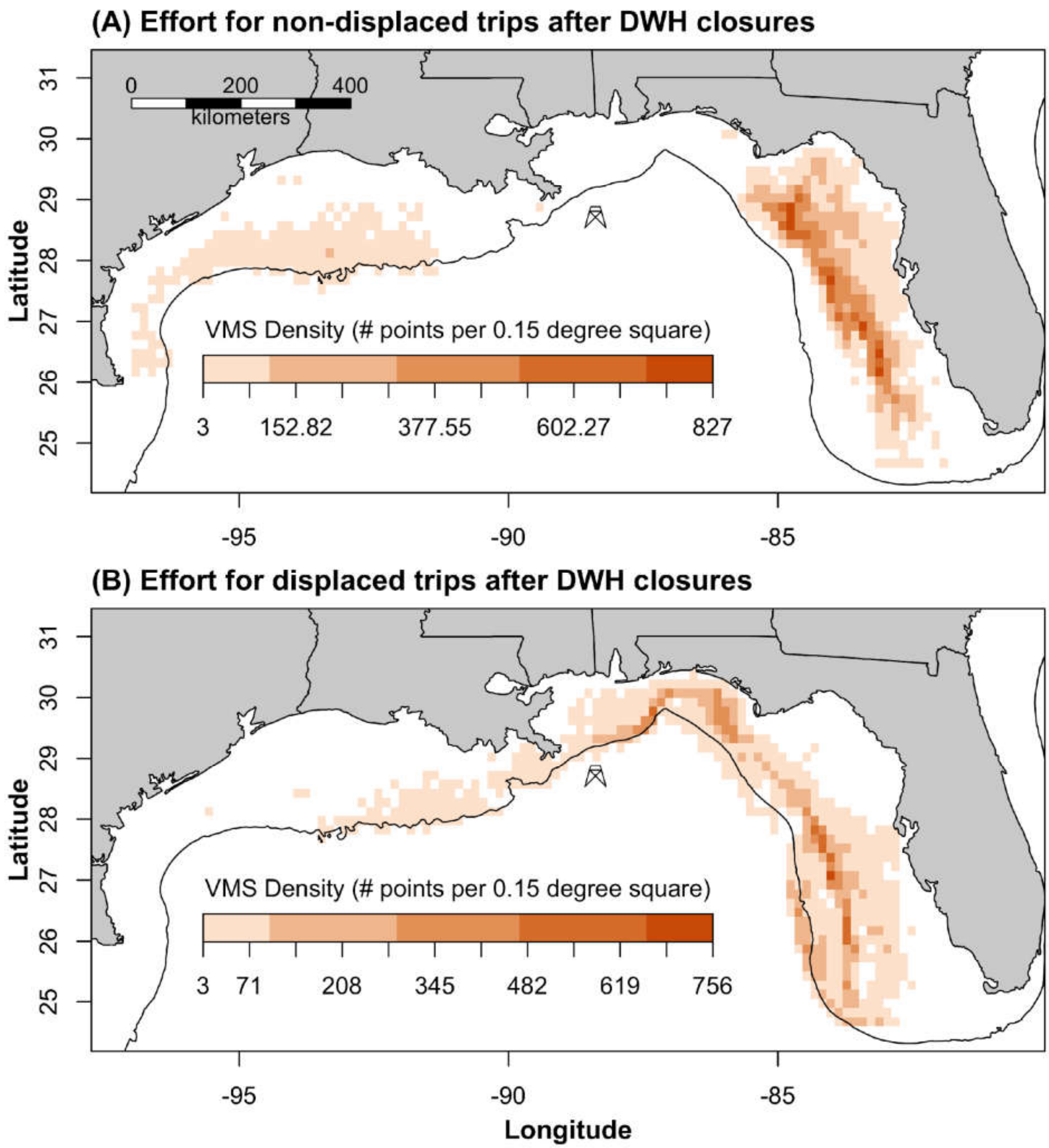


**Figure 3.7.** Density of fishing effort for 2010-2012 trips aggregated after DWH emergency fishing closures. (A) Trips outside of closure boundaries ("intact") and (B) trips inside closures boundaries ("displaced"). Density was calculated as the number of VMS pings per 0.15° × 0.15° grid cell.

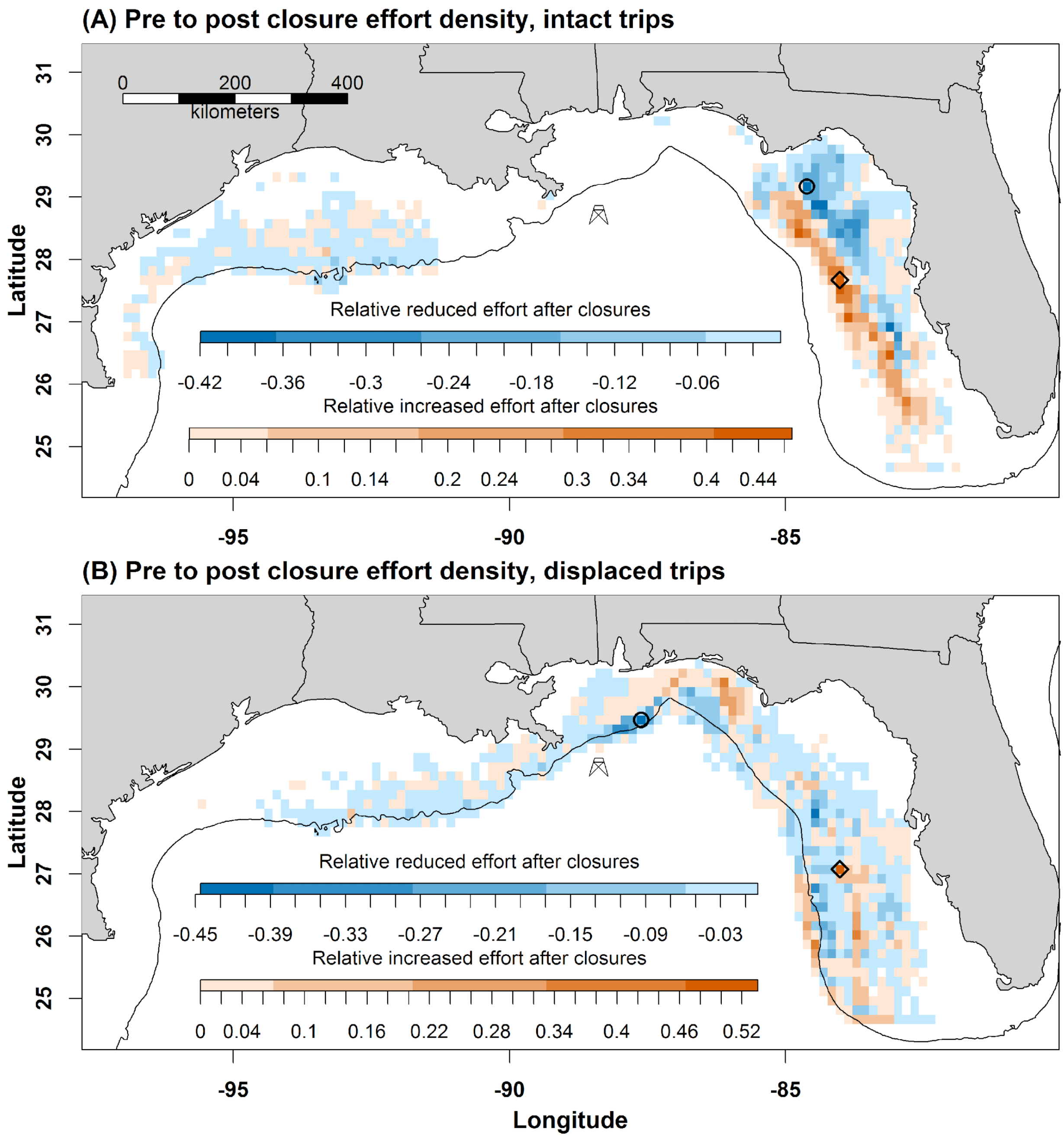


**Figure 3.8.** Difference in standardized effort densities for (A) intact and (B) displaced trips from pre- to post-DWH closures. Blue areas denote a decrease in relative effort density after closures and red areas denote an increase in relative effort density after closures.

# CHAPTER 4. RESILIENCE OF THE COMMERCIAL REEF FISH FISHERY AFTER LARGE-SCALE DISTURBANCE

## Introduction

The impacts of disturbance — whether from climate change, regulatory change, external economic forces, or environmental disasters — are determined in part by vulnerability (i.e., the immediate conditions and state before a disturbance) and resilience (i.e., the ability to absorb, adapt, and cope with disturbance over time). The capacity to adapt to changing fishery resources will ultimately help determine overall social well-being and sustainability for coastal fisheries and fishing communities (Jepson and Colburn 2013, Himes-Cornell and Kasperski 2016). As fisheries management strategies prioritize ecosystem-based management, and continue to recognize that humans are an integral part of fishery systems, there is an increasing need to understand the factors leading to vulnerability, resilience, and sustainability of fishing communities in the face of disturbance. In the long term, the concepts of vulnerability and resilience can help society adapt to or absorb environmental, social, and economic change (Jepson and Colburn 2013).

The commercial reef fish fishery in the GoM has experienced several major perturbations in recent years. Perhaps most notably, during the *Deepwater Horizon* oil spill (DWH) in 2010, a series of emergency fishing closures were put in place from May 2 to November 15, 2010 (197 days), with the area immediately around the wellhead closed until April 19, 2011 (for a total of 352 days). Research since DWH has demonstrated negative effects on the mental and physical health of residents in coastal communities impacted both directly and indirectly by the spill, often disproportionately for those individuals and families involved in the fishing and seafood industries (Grattan et al. 2011, Lee and Blanchard 2011, Cope et al. 2013). Public perceptions of seafood safety after the oil spill impacted demand and consumption of GoM seafood. Yet, the potential outcomes (e.g., leaving the fishery temporally or permanently), impacts on productivity (e.g., a decrease in landings or revenue), and drivers of resiliency (e.g., diverse economic opportunities outside of fishing) for fishers in the wake of the closures remain largely unexplored and poorly understood.

This chapter examined a suite of factors that may have contributed to resilience of the commercial reef fish fishery after the 2010 DWH emergency fishing closures. Resilience in this context was measured at the vessel-level, and quantified as remaining in the fishery after the closures. The main objective of the analysis was to identify potential drivers of fisher resilience and build a set of general linear models (GLMs) to quantify the probability of leaving the fishery after the closures. This work is novel in its use of high-resolution spatial data, coupled with trip logbooks, to build quantitative models identifying drivers of fisher resilience after significant and sudden perturbations to fishery resources in the GoM.

**Materials and Methods**

The analysis for this chapter used the same trip logbook and vessel positioning VMS data as described previously (see Appendices A-D), with the exception that CPUE was calculated for each logbook trip as the total landings divided by either hook-hours (for vertical line gears) or total number of hooks set (for longline). A set of logistic (i.e., binary) GLMs were fitted to predict the probability of individual vessels dropping out of the fishery after the DWH closures in 2010. Pre-closure history of fishing location, trip productivity, and a range of descriptive fishing behavior variables were used to fit the models. The fate of vessels after closures (remained in or dropped out) was used as the model response.

*Vessel Pre-Closure History*

*Productivity history.* For each vessel, logbook trips were used to calculate aggregated pre-closure (January 1, 2000 to May 1, 2010) median trip duration (days), median revenue and median CPUE, revenue and CPUE between-trip variability, median snapper landings, and median grouper landings. All landings, CPUE, and revenue variables were calculated from $\log_e$-transformed values. Outliers were identified and removed using the Grubb's test in the *outlier* package in R (Komsta 2011). Primary and secondary gears used by each vessel, primary and secondary landing states, and primary and secondary species groups targeted were also identified for use as independent variables in the GLMs. Dummy variables for multiple gears used on a trip, multiple gears used among trips, multiple landing states, and multiple species groups targeted were also included as independent variables (Table 4.1).

*Fishing history relative to closures*. A vessel's pre-disturbance VMS record from January 1, 2008 through May 1, 2010 was used to construct a metric of fishing location history relative to DWH closure location. The metric was calculated in order to include fishing location choice in the GLMs. Using the heat map of "closure proportion" (Figure 3.1) and approach described in Chapter 3, pre-closure VMS fishing points for all trips were assigned a closure proportion value and the median spatial impact metric for each trip was calculated. Each vessel's pre-DWH spatial impact metric was then calculated as the median impact metric over all pre-closure trips. Median values were used rather than the mean to reduce any potential skew of the data from especially high or low values (i.e., those vessels that fished very near the wellhead or completely outside the region for a majority of the record).

*Modeling Fisher Resilience*

Longline vessels impacted by an emergency turtle bycatch reduction rule in 2009 were first identified and removed from the data set to avoid conflating effects from the different closures. The turtle bycatch rule was in place from May 18 through October 28, 2009 (164 days), and prohibited bottom longlining for GoM reef fish shoreward of Cape San Blas, Florida, approximately along the 100-m (50-fathom) contour (NOAA Fisheries Service 2009). Vessels reporting longline as the top gear and with logbook trips in 2008 and 2009 (with a start date before the May 18, 2009 closure) were identified. If a vessel did not have any logbook trips with a start date on or after May 18, 2009 it was considered as having "dropped out" and was removed from the dataset.

From the vessels remaining, only those with a consistent trip record in 2008, 2009, and 2010 (with a start date before the April 20, 2010 blowout) were used to fit a set of GLMs[9] to predict the probability of dropping out of the fishery after DWH closures. If a vessel did not have any logbook trips with a start date on or after the initiation of the closures (May 2, 2010), it was considered as having "dropped out." If a vessel had trips after the start of closures, or stopped fishing temporarily and then returned, it was considered as having "remained" in the fishery. This fisher selection approach is similar to other studies that have modeled fisher resilience after closures (e.g., Hackett et al. 2015, Richerson and Holland 2017).

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[9] Separate GLMs were used for revenue productivity variables and CPUE productivity variables (Table 4.1) due to a high collinearity between CPUE variability and revenue variability (Pearson correlation= 0.79, $p<0.001$).

Each GLM used post-closure vessel status (dropped out or remained) as the model response, and was fit using vessel-level aggregate pre-disturbance productivity and fishing location history. Thus, the level of replication was an individual vessel, and there was only one entry per vessel in the model data (n=319 vessels). The predicted probabilities returned from the GLMs were the probabilities of individual vessels dropping out of the fleet after DWH closures. Hierarchical selection of an optimal model using Akaike's Information Criterion (AIC) was conducted in R using the native *stats* package (R Core Team 2016). A null model with only the response and intercept was used as the lower scope of the model selection process, with the full model of all terms (including appropriate interactions) used as the upper scope of model selection (Table 4.1). Both forward and backward selection were used and 'longline' and 'Florida' were used as the standards for gear type and landing state, respectively (GLM coefficients=1). Owing to the skewed nature of the response (i.e., a highly unbalanced number of 0's and 1's), a complementary log-log link function (clog-log) was used for all models. Terms that were marginally significant or non-significant after the hierarchical selection procedure were manually eliminated based on comparison of the scaled deviance between models with a likelihood ratio test; non-significant changes in deviance resulted in deletion of terms, and terms that changed the model deviance the least were removed first. After each variable removal, the model was re-evaluated, and the process was repeated until only significant terms (or one term) remained. All p-values were tested at an alpha of 0.05. Plots of GLM residuals were further used to identify and remove potential outliers.

Estimates of model coefficients, coefficient standard error, and 95% confidence intervals of the final model were transformed from the link function scale (i.e., log linear odds) to the scale of the response (fitted probabilities) by applying the inverse of the link function:

$$(1)\ 1 \quad e^{-e^{X}}$$

where $X$ was either the confidence intervals, GLM estimates, or GLM standard errors. A Hosmer-Lemeshow goodness-of-fit test (Hosmer and Lemeshow 2000) was used from the *ResourceSelection* package in R (Lele et al. 2016) to test the fit between the final logistic GLM and the observed data. This test is based on a Chi-square test statistic and a p-value greater than 0.05 indicates that the model fit is not significantly different than the observed data.

Model results were then grouped into *post-hoc* categories based on the model data means for each significant term returned in the GLMs. If a vessel had a pre-disturbance value less than the mean it was classified as "low", and "high" otherwise. The mean predicted probability of dropping out was then calculated for each *post-hoc* group based on the GLM predicted probability of each vessel in the category, and analysis of variance (ANOVA) tests were used to assess significant differences in predicted model probability among groups. Significant differences from ANOVA were examined further with Tukey-HSD tests for group means; both ANOVA and Tukey-HSD tests were conducted at an alpha of 0.05.

### *Post-Oil Spill Changes in Effort Distribution*

Differences in the spatial distribution of fishing effort were quantified and mapped for: (1) vessels that dropped out of the fishery after DWH closures *vs.* those that remained, and (2) the pre-DWH closure (January 1, 2008 through May 1, 2010) *vs.* post-DWH closure (May 2, 2010 through December 28, 2012) distribution of remaining vessels. Vessels were only included in the effort mapping if they were in the final data set used to fit the GLMs. All effort densities were first calculated as the number of VMS pings per 0.15° × 0.15° grid cell, using the *raster* package in R (Hijmans et al. 2016). Grid cells with less than three VMS pings were reassigned as

"NA" and not mapped, to ensure confidentiality of the data. For ease of comparison and visualization, densities of remaining cells were rescaled (ranging from 0 to 1) relative to the maximum for a given time period or group. Changes in spatial distribution (based on rescaled effort density) were then calculated as the difference between individual density layers. A difference of 0 indicated no change in relative density, or complete overlap in distribution, while a value of 1 or -1 indicated maximum difference, or no overlap in spatial distribution.

A spatial difference index was calculated (Lee et al. 2010) to quantify the overall difference in effort distributions before and after the closures. Using the *raster* package, each raster layer was normalized so that the sum of all cell values was equal to 1. The per-cell absolute difference between two layers was then calculated, summed over the entire study region, and divided by two. This provided an index of difference that varied from zero, such that an index of 0 represented identical spatial distribution of fishing activity, and 1 represented maximum difference, or no overlap, in spatial use.

**Results**

*Modeling Fisher Resilience*

A total of 319 vessels were used to fit the GLMs, 16 of which dropped out post-DWH (5%). Of the 16 that dropped, 6 (38%) reported longline as the top gear, and 10 (62%) reported vertical line as the top gear. The rate of drop out after each closure was within the range that would be expected based on background rates of annual attrition seen in logbook and VMS data (Table 4.2), both prior to and after these closures. From 2000 through 2014, the mean rate of attrition in logbook data was 14% annually, and both logbook and VMS rates of attrition were

~20% annually from 2008-2012. Both logbook and VMS data showed evidence of consolidation in the fleet, with fewer total trips and fewer total vessels over time (Table 4.2).

The final models for DWH closures used revenue variability, median grouper landings, median CPUE, and the interaction between CPUE magnitude and CPUE variability to predict the probability of dropping out of the fishery (Table 4.3). Gear type, state landed, and spatial history of a vessel (i.e., the time spent fishing inside the closure region previous to the closure date) were not significant. The CPUE and revenue GLMs were both significant overall ($\chi^2$=24.5, d.f.=4, $p<0.001$ and $\chi^2$=19.8, d.f.=2, $p<0.001$, respectively) and fitted probability values were not significantly different than observed data based on the Hosmer-Lemeshow test (p=0.35 and 0.19, respectively). The final GLMs for predicting the probability of dropping out after DWH closures were specified as:

CPUE model:

(2) $\text{clog-log}(\pi_i) = \alpha + \beta_1(\text{Median grouper lbs.})_i + \beta_2(\text{Median CPUE})_i + \beta_3(\text{CPUE variability})_i + \beta_4(\text{CPUE variability} \times \text{Median CPUE})_i$

Revenue model:

(3) $\text{clog-log}(\pi_i) = \alpha + \beta_1(\text{Median grouper lbs.})_i + \beta_2(\text{Revenue variability})_i$

where $\pi_i$ was the GLM predicted probability of dropping out on the scale of the complementary log-log link function (i.e., log linear odds) for each vessel *i*, $\alpha$ was the model intercept, and the $\beta_j$'s were the variable coefficients. The fitted probability of dropping out for each vessel *i* on the response scale ($Y_i$) was calculated using the inverse of the link function (see eqn. 1) as:

(4) $Y_i = 1 \quad e^{-e^{\text{clog-log}(\pi_i)}}$

In the first model (eqn. 2), vessels with greater CPUE magnitude were less likely to drop out (p=0.002; Figure 4.1C), but the interaction between CPUE magnitude and CPUE variability was also significant ($p=0.01$; Figure 4.1D; Table 4.3). The interaction term appeared to be driven by CPUE variability, as the increasing pattern between the two were similar (compare curves in Figure 4.1A and D). Thus, the positive relationship with CPUE variability (increasing probability with increasing variability) compensated for the negative relationship with CPUE magnitude. In the second model (eqn. 3), revenue variability displayed a positive relationship with probability of dropping out ($p<0.001$; Figure 4.2A); vessels with less consistent pre-DWH revenues were more likely to drop out. Greater pre-closure grouper landings significantly increased the probability of dropping out in both the CPUE model ($p=0.02$; Figure 4.1B) and the revenue model ($p=0.007$, Figure 4.2B).

*Post-hoc Classification*

Classifying the vessels into *post-hoc* groups relative to variable means revealed combinations of variables that resulted in higher or lower probability of dropping out of the fishery (Table 4.4). In the CPUE model (eqn. 2), vessels with a history of lower grouper landings before DWH were *less* likely to drop out (mean probability=0.02) than those vessels with a history of greater grouper landings (mean probability=0.07). The GLM fitted probability of dropping out was significantly different between grouper landings categories (ANOVA, $F_{1,314}=42.5$, $p<0.001$; Figure 4.3A; Table 4.5). At the same time, vessels with consistently higher CPUE before DWH closures were the *least* likely to drop out (mean probability=0.02), while vessels that had low and variable CPUE were the *most* likely to drop out (mean probability=0.08). The interaction of CPUE magnitude × CPUE variability categories was significant (ANOVA, $F_{1,314}=8.91$, $p=0.003$): vessels with low CPUE had a higher probability of dropping out regardless of variability, while vessels with high CPUE had a significantly greater probability of dropping out when variability was also high (Tukey-HSD, p=0.001; Figure 4.3B). In the revenue model (eqn. 3), vessels with low revenue variability or low grouper landings before closures were *less* likely to drop out (mean probability=0.04 and 0.02, respectively), while vessels with high revenue variability or high grouper landings were *more* likely to drop out (mean probability=0.08 and 0.07, respectively; Table 4.4). The GLM fitted probabilities were significantly different between grouper landings (ANOVA, $F_{1,316}$=63.7, p<0.001; Figure 4.3C) and revenue variability categories (ANOVA, $F_{1,316}$=39.34, p<0.001; Figure 4.3D; Table 4.5).

*Post-Oil Spill Changes in Effort Distribution*

*Dropped and remaining vessels before closures.* The difference in pre-DWH effort distribution among vessels that dropped out *vs.* remained was moderate (Figure 4.4). Remaining vessels had a greater scale of effort density than dropped vessels (maximum density value of 1,525 VMS points per grid cell *vs.* 245, respectively). The overall (2008-2010) spatial difference index between the groups was 0.55, meaning there was ~45% similarity in space use before closures. This overall similarity was only slightly higher than the annual spatial difference index in 2008 (0.63=37% similarity), 2009 (0.61=39% similarity), and 2010 (0.74=26% similarity). The greatest overall difference in relative effort density was 0.95 (i.e., almost no overlap), offshore of the central Florida Peninsula (black diamond in Figure 4.4C). This difference was due to a much greater relative density in the area for remaining vessels. Interestingly, the region with greater relative density of dropped vessels was just slightly northwest of this (black circle in Figure 4.4C). There was also greater relative density of remaining vessels throughout the Big Bend and off the Alabama coast northeast of the DWH wellhead (Figure 4.4C). The difference in the distributions between the two groups of vessels may be attributable to the very low number of vessels that dropped out overall (only 5%), or the geographic difference in distribution for those vessels that remained *vs.* dropped out: effort was distributed throughout the GoM before closures for vessels that remained (Figure 4.4A), while effort was heavily concentrated in the north-central and eastern GoM for those that dropped out (Figure 4.4B). The similarity in effort was concentrated entirely in the north-central and eastern GoM.

*Remaining vessels before and after closures.* Vessels that remained in the fishery after DWH did not undergo a substantial shift in spatial distribution after the initiation of fishing closures. The overall spatial difference index for these vessels was 0.20, meaning that there was a ~80% similarity in space use from before to after closures. The spatial difference index for remaining vessels by gear type was slightly higher than the overall value, but still indicated no substantial shifts in spatial distribution post-closure: 0.21 for vertical line (79% similarity) and 0.23 for longline (77% similarity). The maximum absolute difference in relative effort density from before to after closures was 0.59, and was centered on the 200-m isobath off the Alabama coast. This pattern was driven by a post-closure reduction in relative effort density off the Alabama coast (black circle in Figure 4.5). The Florida Panhandle, Big Bend, and WFS regions also saw moderate changes in relative effort density after closures: effort was generally reduced along the Panhandle and Big Bend, and shifted slightly south/southeastward, increasing the relative density along the middle and southern WFS. The greatest increase in post-closure density for remaining vessels was centered on the southern WFS, with an absolute difference of 0.43 (black diamond in Figure 4.5).

## Discussion

We expected that vessels with a significant history of fishing within closed areas would be more likely to leave the fishery once closures were put in place. The models, however, did not explicitly include spatial distribution metrics or landing state as a significant predictor of leaving the fishery. Instead, vessels with a record of consistently high CPUE, low grouper landings, or a combination of consistent revenue and low grouper landings were the least likely to drop out after DWH closures. This result is somewhat surprising, given the importance of spatial dynamics in fishers' decision making and fishing outcomes (Branch et al. 2006, Dowling et al. 2012, Weninger and Perruso 2013). Yet, pre-closure median grouper landings was a significant predictor of dropping out of the fishery after DWH closures and may be linked to pre-closure effort distribution. Reef fishers in the eastern GoM tend to rely on grouper species more so than fishers in the central or western GoM, in part due to the historically overfished red snapper stock in the eastern GoM (see also the results in Chapter 2). The geographic disparity in distribution between vessels that dropped out *vs.* remained (Figure 4.5) may therefore be reflecting the significance of grouper landings in the models; the probability of dropping out was greater with increased grouper landings and there was a high concentration of vessels that dropped out in the eastern GoM where grouper landings are dominant. A more rigorous analysis of spatially-explicit and spatial proxy variables for dropped and remaining vessels (e.g., mean fishing location and geographical range of fishing locations), may help elucidate the finer-scale spatial differences in vulnerability and resilience in this fishery.

There may have additionally been other motivations or modifications in response to these emergency closures. The seemingly small shift in effort distribution after DWH closures (quantified as an ~80% overall similarity in space use, and gear-specific similarity ranging from

71-77%) suggests either a fidelity to fishing grounds, or the ability to adapt to changing conditions independent of changes in spatial utilization. For instance, fishers on remaining vessels may have had the ability to use different gears or target different species in the reef-species complex. Transferability of vessels, gears, and crew skills and knowledge is an important factor for fishery participation (Hackett et al. 2015). Of the 303 vessels that remained after DWH closures, 160 (53%) used multiple gear types within single trips, 234 (77%) reported multiple top gear types between trips, and 278 (92%) reported more than one top species group landed.

At the same time, payments made through the Vessels of Opportunity (VoO) Program and to commercial fishers, crew, and vessel owners from the Seafood Compensation Program likely buffered against potential oil-related economic losses. Payments through the VoO program for spill remediation efforts totaled $283 million, and $2.2 billion was paid through the Seafood Compensation Program for lost fishing-related income (Deepwater Horizon Claims Center 2018). The financial buffer from these payments likely ameliorated some of the impacts from the oil spill and subsequent fishing closures, thereby allowing more fishers to remain in the fishery than would have otherwise and decreasing the rate of fisher drop out (5% compared to 14-20% background). This mechanism is analogous to insurance programs buffering against financial risk and environmental uncertainty in agricultural production. In the case of fisheries, unanticipated natural or human-induced disaster may lead to revenue shortfalls because of lower catch and/or lower prices, with indemnity payments making up for lost revenue and sustaining individual fishers that might otherwise be forced to leave (Mumford et al. 2009).

The rate of drop out after each closure was well below or within the range that would be expected based on background rates of attrition. Other studies have similarly reported on the resiliency of GoM fishes and fisheries post-DWH (Fodrie et al. 2014, Murawski et al. 2016, Schaefer et al. 2016), and there is evidence that the emergency closures may have had positive effects on the abundance of some near-shore and estuarine juvenile fishes in 2010 through release of fishing mortality on spawning adults (i.e., a closure reserve effect; Fodrie and Heck 2011, Schaefer et al. 2016). Population-level benefits have also been reported for Gulf menhaden (*Brevoortia patronus*) in the northern GoM (Short et al. 2017), owing to reduced predation pressure after high oil-induced mortality of some predators (i.e., seabirds, marsh birds, and bottlenose dolphins) and diversions of fresh water from the Mississippi River that inhibited access to juvenile menhaden for others. Recruitment of the 2010 Gulf menhaden year class was anomalously high and led to a population biomass that was more than twice the average biomass for the preceding decade (Short et al. 2017). This population increase — especially for a major forage fish species at the base of the food web — presents the possibility of additional indirect effects throughout the northern GoM ecosystem *via* increased predation on Gulf menhaden prey or greater availability of Gulf menhaden biomass to surviving predators.

While fishers in the far western GoM were not directly impacted by the DWH closures, it is possible that perceived risks from the DWH oil spill impacted willingness or ability to fish, or reduced overall revenue and profitability as the result of reduced consumer confidence and market prices. There were also relatively few productivity or other descriptive variables that were significant in the models. This may have been due to similarities in catch and revenue values between the vessels that dropped out and those that remained (compare the range of observed values in Figures 4.1 and 4.2). Using additional quantitative variables for the models is therefore warranted. Exploring fishery-level patterns in a multivariate context (e.g., a canonical analysis of species composition and abundance in landings coupled with fishing locations and descriptive behavioral variables), could also reveal additional drivers of differences between vessels that remained and those that left.

A decadal record of fishing trips informed our modeling approach, under the assumption that an aggregate record of pre-impact fishing would influence a fishers' decision making, and therefore the potential for resilience after each closure. However, there may be different time scales operating that were not captured in our analysis. For example, fishers may be using a one year, months- or days-long record of fishing to make decisions. There may also be processes occurring at the regional-, state-, or county-scale that were not explicitly examined (potentially contributing to the result that spatial indices were not significant). While a more in-depth partitioning of the data may help illuminate some of these processes, ethnographic studies in fishing communities would give local context and external validity to our results, and assure that our conclusions make sense in reality (Jacob et al. 2010).

Given the small percentage of vessels that dropped out of the fishery, the comparatively high background rate of attrition, and the small shifts in effort distribution for remaining vessels post-DWH, we might conclude that this commercial reef fishery was largely resilient to the emergency closures put in place during the DWH oil spill. Still, there is some evidence that winners and losers post-DWH were location specific, with a greater concentration of dropped vessels in the north-central and eastern GoM.

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**Table 4.1.** Factors considered as inputs into the logistic general linear models for quantifying fisher resilience after DWH fishing closures.

| Factor | CPUE model | Revenue model |
|---|---|---|
| Spatial impact metric | Distribution of pre-disturbance effort relative to closures | |
| Aggregated pre-disturbance fishing history characteristics | Median trip duration (days)<br>Primary species group landed (Top group)<br>Secondary species group landed<br>Primary landing state (Top state)<br>Secondary landing state<br>Primary gear used (Top gear)<br>Secondary gear used | |
| Aggregated pre-disturbance fishing history characteristics coded as dummy variables | Multiple gears used within a trip<br>Multiple top gears used between trips<br>Multiple top groups between trips<br>Multiple landing states between trips | |
| Aggregated pre-disturbance productivity | Median CPUE<br>CPUE variability<br>Median snapper landings<br>Median grouper landings | Median revenue<br>Revenue variability<br>Median snapper landings<br>Median grouper landings |
| Interactions considered | Spatial impact × CPUE<br>Spatial impact × CPUE variability<br>Spatial impact × snapper landings<br>Spatial impact × grouper landings<br>Spatial impact × Top group<br>CPUE × trip duration<br>CPUE variability × trip duration<br>CPUE × Top state<br>CPUE × Top group<br>CPUE × CPUE variability<br>CPUE variability × Top state<br>CPUE variability × Top group<br>Snapper landings × trip duration<br>Grouper landings × trip duration<br>Snapper landings × Top state<br>Grouper landings × Top state<br>Trip duration × Top state<br>Trip duration × Top group | Spatial impact × revenue<br>Spatial impact × revenue variability<br>Spatial impact × snapper landings<br>Spatial impact × grouper landings<br>Spatial impact × Top group<br>Revenue × trip duration<br>Revenue variability × trip duration<br>Revenue × Top state<br>Revenue × Top group<br>Revenue variability × revenue<br>Revenue variability × Top state<br>Revenue variability × Top group<br>Snapper landings × trip duration<br>Grouper landings × trip duration<br>Snapper landings × Top state<br>Grouper landings × Top state<br>Trip duration × Top state<br>Trip duration × Top group |

**Table 4.2.** Number of unique vessels and rates of vessel entry and attrition as calculated from the logbook and VMS data.

| | Year | Number of unique vessels in first year | % Remaining | % Dropped | % Entering |
|---|---|---|---|---|---|
| **Logbook** | 2000-2001 | 417 | 91.6 | 8.4 | 11.2 |
| | 2001-2002 | 431 | 93.7 | 6.3 | 14.3 |
| | 2002-2003 | 470 | 90.2 | 9.8 | 10.2 |
| | 2003-2004 | 474 | 92.2 | 7.8 | 14.5 |
| | 2004-2005 | 511 | 89.8 | 10.2 | 13.8 |
| | 2005-2006 | 530 | 88.7 | 11.3 | 14.9 |
| | 2006-2007 | 552 | 86.6 | 13.4 | 11.3 |
| | 2007-2008 | 540 | 85.5 | 14.5 | 14.9 |
| | 2008-2009 | 543 | 86.9 | 13.1 | 15.3 |
| | 2009-2010 | 556 | 73.0 | 27.0 | 11.0 |
| | 2010-2011 | 457 | 79.8 | 20.2 | 18.2 |
| | 2011-2012 | 447 | 82.0 | 18.0 | 15.3 |
| | 2012-2013 | 432 | 77.3 | 22.7 | 11.7 |
| | 2013-2014 | 381 | 81.7 | 18.3 | 14.0 |
| | 2014 | 359 | n.a. | n.a. | n.a. |
| | Mean 00-14 | 473 | 85.6 | 14.4 | 13.6 |
| | Mean 08-12 | 487 | 80.4 | 19.6 | 14.9 |
| **VMS** | 2008-2009 | 516 | 84.3 | 15.7 | 15.7 |
| | 2009-2010 | 516 | 71.9 | 28.1 | 12.5 |
| | 2010-2011 | 424 | 80.4 | 19.6 | 17.6 |
| | 2011-2012 | 414 | 80.4 | 19.6 | 14.8 |
| | 2012 | 391 | n.a. | n.a. | n.a. |
| | Mean 08-12 | 452 | 79.3 | 20.7 | 15.2 |

**Note:** Percentages were calculated using the number of unique vessels (as identified by vessel hull numbers) in each year. Only the number of vessels is reported for the terminal year of each data set. n.a. = not applicable.

**Table 4.3.** Results of logistic general linear models (GLMs) for the probability of dropping out of the fishery after 2010 DWH emergency fishing closures.

| GLM | Effect | Coefficient | SE | P | Lower CI | Upper CI |
|---|---|---|---|---|---|---|
| CPUE | (Intercept) | 0.02 | 0.93 | < 0.001 | 0.002 | 0.08 |
| | Median CPUE | 0.35 | 0.73 | 0.002 | 0.21 | 0.52 |
| | CPUE variability | 0.77 | 1.00 | 0.89 | 0.001 | 1.00 |
| | Median grouper landings | 0.76 | 0.69 | 0.02 | 0.67 | 0.87 |
| | Median CPUE × CPUE variability | 1.00 | 0.95 | 0.01 | 0.81 | 1.00 |
| Revenue | (Intercept) | 0.004 | 0.92 | < 0.001 | 0.001 | 0.02 |
| | Revenue variability | 1.00 | 0.996 | < 0.001 | 0.99 | 1.00 |
| | Median grouper landings | 0.78 | 0.69 | 0.007 | 0.69 | 0.87 |

**Note:** Coefficients, standard errors (SE), and probability confidence intervals (CI) are on the scale of the response (probability), and were calculated using the inverse of the link function used in the general linear model (see eqns. 1 and 4). Models for CPUE and revenue were run separately due to the high collinearity between CPUE and revenue variability. The models were fit using pre-closure $\log_e$-transformed values.

**Table 4.4.** Mean logistic general linear model (GLM) fits for the probability of dropping out of the fishery after DWH emergency closures, based on *post-hoc* category groupings for vessels.

| GLM | Variable | Variable Mean | Grouping | Mean GLM probability | N |
|---|---|---|---|---|---|
| CPUE | CPUE magnitude | 5.4 | High | 0.03 | 173 |
| | | | Low | 0.08 | 146 |
| | CPUE variability | 1.1 | Low | 0.04 | 225 |
| | | | High | 0.06 | 94 |
| | Grouper landings | 21.2 | Low | 0.02 | 124 |
| | | | High | 0.07 | 195 |
| | CPUE magnitude × CPUE variability | n.a. | High × Low | 0.02 | 122 |
| | | | High × High | 0.05 | 51 |
| | | | Low × Low | 0.08 | 103 |
| | | | Low × High | 0.08 | 43 |
| Revenue | Revenue variability | 1.1 | Low | 0.04 | 227 |
| | | | High | 0.08 | 92 |
| | Grouper landings | 21.2 | Low | 0.02 | 124 |
| | | | High | 0.07 | 195 |

**Note:** The categories were assigned based on the means of each variable found to be significant in the GLMs, with "low" defined as less than the model data mean. Means given are back-transformed values. n.a. = not applicable.

**Table 4.5.** ANOVA results from *post-hoc* category comparisons among significant variables after fitting general linear models (GLMs).

| GLM | Effect | d.f. | SS | MS | F | P |
|---|---|---|---|---|---|---|
| CPUE | CPUE magnitude | 1 | 0.21 | 0.21 | 46.6 | < 0.001 |
| | CPUE variability | 1 | 0.03 | 0.02 | 5.4 | 0.02 |
| | Grouper landings | 1 | 0.19 | 0.19 | 42.5 | < 0.001 |
| | CPUE magnitude × CPUE variability | 1 | 0.04 | 0.04 | 8.9 | 0.003 |
| | Residual | 314 | 1.43 | 0.005 | | |
| | | d.f. | SS | MS | F | P |
| Revenue | Revenue variability | 1 | 0.13 | 0.13 | 39.3 | < 0.001 |
| | Grouper landings | 1 | 0.21 | 0.21 | 63.7 | < 0.001 |
| | Residual | 316 | 1.04 | 0.003 | | |

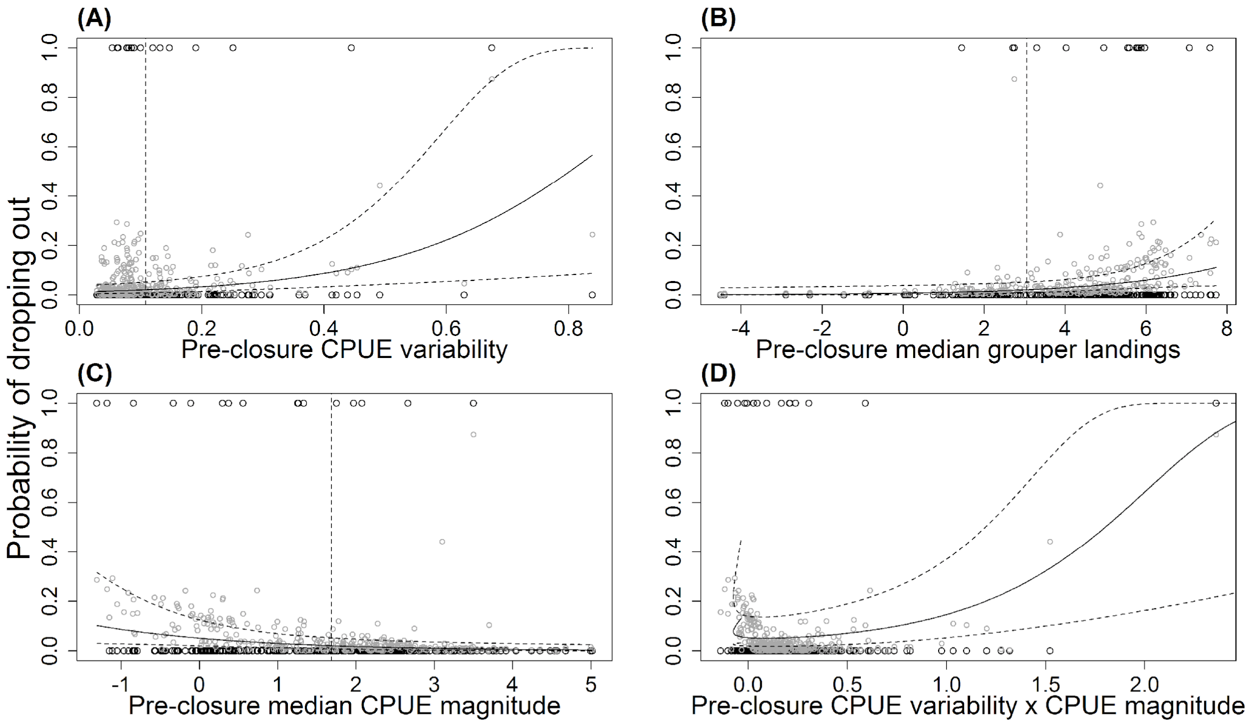


**Figure 4.1.** CPUE model of fitted GLM probabilities as a function of (A) pre-closure between-trip CPUE variability, (B) pre-closure median grouper landings, (C) pre-closure median CPUE magnitude, and (D) the interaction of CPUE variability and CPUE magnitude. Black circles are observed values (1=dropped out, 0=remained), gray circles are model fitted probabilities, and the dashed vertical lines denote the mean value of each variable (used to group vessels into *post-hoc* groups, see Table 4.4). Solid black lines are predicted probability values (holding other model variables constant at their mean value) and dashed lines are 95% confidence intervals.

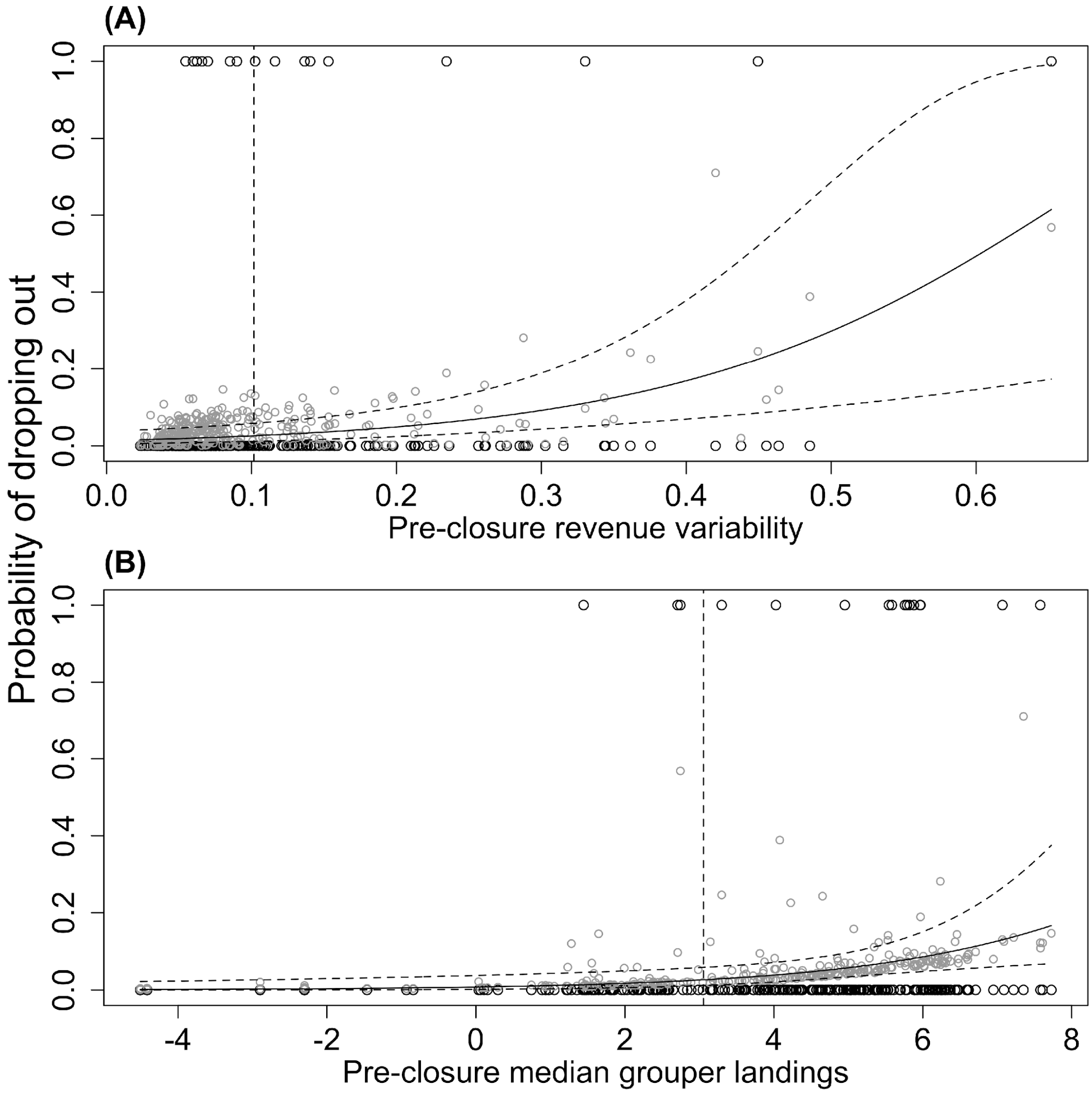


**Figure 4.2.** Revenue model of fitted GLM probabilities as a function of (A) pre-closure between-trip revenue variability and (B) pre-closure median grouper landings. Black circles are observed values (1=dropped out, 0=remained), gray circles are model fitted probabilities, and the dashed vertical lines denote the mean value of each variable (used to group vessels into *post-hoc* groups, see Table 4.4). Solid black lines are predicted probability values (holding other model variables constant at their mean value) and dashed lines are 95% confidence intervals.

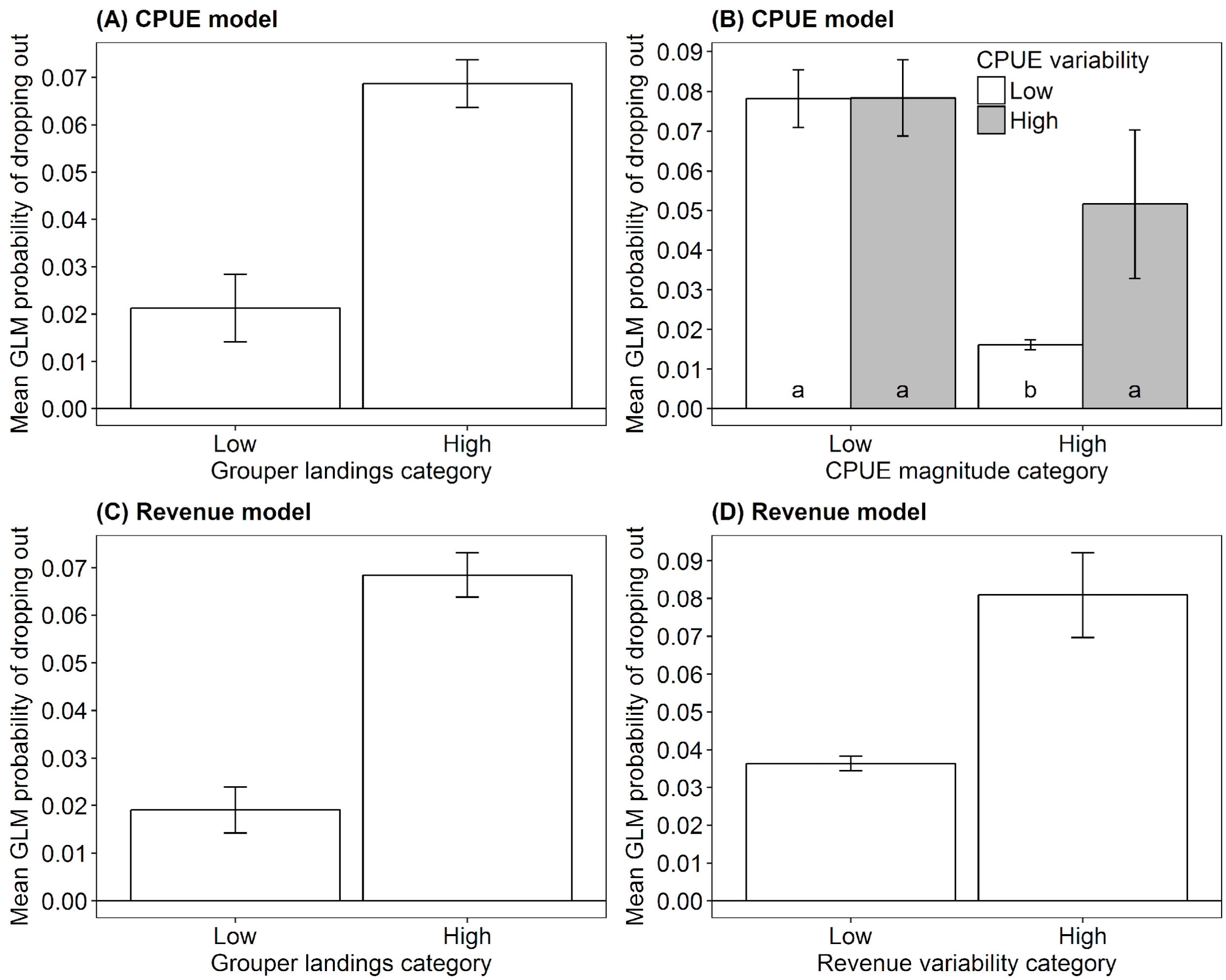


**Figure 4.3.** Fitted GLM probability of dropping out after DWH closures, based on *post-hoc* categorical grouping of vessels from the CPUE model (panel A-B) and the revenue model (panel C-D). The categories were assigned based on the means of each variable found to be significant in the model, with "low" defined as less than the model data mean (see Table 4.4). Different letters in the bars in panel B indicate significant differences from a Tukey-HSD *post-hoc* test on group means after ANOVA. All values are means ± SEM.

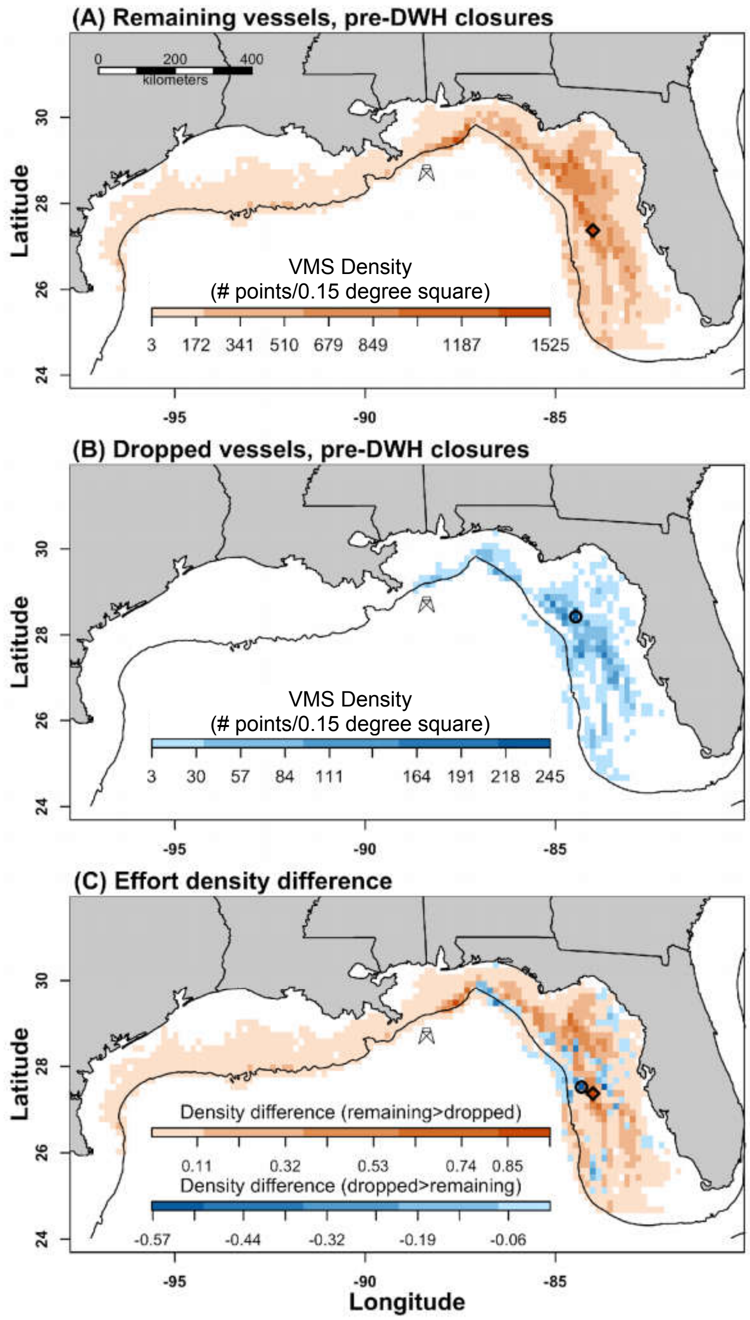


**Figure 4.4.** Fishing effort distribution and difference between vessels before DWH closures. (A) Effort density for vessels that remained in, (B) effort density for vessels that dropped out, (C) difference in relative effort density (scaled to 1) between dropped and remaining vessels. The highest density region is marked with a black diamond (A) and black circle (B). In panel C, the red color indicates regions where the density of remaining vessels was greater than dropped vessels, and the blue color indicates regions where the density of dropped vessels was greater than remaining vessels. The greatest difference for dropped vessels (i.e., where the pre-closure relative density was greatest over remaining vessels) is marked with a black circle (-0.57) and the greatest difference for remaining vessels over dropped (0.95) is marked with a black diamond.

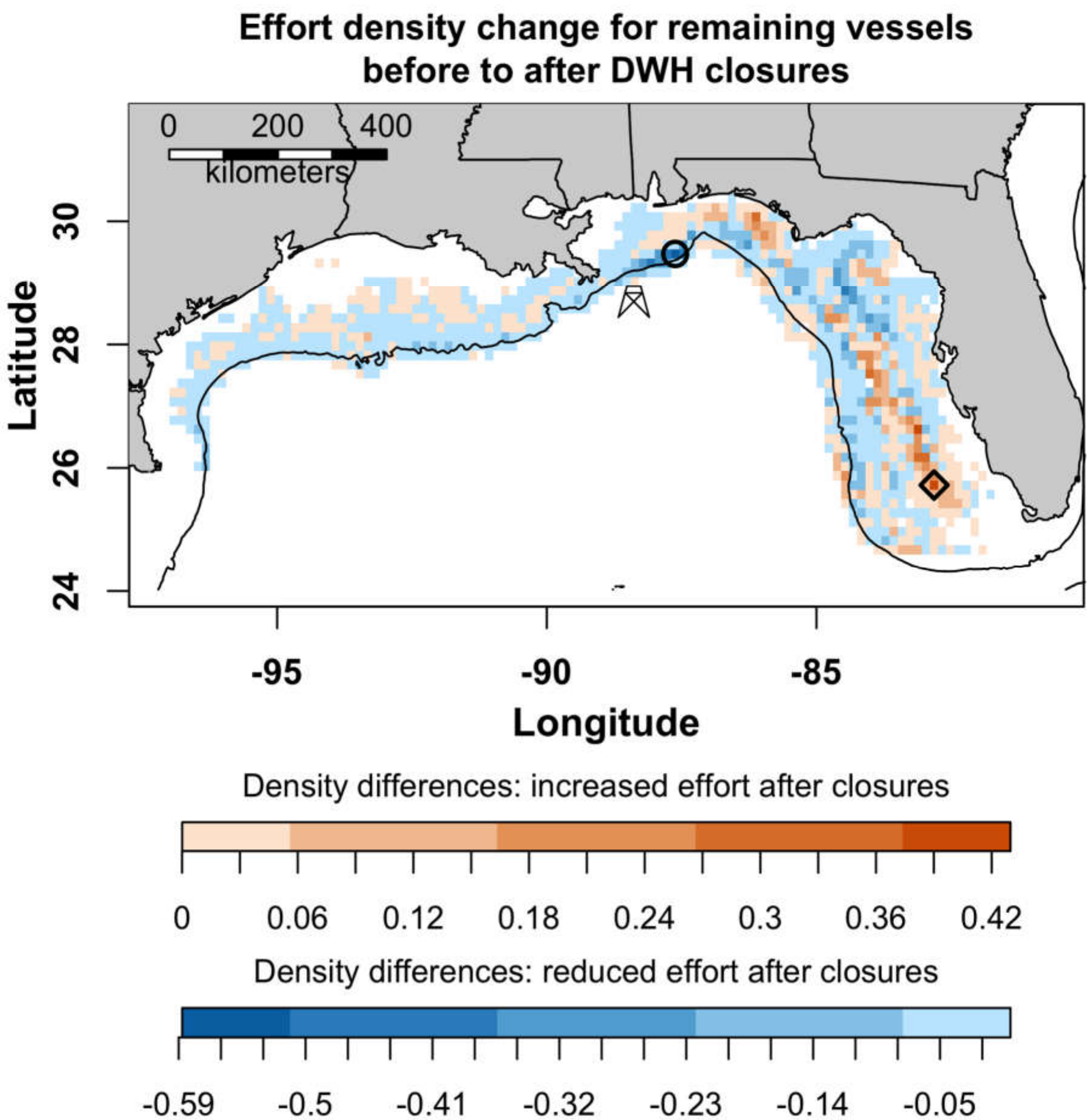


**Figure 4.5.** Differences in fishing effort distribution from before to after DWH closures for vessels that remained in the fishery. Areas with reductions in effort density after DWH closures are in blue, with the maximum difference (-0.59) indicated by the black circle. Areas with increases in effort density are in red, with the maximum difference (0.43) indicated by the black diamond.

# CHAPTER 5. CONCLUSIONS

## Summary of Dissertation

This dissertation characterized the spatiotemporal patterns of productivity and fishing effort for the commercial reef fish fishery in the Gulf of Mexico (GoM) and quantified responses to large-scale disturbance in the form of the *Deepwater Horizon* oil spill fishing closures. Analyses utilized traditional fisheries-dependent datasets (onboard observer and trip logbooks), but were novel in the use of complementary high-resolution VMS data to quantify displacement, characterize changes in space use over time, and build quantitative models identifying drivers of fisher resilience.

Trip-level productivity (quantified as ex-vessel revenue and CPUE) for total species, snappers, and grouper CPUE increased significantly over the study period. Although grouper revenue was not significantly different between the start and end of the study period (2000 *vs.* 2014), there were significant fluctuations in the interim years, including a significant decrease in grouper revenue in the central and eastern regions in 2010 and 2011. Snapper and grouper productivity were separated into distinct west and east regions, and effort between vertical line and longline gear was clearly delineated based on existing regulations for longline gear. Productivity was consistently highest in the western GoM over the study period, although fishing effort was most dense in the central and eastern regions. This result was likely driven by the dominance of snappers in catch for trips in the west.

Similarly, Gulf-wide productivity patterns for trips inside *vs.* outside DWH fishing closure boundaries did not change over time. There were, however, increases in CPUE for trips *inside* closure boundaries in the east both during and after closures as well as post-closure increases in revenue for trips inside closure boundaries in the east. This pattern may have been the result of reduced competition from fewer vessels or trips in and after 2010. Consolidation of the fleet after 2010 was evident in both the logbook and VMS data and vessels that dropped out of the fishery post-DWH were concentrated in the north-central and eastern GoM (*Chapter 4*). Fewer vessels could have led to more available fish biomass, and therefore overall greater landings, CPUE, and revenue per trip over time in this area.

While there were regionally varying outcomes for individual fishers — with a greater concentration of dropped vessels in the north-central and eastern GoM — the overall attrition rate after DWH was well below what was expected based on the background annual attrition rate alone (5% *vs.* 14-20%). Given the magnitude of the oil spill on the environment, businesses, tourism, and the seafood industry in the GoM, this is a surprising and significant result. The drivers of attrition were found to be a pre-closure history of revenue variability, median grouper landings, median CPUE, and the interaction between CPUE magnitude and CPUE variability. At the same time, resilience was likely enhanced by the significant emergency compensation payments made to fishers and vessel owners for lost income and assistance with spill remediation efforts (Figure 5.1). Continuing work on this fishery would certainly benefit from including additional variables in the models, including landings and revenue for individual species and finer-scale information on individual households and communities.

Reduced competition may not have been only a consequence of DWH fishing closures. Results presented here suggest that the implementation of the Grouper-Tilefish IFQ in 2010 may have played a role in consolidation in the eastern GoM. Fishing effort for vessels that left the fishery after DWH closures was most concentrated in the eastern GoM, and trip productivity in the east was dominated by grouper species (*Chapters 2 and 4*). Yet, the spatial impact metric quantified as a measure of displacement was lowest in the eastern region, meaning that fewer trips were located inside closure boundaries (*Chapter 3*). It is therefore possible that the loss of vessels after 2010 was a consolidation effect from the grouper IFQ. Consolidation of participants has been previously documented for the Grouper-Tilefish IFQ (Brinson and Thunberg 2016), and would not be unexpected since reducing overcapacity is a primary objective of restricted access management. Surely, the IFQ and the oil spill could have been working in tandem to force out more fishers than might have dropped out from either disturbance alone.

On the other hand, the Red snapper and Grouper-Tilefish IFQs (implemented in January 2007 and January 2010, respectively) may have "primed" the fishery for increased resilience during and after the DWH oil spill. That is, if inconsistent or marginally productive fishers (i.e., the *least* resilient to DWH closures, as quantified in *Chapter 4*) left after IFQ implementation, the baseline capacity for resilience in the fishery may have been enhanced. The exact reasons for attrition in the Grouper-Tilefish and Red snapper components of this fishery at the time of both DWH and IFQ implementation requires more data than is currently available for this work, including more fine-scale ethnographic and economic information.

The results presented here suggest an optimal utilization of fishing grounds on the part of fishers in response to gear restrictions and DWH fishing closures. Trip-level revenue and CPUE continued to increase annually even as DWH closures, the longline seasonal closure, and IFQ systems for the two major species groups in the fishery were put into place. Effort was clearly concentrated based on gear type and existing longline regulations, and moved in response to the initiation and cessation of DWH fishing closures. CPUE magnitude and variability and revenue variability were significant GLM variables (*Chapter 4*) and the distribution of trip revenue hot spots appeared to contribute to the overall patterns of effort distribution (*Chapter 2*).

The fleet displayed a largely similar effort distribution before and after closures (66-78% overall, and 63-79% by gear), including a return to fishing grounds that were inside closure boundaries during DWH. Still, once closures were removed there was a decrease in effort density off the Alabama coast and Florida Big Bend and concomitant increases in effort density on the West Florida Shelf (WFS). Shifts in effort onto the WFS during DWH persisted after the closures were removed for both fishers that were inside and outside closure boundaries during DWH. It is possible that fishers began to take advantage of an already productive fishing ground on the southern WFS during and after DWH. Thus, changes in space use during and after DWH could have contributed to the observed increases in both CPUE and revenue over time. Characterizing and quantifying the productivity and profitability of this specific region warrants further investigation.

In addition, the increased CPUE and revenue in the years following closure re-openings, suggest that fishermen remaining in the fleet after DWH may have benefitted from a closure reserve effect and/or reduced competition. A closure reserve effect – wherein emergency closures protected targeted stocks by virtue of not allowing any fishing – could have resulted in increased available biomass in the short term, and therefore increased landings and CPUE for individual fishers once closures were removed. Evidence for a closure reserve effect has been suggested for other coastal species (Fodrie and Heck 2011), including Spotted seatrout (*Cynoscion nebulosus*) in the north-central GoM that had orders-of-magnitude increases in juvenile abundance and elevated catch rates after closures in 2010.

## Environmental Considerations

The environment is an integral part of any fisheries system. While environmental variability and oceanographic conditions were not a primary focus of this dissertation, it is important to place the results into a broader environmental and oceanographic context.

*Species-Habitat Relationships*

The benthic habitat across the GoM varies from being dominated by muds in the west (Texas and Louisiana) to sands and gravel in the north-central and east (Mississippi, Alabama, and Florida).[10] This spatial distribution of bottom habitat types impacts the distribution of major fishery species. Red snapper in particular rely on intermediate and high relief reef habitats throughout its lifetime, with requirements for increasingly complex structure as a fish grows (Gallaway et al. 2009). While natural reef habitat is generally sparse in the northern GoM (Parker et al. 1983), the proliferation of nearly 4,000 oil and gas platforms in the western and north-central GoM and ~20,000 artificial reefs off of Alabama's coast has supported significant commercial and recreational Red snapper fisheries for at least the past 50 years (Gallaway et al. 2009, Shipp and Bortone 2009). Similarly, Gag grouper depend on deep hard-bottom reefs on the edge of the West Florida Shelf for spawning, coastal seagrass beds for larval settlement, and near-shore hard-bottom reefs as juveniles and adults (Koenig and Coleman 1998, Ellis and Powers 2012). Red grouper rely on karst topography and carbonate sands in the eastern GoM to excavate pits that are used for home territories and spawning sites (Coleman et al. 2010, Wall et al. 2011, Harter et al. 2017). The regional patterns of logbook species composition quantified in

[10] NOAA Gulf of Mexico Data Atlas. 2017. *Marine Geology: Dominant Bottom Types and Habitats.* Available online at: https://gulfatlas.noaa.gov/catalog/products/physical/marine-geology/.

this work (i.e., snappers in the west and central GoM and groupers in the east) fall in line with the distributions that would be expected based on these bottom habitat types.

The interaction between habitat, species distribution, area or seasonal closures, and gear-specific functionality could have important ramifications for effort redistribution, profitability of fishing locations, and ultimately the outcomes of regulatory change. The distribution of fish species is strongly related to the distribution of suitable habitat (Yeager et al. 2011, Arias-González et al. 2012), and it has been demonstrated that snapper and grouper species have strong site fidelity to reef or hard-bottom structure (e.g., Saul et al. 2013). While habitat maps were not used in this analysis, the high percentage of trips reporting Red snapper, Vermilion snapper, Red grouper, and Gag (Table 2.5) make it unlikely that fishers are *not* fishing in these habitats.

Furthermore, the functionality and effectiveness of particular gear types will vary based on the environment and the target species. For example, Stelzenmüeller et al. (2008) found fishing effort in offshore waters of the United Kingdom concentrated regionally based on seafloor sediment type, which was additionally linked to the functionality of the different gears used in the fleet. If only particular gear types are restricted from an area (i.e., bottom longline), the cost of fishing could increase disproportionately for some fishers (e.g., due to increased travel time to avoid the closure), thereby reducing overall profitability for certain fishing locations. While fishers can change targeting behavior to account for regulations, the relationship between habitat, species distributions, and gear functionality for particular species in particular habitats can lead to serious and complex outcomes for a fishery (Valcic 2009).

*Phytoplankton Blooms*

Spring phytoplankton blooms have been observed in the Florida Big Bend and across the WFS, owing to seasonal high-nutrient discharges from the Mississippi River, Apalachicola River, and local rivers throughout northwest Florida (Gilbes et al. 1996, Gilbes et al. 2002). These high-nutrient river inputs are transported across the WFS by east/southeastward currents, which are set up by seasonal winds, large-scale surface heat fluxes, and across-shelf temperature gradients (Weisberg et al. 2005). Strong winds or Loop Current (LC) intrusions can further intensify seasonal currents and upwelling circulation, thereby increasing advection and mixing of nutrient-rich deep water onto the WFS (Weisberg and He 2003), and creating conditions for a phytoplankton bloom. For example, in December 2010 physical forcing from anomalously strong upwelling-favorable winds increased advection of nutrients across the shelf break onto the WFS, and led to a phytoplankton bloom through January 2011 that spanned from Mobile Bay to the Florida Keys (Hu et al. 2011).

In addition, there is indirect evidence from satellite observations and circulation modeling that the DWH oil spill led to the formation of a large, contiguous patch (~11,000 km$^2$) of anomalously high phytoplankton biomass in the northeastern GoM from August to September 2010 (Hu et al. 2011). Several smaller patches were also present in the area and southwest of the Mississippi River, but were not as spatially coherent. Reduced predation pressure from zooplankton grazing or increased nutrient regeneration from dead zooplankton or other organisms may have been responsible for the observed patch, although zooplankton surface densities were highly variable across space (ranging from 10,000 to over 70,000 individuals/m$^3$). An influx of nutrients from upwelled deep water may have also played a role in the phytoplankton bloom; LC interactions with the shelf slope at the Dry Tortugas led to a prolonged upwelling event across the region through the spring and fall (Weisberg et al. 2014a). It is possible that this large increase in available phytoplankton biomass (both in August and again in December of 2010) increased survival and recruitment of some larval and juvenile fishes in the region, and thus contributed to increased productivity of the commercial reef fish fishery after 2010 (these data) as well as to the strong year classes observed for some juvenile coastal fishes (Fodrie and Heck 2011) and Gulf menhaden (Short et al. 2017) in 2010.

*Larval Dispersal Mechanisms*

Successful larval dispersal is a major determinant of population dynamics, and is ultimately driven by interactions between biology (e.g., larval behavior, growth rate, condition, survival rate) and physical circulation properties (e.g., advection, upwelling) at a range of spatial and temporal scales (Cowen and Sponaugle 2009). Seasonally varying wind stress has been described as a determinant of Red snapper larval dispersal in the northern GoM (Johnson et al. 2009); larval transport pathways were modeled to be westward during May, September, and October (under the influence of strong westward wind stress) and eastward during peak spawning months of June, July, and August (under the influence of weaker shoreward wind stress). The varying circulation of WFS bottom currents similarly plays a role in the transport of Gag larvae to inshore settlement habitats. The bottom currents over the WFS are upwelling-favorable during Gag spawning months (i.e., late winter through early spring) and have been demonstrated to transport demersal larvae eastward near to coastal seagrass habitats (Weisberg et al. 2014b). However, interannual variability in the intensity or timing of seasonal upwelling (e.g., from strong winds or interactions with the Loop Current) can influence annual, seasonal, and temporal variability in Gag recruitment success. Significant interannual variability in juvenile

Gag recruitment has been reported by Switzer et al. (2012), who additionally noted a pattern of strong juvenile recruitment every two to four years. The WFS upwelling event in December 2010 (as described above) may have thus acted to enhance Gag larval transport and recruitment success in the eastern GoM, resulting in the observed pattern of increased grouper productivity in the region from 2012 through 2014.

*Harmful Algal Blooms*

Harmful algal blooms in the GoM known as "red tide" typically occur in low nutrient conditions supplied by aeolian dust and local estuarine nutrients, in the presence of colored dissolved organic matter (CDOM) that acts as a sunscreen for the dinoflagellate cells, and with localized wind-driven upwelling that concentrates cells at coastal fronts (Walsh et al. 2003, 2009). Red tides composed of the ichthyotoxic dinoflagellate *Karenia brevis* are now frequent (near annual) events in the GoM, and have been observed off the coast of Texas since at least the 1930's and the Florida coast since the mid-1800's (Walsh et al. 2006). Massive fish kills — some comparable to the yields of a directed fishery — for groups including sardines, menhaden, drums, groupers, and snappers can be attributed to the potent neurotoxins produced by *K. brevis* during red tide events.

There have been significant red tide events in the eastern GoM from fall 2004 through winter 2007,[11] including a severe event that lasted for nearly 13 months off Florida's coast from January 2005 to February 2006 (Flaherty and Landsberg 2011). The 2005 event started offshore along west-central Florida and eventually reached more than 1,300 km$^2$ in surface area, spanning from Tampa Bay to the Florida-Alabama border in June 2005 (FWRI 2018). The event

[11] Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute. 2017. *HAB Monitoring Database*. Available online at: http://myfwc.com/research/redtide/monitoring/database/.

contributed to the formation of widespread hypoxic zones off west-central Florida, in Tampa Bay, Sarasota Bay, and Charlotte Harbor in July-August 2005 (Hu et al. 2006), and affected benthic communities in an estimated 5,600 $km^2$ area (FWRI 2018). The extent and duration of the event caused mass mortalities for benthic communities, sea turtles, marine mammals, birds, and fish, including Gag, Red grouper, and Red snapper.[12]

In this work, productivity for groupers was significantly lower overall in 2005 than either 2004 or 2006 (Figure 2.7); the eastern GoM displayed the same pattern but 2005 differences were not statistically significant (Figure 2.9). The broad spatial scale of the eastern zone, however, may be dampening finer-scale regional impacts of the 2005 red tide on fishery productivity. Snapper and grouper productivity in the western GoM similarly decreased in 2005 compared to 2004 and 2006, although grouper CPUE was the only metric with a statistically significant change (Figure 2.9). Given the location of the red tide events in the eastern GoM, any decreases in western productivity attributable to red tide would likely manifest through indirect trophic mechanisms. Grazing on *K. brevis* by some copepod species may pass brevetoxins on to higher trophic levels (Walsh et al. 2003), and seagrass, shellfish, and omnivorous and planktivorous fish have been found to act as brevetoxin vectors for higher trophic levels (including marine mammals, birds, and humans) through retention and accumulation in their tissues (Landsberg 2002, Flewelling et al. 2005, Naar et al. 2007). More complex ecological dynamics may occur as the result of red tide events as well. For example, if *K. brevis* affects competitors, per-capita consumption rates may increase after competition for prey is reduced (e.g., Sagarese et al. 2015). Productivity of younger fish (typically with faster growth rates) may

[12] Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute. 2018. *Fish Kill Database Directory.* Available online at: http://myfwc.com/research/saltwater/health/fish-kills-hotline/. Searched results for all counties as the result of red tide from January 1, 2005 through February 28, 2006.

similarly increase if red tide mortality disproportionally affects older individuals. These types of compensatory mechanisms may in part explain the rapid productivity increases observed here for snappers and groupers after 2006.

On the other hand, deep water upwelling over the shelf break brings inorganic nutrients to otherwise oligotrophic surface waters and *suppresses* red tide blooms, because increased nutrients allow for the growth of phytoplankton species that can outcompete *K. brevis* (Walsh et al. 2003, 2009). In 1998 and 2010, LC interactions with the shelf slope at the Dry Tortugas led to strong and prolonged upwelling across the WFS from spring through the fall. This LC mediated upwelling has been implicated in the absence of a red tide on the WFS in those same years (Weisberg et al. 2014a). Thus, the anomalous physical circulation of the WFS in 1998 and 2010 may have acted to indirectly enhance survival and recruitment of several fish species through release from red tide mortality.

Due to the narrow shelf width and onshore direction of bottom currents, the region between Tampa Bay and Charlotte Harbor is one of maximum near-shore upwelling (Weisberg et al. 2000). This region is also one of prolific Gag recruitment (Weisberg et al. 2014b, Switzer et al. 2012) and a major epicenter of recurring red tide events in the eastern GoM (Walsh et al. 2006). If Gag larvae (spawned from late winter through early spring) and *K. brevis* were concentrated together near the coast, there could be serious negative consequences for the population and fishery in subsequent years. Changes in fish abundance, community structure, and declines in juvenile recruitment were reported for several species in the Tampa/Sarasota region after the 2005 red tide event (Gannon et al. 2009, Flaherty and Landsberg 2011). At the same time, chronic exposure to brevetoxins through the diet could lead to impaired feeding, growth, immune function, behavior, or reproduction for organisms at all trophic levels throughout the ecosystem (Landsberg 2002 and references therein). In the long term, chronic exposure to toxins from harmful algal blooms poses a major threat to ecosystems and fisheries sustainability.

**Implications**

While this dissertation is a step toward understanding spatiotemporal changes in productivity, distribution of fishing effort, and resilience in this fishery, it could also serve as a bridge to more rigorous and data-intensive modeling of responses to sudden disturbances (e.g., oil spills or emergency regulatory rules) as well as gradual changes in spatial management (e.g., implementation of marine protected areas). This work includes only a portion of the GoM reef fish fishery and most certainly only a small portion of overall fishing in the GoM (Figure 5.1). For instance, the total (vessel standardized) 2010-2014 revenue and landings for the data used for this work totaled \$43.2M (adjusted to \$2008) and 14.1 million gutted pounds. In comparison, NOAA fisheries reported \$3.8B in commercial revenue and 1.8 billion lbs. in total commercial landings for all GoM key species or groups[13] over the same time period (NMFS 2016a). Nevertheless, major key species such as Red snapper, Vermilion snapper, and groupers were included in the data used here; in 2015, Red snapper and Vermilion snapper were two of the top ten commercial species landed Gulf-wide (NMFS 2016b).

The patterns detailed here must be studied further, and with a more discrete eye toward other regulatory changes that occurred around this same time. For example, in 2010 Amendment

[13] Key species and groups for the commercial sector include Blue crab, Stone crab, Crawfish, Red snapper, groupers, mullets, oysters, shrimp, and tunas. Menhaden were not included in the total values reported here, since the group represents a disproportionate fraction of the commercial fishery (e.g., ~73% of total key species landings and ~10% of key species revenue from 2010-2014). Menhaden contributed 5 billion lbs. in landings and \$394M in revenue from 2010-2014 (NMFS 2016a).

31 to the Reef Fish Management Plan (GMFMC 2010) established a series of new regulations for longline fishers in the eastern GoM. These included the longline seasonal closure (June-August) for reef fish, a longline endorsement for the reef fish permit (requiring average annual reef fish landings of at least 40,000 pounds from 1999 through 2007 to qualify), and a gear limitation of 1,000 hooks per vessel per trip, of which no more than 750 can be rigged for fishing or fished. These regulations almost certainly impacted the ability of individual longline fishers to remain in the fleet after 2010, independent of any effects from the DWH oil spill or implementation of the Grouper-Tilefish IFQ in 2010. In the Red snapper fishery, harvest reduction regulations have led to accelerated recovery rates of Red snapper since 2007 (Gallaway et al. 2017). Juvenile bycatch mortality has also been reduced in part due to shrimp trawling effort reductions as mandated in the 2004 Red snapper rebuilding plan,[14] although shrimp trawling effort has declined primarily from adverse economic conditions (i.e., high fuel costs and reduced prices from cheap imports; Gallaway et al. 2017). Note, however, that the Red snapper IFQ was also implemented in 2007 and recovery patterns from pre-IFQ regulations are likely confounded with IFQ impacts. Additional concomitant regulatory changes included a 164-day emergency turtle bycatch reduction closure for longline fishers in the eastern GoM in 2009 (NOAA Fisheries Service 2009). In addition, the GMFMC recently voted to remove the 1,000 hook limit on longline vessels that was established with Amendment 31 (GMFMC 2017), thus creating an interesting opportunity to compare longline effort distribution, fleet economics, and productivity before, during, and after the hook limitation regulation.

A wider range of fisher characteristics, decisions, and outcomes should be included in future modeling work as well. For instance, fishers that returned to the fishery after a hiatus,

[14] The GMFMC approved an updated Red snapper rebuilding plan in 2004 with Amendment 22 to the Reef Fish Fishery Management Plan (GMFMC 2004).

moved to the recreational for-hire sector, or transitioned into working in other fishery-related or non-fishing sectors. It is feasible that fishers moved into other jobs in the fishing industry for a period of time, and returned to fishing after the initial environmental and economic risks from the oil spill had dissipated. To be sure, there are a variety of social and economic drivers behind the decision to stay active in a commercial fishery. These can include fishing income diversification and income stability; market channel relationships with processors, fish markets, restaurants, and others; age, health, and disability status; education level, experience, and skills that can be transferred to non-fishing sectors; other household income and employment opportunities; and the location of job opportunities outside of fishing relative to household mobility (Hackett et al. 2015). These data could be obtained through surveys, interviews, or workshops within fishing communities, or quantified with proxies from existing fishery datasets. Discrete choice modeling could be used to approach these questions, and has been used in the past to understand displacement and spatial shifts in fishing effort after closures (Valcic 2009), participation choice in a multispecies fishery (Larson et al. 1999), long-term decisions such as entry and exit (Ward and Sutinen 1994), and shorter-term daily decisions of where to fish (Hicks and Schnier 2005). Recent econometric work (Zhang and Smith 2011) on the GoM reef fish fishery — using captain survey data and a similar logbook data set to that used here — revealed that travel costs, species price, captain age, and perceptions on the effectiveness of marine reserves were all drivers of fishing behavior and choice of fishing grounds after implementation of two marine reserves.

In the case of the DWH oil spill, fishers may have opted to aid in oil spill response through the Vessels of Opportunity (VoO) program. Approximately 3,500 commercial and charter boats were employed in the VoO program over its lifetime (Upton 2011), with payments of $283 million paid out to 5,401 individual fishers (Deepwater Horizon Claims Center 2018; Figure 5.1). An additional $2.2 billion was paid out to commercial fishers, crew, and vessel owners as part of the Seafood Compensation Program, and all economic and property damages compensation totaled $10.4 billion (Deepwater Horizon Claims Center 2018). While some may have opted to leave fishing altogether in exchange for a monetary settlement, the compensation payments possibly gave others some financial security and incentive to remain in the fishery during a very uncertain and risky period of time (see also the Discussion in Chapter 4). A more focused analysis of the relationship between DWH emergency compensation and the decision to remain in the fishery is warranted. At the time of this work, data on individual participation in the VoO program, transition to other job opportunities, or specific compensation amounts for individual fishers were not available.

**Final Thoughts**

There is a growing body of literature that suggests that the acute population-level impacts of the DWH oil spill were not as severe or significant as might have been expected. Fodrie and Heck (2011) and Schaefer et al. (2016) concluded that the oil spill did not significantly impair the community of northern GoM coastal fishes examined at the ecosystem level, and no significant post-spill shifts in community composition, structure, or biodiversity were observed. Peterson et al. (2017) similarly concluded that the DWH oil spill did not significantly impact the abundance or food-web structure of large coastal fishes in the Florida Big Bend. To be sure, the impacts of DWH will propagate through the GoM ecosystem over different time horizons and with different outcomes for individual populations or systems. Economic and environmental impacts will likely be more severe and require longer recovery time for benthic fishery species such as shrimp and shellfish (Sumaila et al. 2012).

It has been previously estimated that the DWH oil spill will result in over 22,000 lost jobs and have a total economic impact of $8.7 billion through 2019 for mariculture, commercial, and recreational fisheries in the GoM (with total economic impact to commercial fisheries estimated at $4.9 billion; Sumaila et al. 2012). While the full scope of population- and fisheries-level responses to the DWH oil spill may take many years to be realized, it appears from this work that the resilience and recovery of this fishery have been better than initially anticipated. This conclusion is supported by the following results:

(1) Productivity for snappers, groupers, and total species were generally either stable or increased after 2010 (*Chapter 2*);

(2) Regional productivity patterns and relative productivity for trips inside *vs.* outside closure boundaries were constant over time (*Chapters 2 and 3*);

(3) Fleet-level and gear-specific fishing grounds were similar before and after DWH (including the location of trips relative to closure boundaries), with shifts in effort after closures onto the West Florida Shelf (*Chapters 3 and 4*); and

(4) A lower rate of vessel attrition after the DWH closures than would be expected based on the background rate (*Chapter 4*).

It is important to note that the conclusion of resiliency for *this particular fishery* is not meant to negate or trivialize the loss of jobs, income, or financial stability that resulted from the oil spill for many businesses, families, and coastal communities across the GoM.

Ultimately, understanding the factors that contribute to vulnerability, resilience, and response of fishers to regulations and disturbance will improve decision making about fisheries resources. This dissertation stands to make a significant contribution to our understanding of how the DWH oil spill impacted fisheries and communities in the GoM. This work will also contribute more broadly to our understanding of how large-scale perturbations are absorbed and propagated through coastal marine fisheries.

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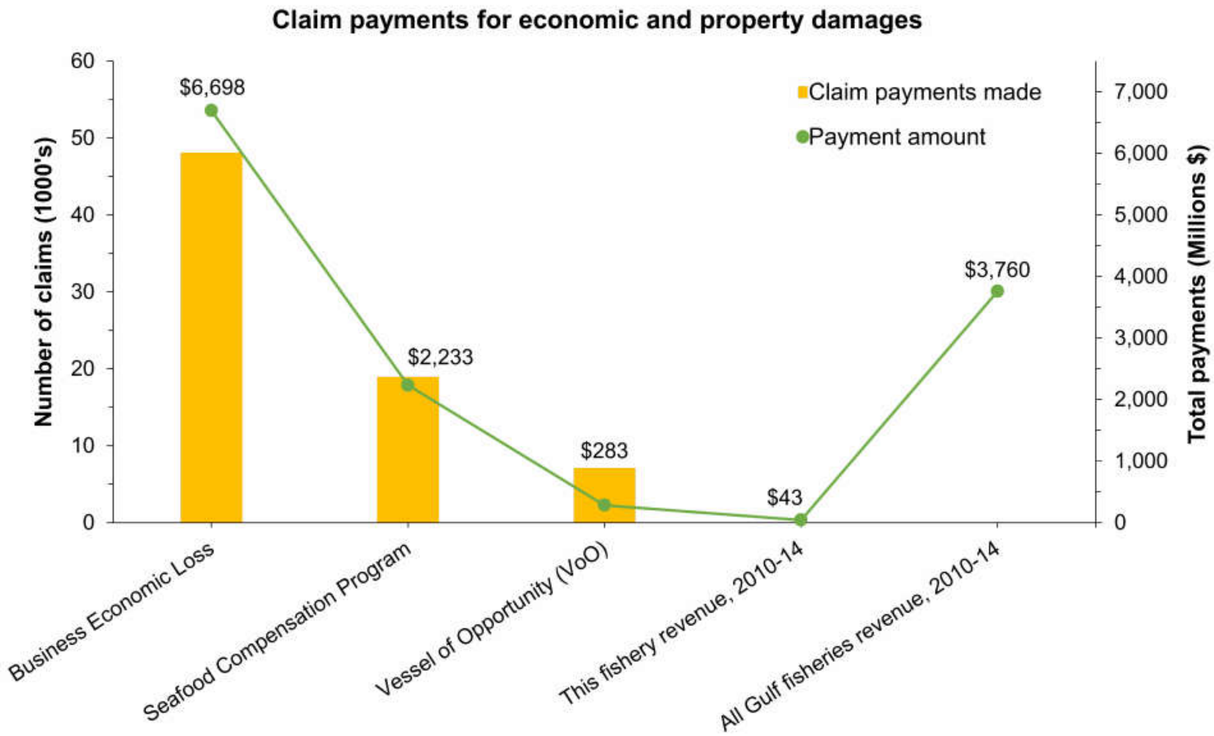


**Figure 5.1.** *Deepwater Horizon* economic and property damage claim payments for a subset of claim categories, with a comparison of the total revenue for this fishery and total revenue for GoM key species from 2010-2014. The total number of claims paid out in each category is shown with the bars (in thousands; primary axis) and the total payment values are shown with the green line (in millions; secondary axis) and the numbers above bars. Claims data were obtained from the Deepwater Horizon Claims Center (2018).

# APPENDICES

## Appendix A. Logbook Data Selection

Logbook records were filtered by first eliminating trips that reported no reef fish landings, revenue, vessel length, or effort (i.e., average hooks per line, average number of lines used, or total number of sets). Duplicate records were removed based on the unique trip identifier assigned to records by the NOAA NMFS Southeast Fisheries Science Center. Only trips reporting longline or vertical line (handline or bandit-reel) as the top-revenue producing gear were used in analyses, since these are the main gears used in this fishery (Scott-Denton et al. 2011), and represent 89% of logbook-reported trips from 2000-2014. Only data for snappers, groupers, tilefish, jacks, and triggerfish were used to quantify catch and revenue; total revenue and total landings were calculated for each trip as the sum of reported values for the species in these five groups. Total landings and revenue for snappers and groupers were also calculated similarly, using only the species in the respective group (see Table 1.2). Trips that had zero total calculated landings were eliminated (under the assumption that these trips were not targeting reef fish), and remaining records had a constant of 0.1 added to the value for snapper and grouper landings and revenue in order to permit natural log ($\log_e$) transformation of data.

Catch-per-unit-effort (CPUE) was calculated for each logbook trip as the total landings (or total snapper or grouper landings) divided by the total number of hooks fished. Total revenue, snapper revenue, and grouper revenue for each trip were inflation adjusted to 2008 US dollars ($2008) using the United Nations Food and Agriculture Organization (FAO) fish price index (FPI) price series (Tveterås et al. 2012). Analogous to a consumer price index, the FPI collapses price and quantity information into one number that tracks change in seafood price as a whole. The FPI is an improvement over other food commodity indices, however, in that it incorporates aquaculture production, import and export flows, and the extent of international trade competition for 608 unique trade data categories of fish and seafood. The value of the FPI for 2008 (FAO value of 136) was set to 100 as the standard and all other values were scaled accordingly (i.e., multiplied by 0.74).

Two outliers for CPUE and one for landings were identified and removed using the Grubb's test in the *outlier* package in R (Komsta 2011). Vessels that had anomalous gaps in their VMS reporting frequency were also eliminated (see Appendix B), for a total of n=96,668 trips in the final logbook data set. All revenue, landings, and CPUE data were $\log_e$-transformed in order to linearize relationships and meet normality assumptions of linear regression and ANOVA.

Total, snapper, and grouper landings, CPUE, and inflation-adjusted revenue were additionally standardized to account for effects of vessel size, as larger vessels have the capacity to hold more fish, make longer trips, and therefore report greater landings or revenue overall. To eliminate this potential confounding factor, data were divided by a fishing power coefficient, calculated based on empirically-determined vessel size categories and ANOVAs of $\log_e$-transformed values *vs.* vessel size category (Murawski et al. 2005; see Appendix C).

## Appendix B. Vessel Monitoring System Reporting Frequency Quality

Vessel monitoring system (VMS) locations are meant to be reported approximately every hour for the duration that the transponder is active. However, there were vessels within the dataset that did not adhere to this reporting frequency. The VMS dataset is patchy in places, with some vessels having highly irregular tracking records; including these data in the analysis would add considerable noise for limited benefit. In order to eliminate lower quality reporting vessels, a test was run to determine which vessels had the best quality data reporting. A linear model was fit to each individual vessel's VMS record (from 2006-2013), with ordered record number as the predictor and timestamp as the response. Using this approach, a perfect VMS record would make a straight line and a fitted model would have an $r^2$=1. Conversely, gaps in the VMS record will cause discontinuities in the ordered data series with a resultant decrease in the $r^2$ value of a fitted regression model for that vessel. The VMS dataset used in this study was filtered by selecting vessels with a high reporting frequency quality $r^2$ (i.e., a regular reporting record; $r^2$=0.75 or greater) and 10 or greater total VMS records. There were 1,302 unique vessels in the VMS data set to start, reduced to 1,104 unique vessels once the data reporting quality filter was applied (~85% of all unique vessels). The mean reporting frequency quality $r^2$ value for these vessels was 0.96.

## Appendix C. Logbook Data Standardization to Vessel Size

Fisher's Natural Breaks Classification was applied to the selected logbook data (see Appendices A-B) from 2000-2014 (n=96,668 records) using the *classInt* package in R (Bivand et al. 2015) to obtain the size bins for four vessel size classes (Table C.1). Each trip was then assigned to a vessel class based on the reported length of the vessel (in meters). A linear regression for the effect of vessel length on landings, CPUE, and revenue for the total, snapper portion, and grouper portion were run separately for each variable to calculate the fishing power coefficients, with vessel class 1 as the baseline in each test. All variables were linearized with a $\log_e$ transformation before running the regressions. The formula for calculating the coefficient for each vessel size class was:

$$(1)\ e^{\text{regression coefficient estimate}+(0.5\times\text{coefficient std. error})}$$

The untransformed original values for landings, CPUE, and revenue were then divided by the appropriate fishing power coefficient (Table C.1) to obtain the final vessel-class standardized values. To ensure that there was no relationship between the standardized variables and assigned vessel class, an ANOVA was run on $\log_e$-transformed standardized variables *vs.* vessel size class after each transformation and confirmed visually with boxplots and scatterplots.

**Table C.1.** Fishing power coefficients used for standardizing variables to vessel size.

| Vessel length (m) | Vessel category | Total revenue | Total landings | Total CPUE | Snapper revenue | Snapper landings | Snapper CPUE | Grouper revenue | Grouper landings | Grouper CPUE |
|---|---|---|---|---|---|---|---|---|---|---|
| [4.87-9.9] | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| (9.9-12.63] | 2 | 3.26 | 3.18 | 1.02 | 2.00 | 1.93 | 0.62 | 10.12 | 8.28 | 2.66 |
| (12.63-16.46] | 3 | 7.12 | 7.06 | 0.12 | 5.61 | 5.27 | 0.09 | 10.89 | 9.07 | 0.15 |
| (16.46-26.21] | 4 | 9.15 | 8.92 | 0.15 | 22.07 | 18.56 | 0.32 | 1.90 | 1.81 | 0.03 |

**Note:** Vessel size class 1 was the standard for all calculations. All size class intervals were closed on the right

## Appendix D. Vessel Monitoring System Data Selection

The VMS data (Rivero 2015) were filtered to retain only active fishing points, as determined by: (1) linking VMS points to logbook trips based on a unique vessel identifier and trip start and end dates, (2) filtering VMS-logbook linked data to the times of peak fishing activity as quantified in observer data (see Appendix E), and (3) applying empirically-determined "speed filters" based on the cumulative distribution of ranked vessel speeds (see Appendix F). After identifying fishing activity with the time and speed filters (based on gear type used), an additional three nautical mile (5.56 km) coastal "buffer" was added to avoid false positives for fishing activity near the coast. Points that were on land (likely representing a VMS system turned on early or left on after a trip), outside the GoM basin, or deeper than 2000-m (where reef fishing activity is highly unlikely to occur) were also eliminated. Vessels that had anomalous gaps in their VMS reporting frequency were removed (see Appendix B), and vessels with fewer than three VMS points after filters were applied were removed from the data set to allow for proper calculation of spatial metrics.

## Appendix E. Defining Peak Fishing Activity in Observer Data

Mandatory onboard observer efforts began in 2006 for the commercial reef fish fishery (Scott-Denton et al. 2011). Observer data were available for bottom longline (n=5,839) and bandit-reel (n=20,646) trips. Fishing activity in the observer data was defined as the time spanning the recorded start of a gear set plus the recorded soak time (for bottom longline) or fishing time (for bandit-reel). The median number of observations for set start times and set end times were calculated, the earliest set start and latest set end time corresponding to the median frequency were determined, and an additional 30 minute "deviation window" was added on either side to get the final fishing time window. Fishers are often engaged in fishing activity before and after gear deployment and actual recorded set times; the deviation window was added to account for this possibility as well as individual VMS pings that may have been transmitted just before or after the recorded set times and would otherwise be considered "non-fishing." Complementary work to this study has also revealed that algorithms trained to identify fishing activity with this "window-labeling" method are more accurate in positively identifying fishing behavior than traditional "point-labeling" methods, especially in fisheries using gears such as bandit-reel that have short-duration sets (O'Farrell et al. 2017). The majority of fishing activity as recorded in the observer data set was between 0615 and 2245 hours for bottom longline (97% of n=5,839 observer records) and between 0745 and 2000 hours for bandit-reel (92% of n=20,646 records; Figure E.1). The same peak activity time window was used for both vertical line gears (i.e., bandit-reel and handline).

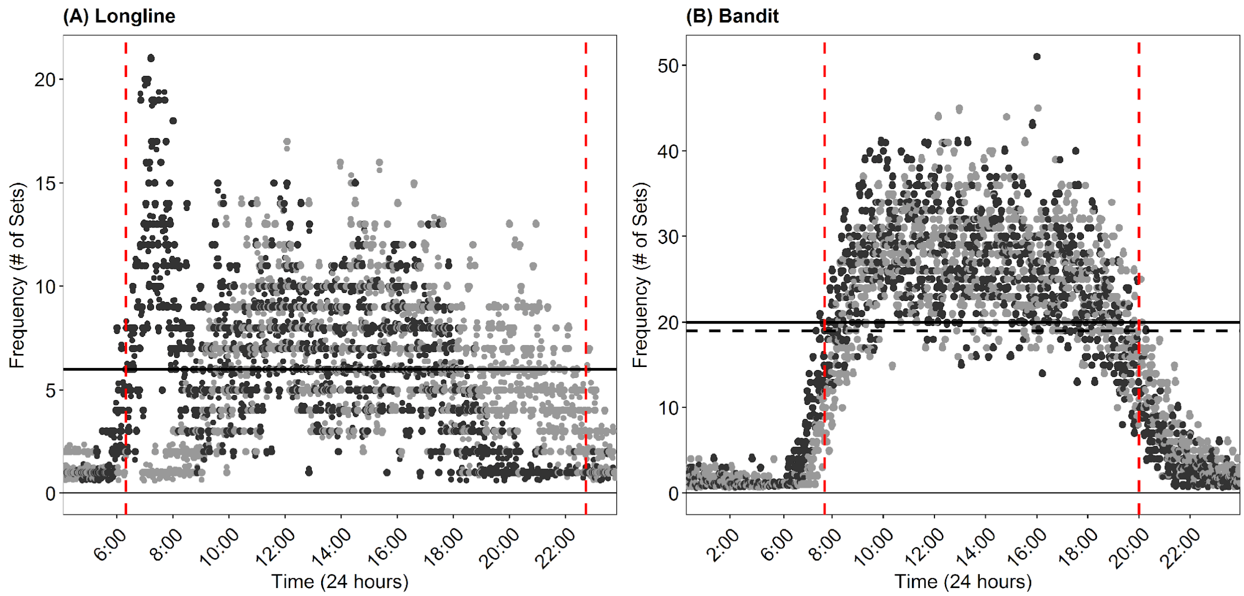


**Figure E.1.** Diel pattern of fishing sets from 2008-2012. (A) Bottom longline and (B) bandit-reel observer data. Times at the start of sets are marked with black circles and times at the end of sets are marked with grey circles. Red vertical lines denote the time window used to filter VMS points for fishing activity. Time windows were chosen based on correspondence with the median number of observations for set start (solid horizontal line) and end times (horizontal dashed line). Median values are equal for bottom longline. The y-axis has been jittered to better visualize overlapping points of observations made at the same times.

**Appendix F. Determining Fishing Speed Filters**

Speed filters were determined empirically from the data set. First, VMS points and logbook records were linked by using unique individual vessel identifiers and the start and end date of the logbook trips; only those points that fell on or within the dates of each logbook trip for a given vessel were kept. Second, these VMS-logbook linked records were matched to bottom longline and/or bandit-reel observer data based on the top gear reported in logbooks, unique vessel ID, date, and times of peak observed fishing activity (see Appendix E). This resulted in only VMS points that were matched to logbook trip dates as well as fishing activity as reported by observers. Records were further filtered by eliminating records with speeds equal to zero or in excess of 20 m/s, and keeping only records for vessels that met the minimum reporting frequency quality (see Appendix B).

Next, the cumulative distributions of calculated VMS speeds from 2008-2012 were used to empirically determine the speed rules for bottom longline and bandit-reel gear fishing activity. The VMS speed data were ranked; the lower and upper 5% of the data were subsequently removed, keeping only the middle 90% of calculated vessel speeds (Figure F.1). The mean speed from the middle 90% of the distribution (Figure F.2) as well as boxplots of the data (Figure F.3) helped visualize the range of vessel speeds and determine the speed rules. The speed rules were chosen to include the overall mean value for each gear, as well as the majority of speeds that fell within the middle 90% of the cumulative distribution.

Based on the ranked data and cumulative distributions of VMS speeds, speed rules of 1-4 m/s were used for both longline and vertical line gears (representing ~35% and 20% of the logbook and observer matched VMS data, respectively). The average (mean ± SEM) calculated speed from the middle 90% of ranked data from 2008-2012 for bottom longline and bandit-reel gears was $2.04 \pm 0.008$ m/s and $1.19 \pm 0.01$ m/s, respectively. Bandit-reel had an especially large variation in mean vessel speed from year to year.

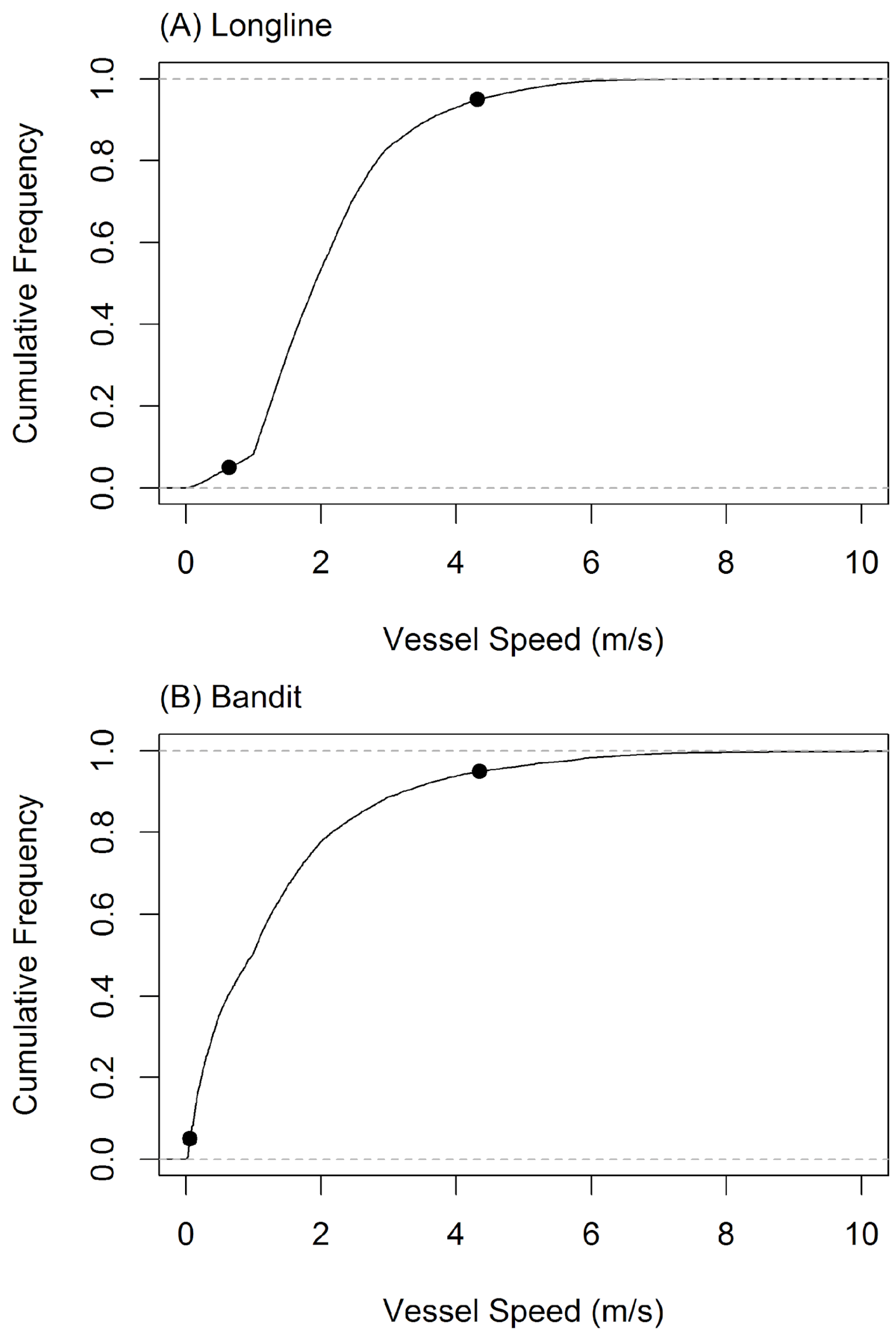


**Figure F.1.** Cumulative frequency distributions of calculated VMS vessel speeds. From 2008-2012 VMS data matched to logbook trip dates and peak time of fishing activity as quantified in observer data. (A) Trips reporting bottom longline as the top gear, and (B) trips reporting bandit-reel as the top gear. Circles on the curves denote the speeds corresponding to the lower and upper 5% of the distribution.

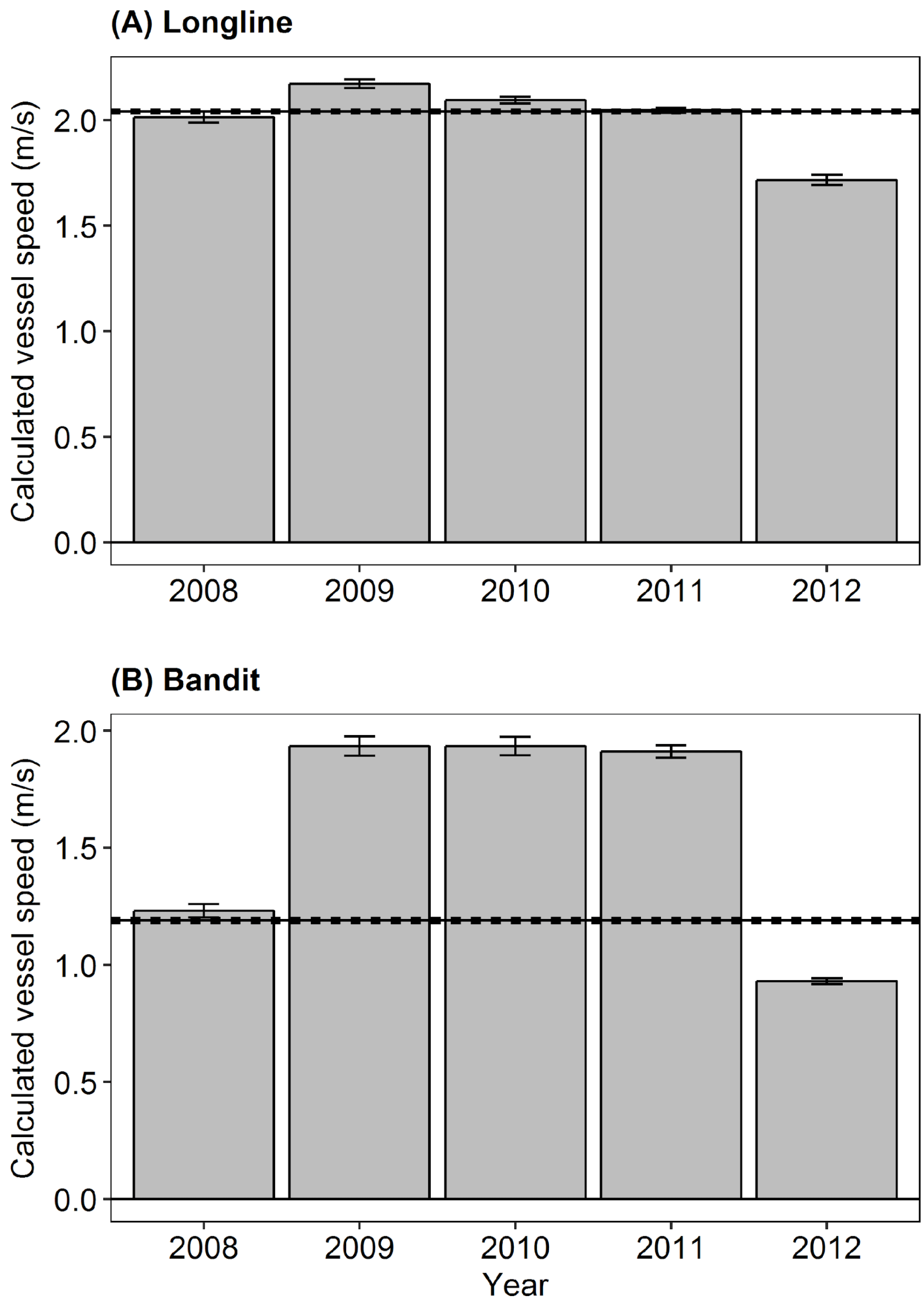


**Figure F.2.** Calculated vessel speeds for the middle 90% of ranked VMS speed data. From 2008-2012 VMS points matched to logbook trip dates and peak time of fishing activity as quantified in observer data, for (A) bottom longline (mean ± SEM) and (B) bandit-reel gears (mean ± SEM). The horizontal solid and dashed lines represent the overall mean and SEM, respectively, for the respective gear.

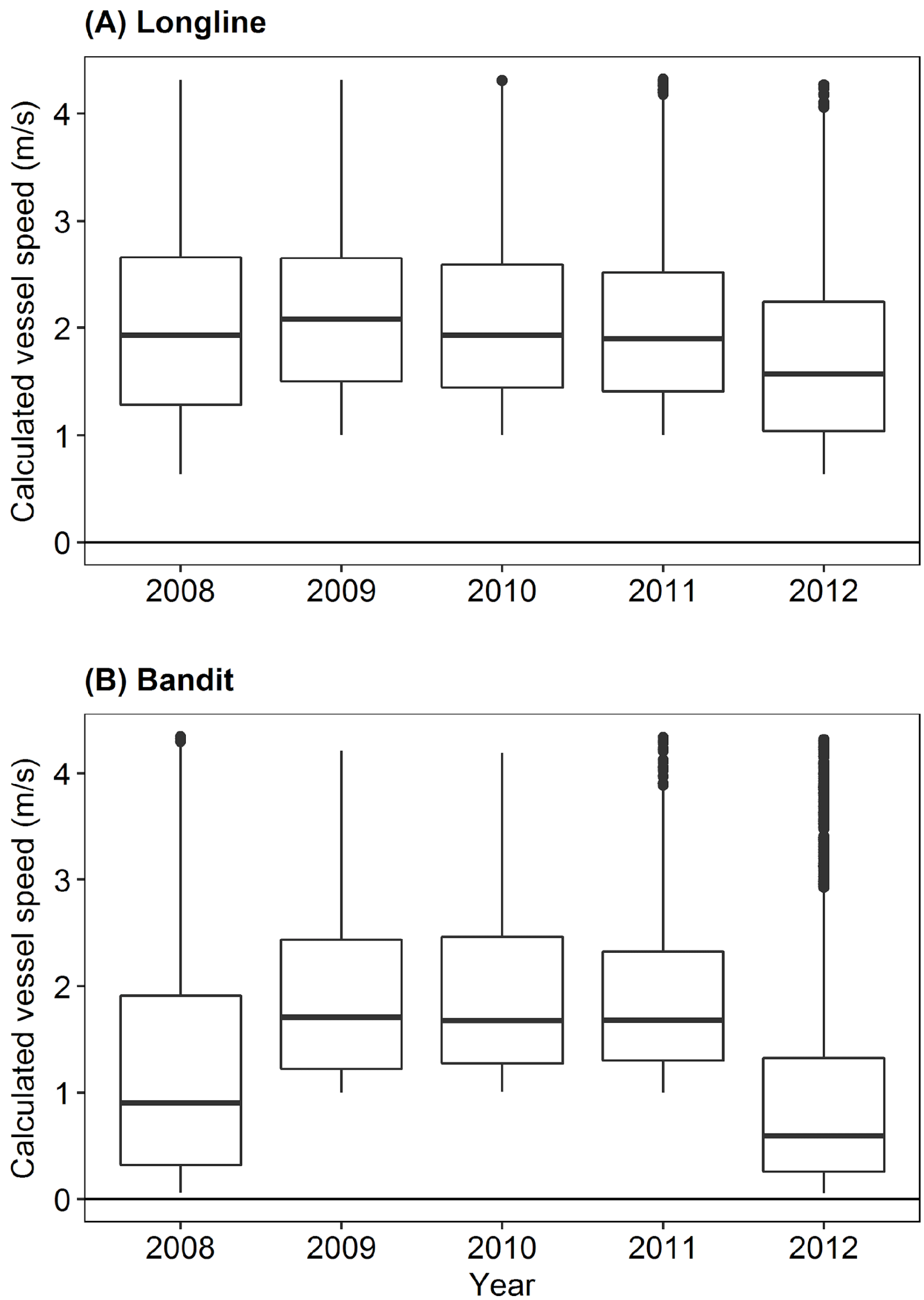


**Figure F.3.** Boxplots of calculated vessel speeds for the middle 90% of ranked speed data. From 2008-2012 VMS points matched to logbook trip dates and peak time of fishing activity as quantified in observer data for (A) bottom longline and (B) bandit-reel gears. Whiskers for each year are 1.5 times the interquartile range.

## Appendix G. NOAA Fisheries Logbook Reporting Grids

Logbooks require reporting of the top area fished (i.e., the area producing a plurality of revenue for the trip) based on statistical grid areas (Figures G.1 and G.2). In 2013 and later, the grid areas were on a much finer scale (Figure G.2). These grid areas were used to categorize trips into west, central, and east regions for an analysis of productivity over space (see Chapter 2, Figure 2.1). Maps of logbook reporting grids were obtained online through NOAA Southeast Fisheries Science Center *Fishermen and Seafood Dealers Forms Archive* for this purpose (www.sefsc.noaa.gov/fisheries/reporting_archive.htm).

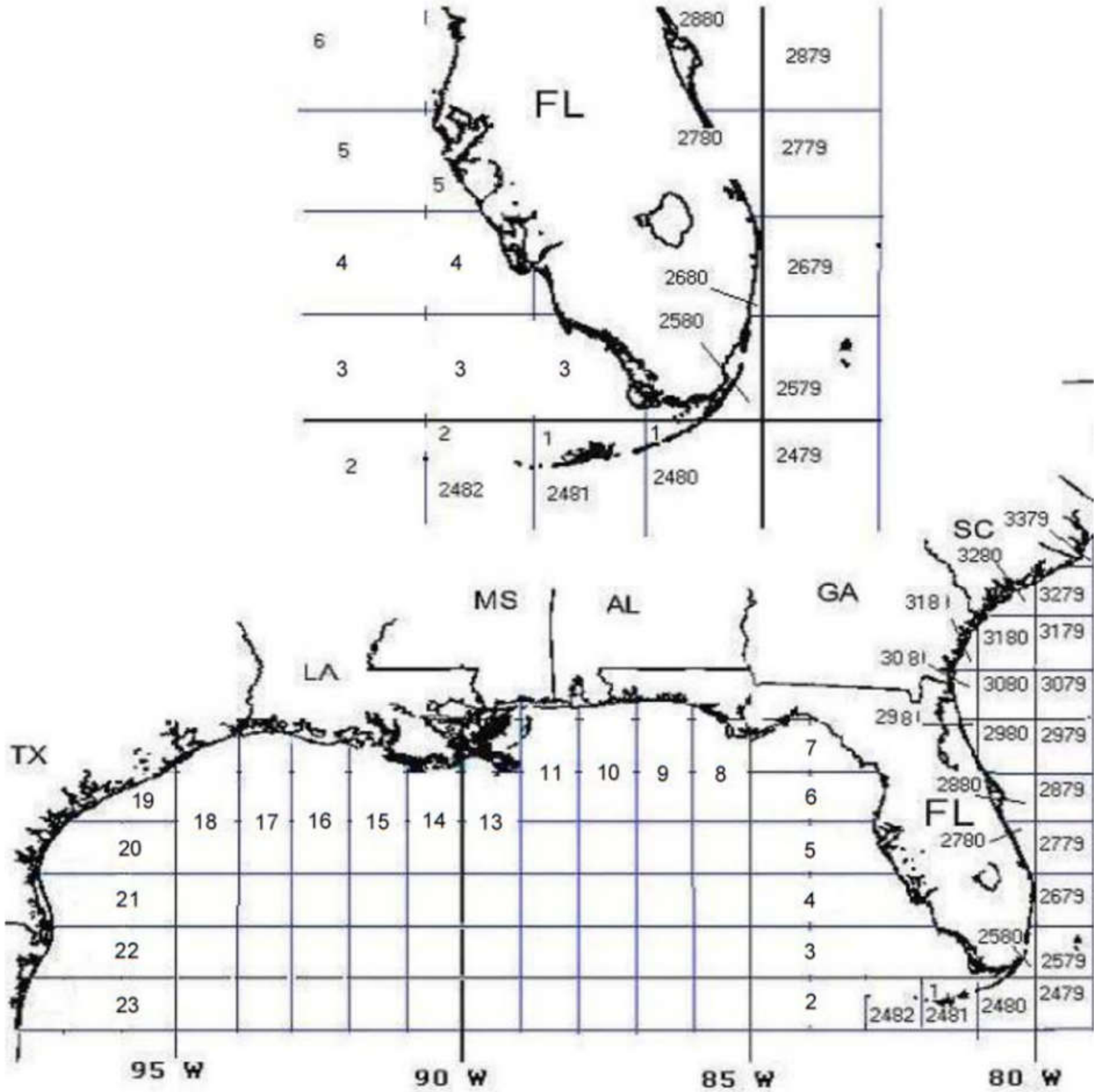


**Figure G.1.** NOAA Fisheries logbook statistical reporting grids for years 2012 and earlier. Available from www.sefsc.noaa.gov/fisheries/reporting_archive.htm.

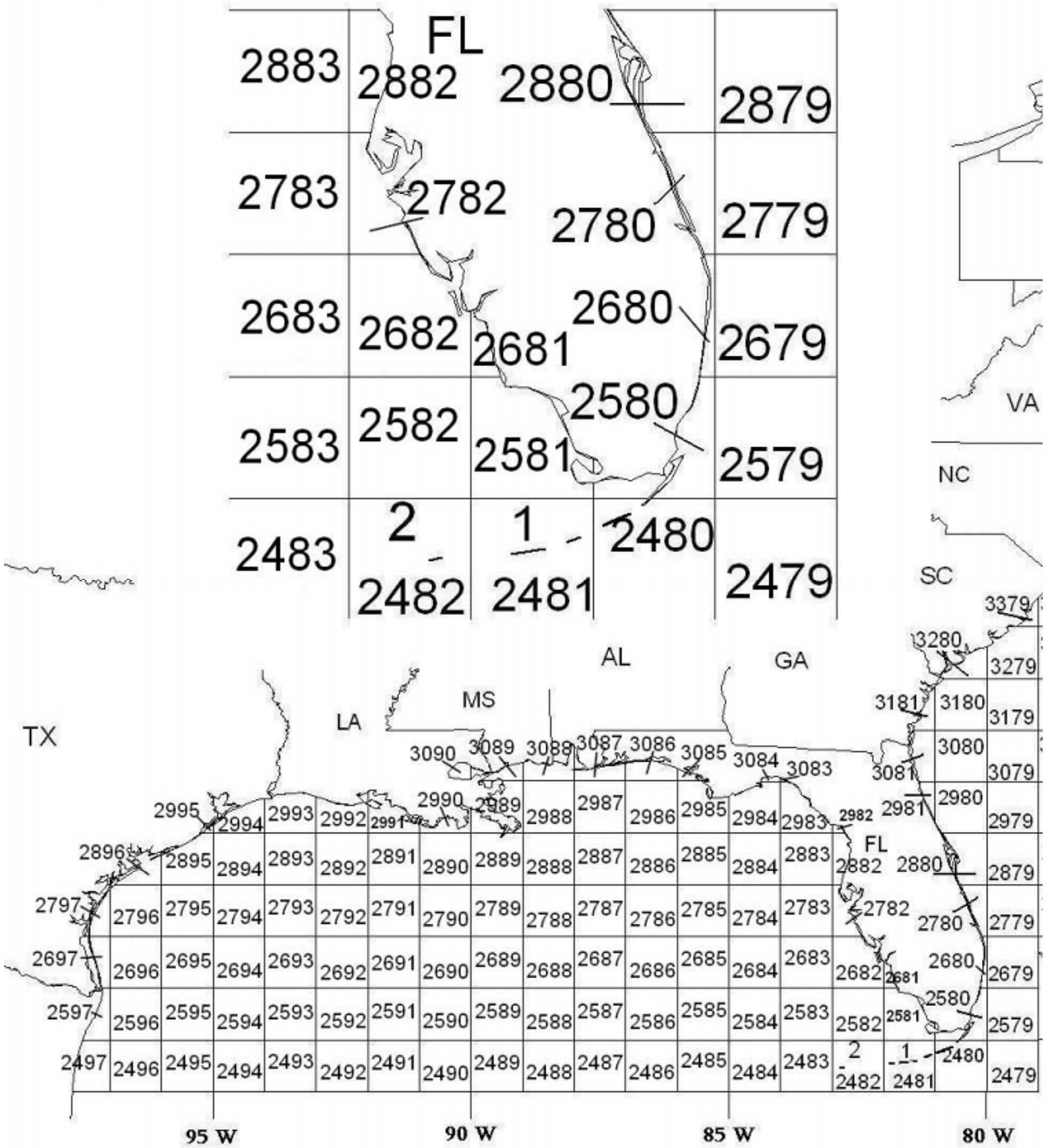


**Figure G.2.** NOAA Fisheries logbook statistical reporting grids for years 2013 and later. Available from www.sefsc.noaa.gov/fisheries/reporting_archive.htm.

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