

**A peer-reviewed version of this preprint was published in PeerJ on 16 August 2017.**

[View the peer-reviewed version](https://doi.org/10.7717/peerj.3680) (peerj.com/articles/3680), which is the preferred citable publication unless you specifically need to cite this preprint.

Deis DR, Fleegeer JW, Bourgoïn SM, Mendelssohn IA, Lin Q, Hou A. 2017. Shoreline oiling effects and recovery of salt marsh macroinvertebrates from the *Deepwater Horizon* Oil Spill. PeerJ 5:e3680  
<https://doi.org/10.7717/peerj.3680>

# Effects of shoreline oiling on salt marsh epifaunal macroinvertebrates

Donald R Deis, John W Fleeger, Stefan M Bourgoin, Irving A Mendelssohn, Qianxin Lin, Aixin Hou

The *Deepwater Horizon* oil spill resulted in varying degrees of oiling in the salt marshes of northern Barataria Bay, Louisiana, USA. This study examines the effects of oiling intensity and recovery on two conspicuous marsh-platform macroinvertebrates, *Uca* spp., fiddler crabs, and *Littoraria irrorata*, the salt marsh periwinkle, from 2.5 to 4.5 years after the spill. The dominant fiddler crab within these marshes, *Uca longisignalis*, was the only species observed in field collections, and no significant difference in burrow density or burrow size was found among oiling levels over the study period indicating recovery from any negative effects of oiling already occurred for this species. The highest density of *L. irrorata* was found at moderately oiled sites compared to both reference (without visible oiling) and heavily oiled stations. *Spartina alterniflora* density recovered within two years after the spill at the moderately oiled stations facilitating recovery of *L. irrorata* approximately one year later. *L. irrorata* average shell length and length-frequency distributions were equivalent at moderately oiled and reference stations but snails were shorter at heavily oiled stations because of a greater proportion of subadults. Shell length data from the heavily oiled sites indicate that direct mortality due to oiling or oil-induced reductions in recruitment occurred in 2010 and that recovery was starting to occur at 48 months after the spill. The extent and duration of oil in the water during the spill and the biological responses we measured indicates that *L. irrorata* and *Uca longisignalis* were both affected in their ability to carry out their life cycle on the marsh and/or in the water column at all stations including the reference stations for some period following the entry of oil into the region.

# Title Page

Effects of Shoreline Oiling on Salt Marsh Epifaunal Macroinvertebrates

Donald R. Deis<sup>1\*</sup>, John W. Fleeger<sup>2</sup>, Stefan, M. Bourgoïn<sup>3</sup>, Irving A. Mendelssohn<sup>4</sup>, Qianxin Lin<sup>4</sup>, and Aixin Hou<sup>5</sup>

<sup>1</sup>Atkins , Jacksonville, Florida 32256, US; <sup>2</sup>Department of Biological Sciences, Louisiana State University, Baton Rouge, Louisiana 70803, US; <sup>3</sup>Atkins , Tallahassee, Florida 32256, US; <sup>4</sup>Department of Oceanography and Coastal Sciences, Louisiana State University, Baton Rouge, Louisiana 70803, US; <sup>5</sup>Department of Environmental Sciences, Louisiana State University, Baton Rouge, Louisiana 70803, US

\* Corresponding author phone: (904) 363-6992; fax: (904) 363-8811; email: [don.deis@atkinsglobal.com](mailto:don.deis@atkinsglobal.com)

Keywords: *Deepwater Horizon* oil spill; *Uca*, fiddler crab, *Littoraria irrorata*, marsh periwinkle

# Abstract

The *Deepwater Horizon* oil spill resulted in varying degrees of oiling in the salt marshes of northern Barataria Bay, Louisiana, USA. This study examines the effects of oiling intensity and recovery on two conspicuous marsh-platform macroinvertebrates, *Uca* spp., fiddler crabs, and *Littoraria irrorata*, the salt marsh periwinkle, from 2.5 to 4.5 years after the spill. The dominant fiddler crab within these marshes, *Uca longisignalis*, was the only species observed in field collections, and no significant difference in burrow density or burrow size was found among oiling levels over the study period indicating recovery from any negative effects of oiling already occurred for this species. The highest density of *L. irrorata* was found at moderately oiled sites compared to both reference (without visible oiling) and heavily oiled stations. *Spartina alterniflora* density recovered within two years after the spill at the moderately oiled stations facilitating recovery of *L. irrorata* approximately one year later. *L. irrorata* average shell length and length-frequency distributions were equivalent at moderately oiled and reference stations but snails were shorter at heavily oiled stations because of a greater proportion of subadults. Shell length data from the heavily oiled sites indicate that direct mortality due to oiling or oil-induced reductions in recruitment occurred in 2010 and that recovery was starting to occur at 48 months after the spill. The extent and duration of oil in the water during the spill and the biological responses we measured indicates that *L. irrorata* and *Uca* spp. were both affected in their ability to carry out their life cycle on the marsh and/or in the water column at all stations including the reference stations for some period following the entry of oil into the region.

# Introduction

The release of a judge-ruled 3.19 million barrels of oil from the *Deepwater Horizon* (DWH) oil spill (Malakoff, 2015) exposed the nation's largest and most productive wetland-estuarine environment to an

unprecedented potential for environmental damage. Oil spills can cause widespread impacts to the structure, function, resilience, and sustainability of coastal wetlands depending upon the oil type, extent of contamination of the vegetation and marsh soils, exposure to waves and currents, time of year of the spill, and species sensitivity to oiling (Michel & Rutherford, 2014). Although the oil that made landfall in the *DWH* oil spill was relatively “weathered” and consisted of emulsions of crude oil depleted of its more volatile and toxic components, the spill resulted in the oiling of 796 km of coastal marsh shoreline, as documented by Shoreline Cleanup Assessment Technique (SCAT) teams (Michel *et al.*, 2013). Of that total, approximately 135 km were described as heavy and 165 km as moderate marsh oiling (Michel *et al.*, 2013). Approximately 95% of the total marsh oiling occurred in coastal Louisiana, and the heaviest marsh oiling was most widespread in northern Barataria Bay marshes dominated by *Spartina alterniflora* and to a lesser extent *Juncus roemerianus* (Zengel *et al.* 2014, Michel *et al.* 2013). The plant communities in northern Barataria Bay experienced strong responses to oiling, including heavy plant mortality that denuded shorelines (Lin and Mendelssohn 2012; Silliman *et al.* 2012; Zengel and Michel 2013; Zengel *et al.* 2014; Zengel *et al.*, 2015).

The marshes of the Mississippi River delta system provide a suite of environmentally and economically important services including marsh platform elevation, the prevention of soil erosion, soil nutrient cycling, food web support both grazing and detrital, and many others (McCall & Pennings, 2012; Silliman *et al.*, 2012). Most, if not all, of these ecosystem services depend upon a healthy, functioning plant-microbial-benthic system. Oil spills can exacerbate wetland loss by destabilizing these interrelated processes that control the capacity of coastal wetlands to function properly. Initial impacts of the *DWH* oil spill on coastal wetlands have been and are being reported (e.g., Whitehead, 2011, Lin and Mendelssohn, 2012, McCall & Penning, 2012, Silliman *et al.*, 2012). This study is a part of a program documenting the longer term (currently 4 years) impacts and recovery of the oiled marsh systems and their potential for sustainability.

Other members of the study team are analyzing and providing data on plant parameters, soil parameters, soil bacteria, benthic microalgae, and benthic meiofauna (Lin *et al.*, unpublished data; Fleeger *et al.*, 2015). This and other studies have shown that different marsh plant species respond differently to oiling (Alexander & Webb, 1987; Baca *et al.*, 1987; Mendelssohn *et al.*, 1990, 2012; Hoff *et al.*, 1993; Lin & Mendelssohn, 1996, 1998, 2008, 2009; Hester & Mendelssohn, 2000; Pezeshki *et al.*, 2000; Lin *et al.*, 2002; DeLaune *et al.*, 2003; Culbertson *et al.*, 2008; Michel & Rutherford, 2014), demonstrating impacts to coastal wetland vegetation, e.g. reduced plant photosynthesis, transpiration, shoot height, stem density, and above and below ground biomass. These impacts on vegetation and the marsh surface can result in cascading effects on soil bacteria and macro-, meio-, and micro-faunal communities that inhabit wetlands and depend on the plant organic matter as carbon sources or as foundation species. For example, benthic microalgae and meiofauna are, both individually and when monitored together, important indicators of ecosystem function regarding food-web support and a broad range of effects and recovery from oil spills (Fleeger *et al.*, 2015).

Our sampling focused on two conspicuous and foundational marsh macroinvertebrates, *Uca* spp., marsh fiddler crabs (Montague, 1980), and *Littoraria irrorata*, the salt marsh periwinkle (Silliman & Zieman, 2001). Fiddler crabs are one of the most thoroughly-studied shore crab in North America, with robust literature available that examines individual species population dynamics, life history and ecology (Grimes *et al.*, 1989). Fiddler crabs greatly influence the marsh through burrowing and feeding activities, in effect enhancing effects on vegetation productivity and biomass, altering sediment and nutrient characteristics, altering biogeochemical cycles, altering microbial processes by aerating the marsh sediment, increasing soil drainage, and facilitating nutrient transport. Generally, the presence of fiddler crabs has been noted as indicating greater diversity of other marsh organisms, and crab population densities can reflect the productivity of a wetland (Montague, 1980; Mouton & Felder, 1996). Fiddler

crabs have been shown to be sensitive to oil spills, making them a valuable environmental indicator species (Burger *et al.* 1991, 1992; Burger & Gochfeld, 1992).

Various fiddler crab species inhabit the northern Gulf of Mexico coast. Two species in particular are found in the Louisiana marshes, *Uca spinicarpa* and *U. longisignalis*. *Uca spinicarpa* prefers clay-dominated substrates in brackish marshes ranging from nearly fresh to hypersaline. *Uca longisignalis* is restricted to sediments of terrigenous origin ((i.e. mucky soils) and found primarily in lower salinity environments (upper estuaries). The two species can be found in close proximity but have preferred habitats based on elevation, vegetation and sediment character (Mouton & Felder, 1996; Zengel *et al.*, 2014). Although there may be more than one species of fiddler crab, we will refer to them collectively as *Uca* except when we refer to a specific species.

*L. irrorata* also is an indicator species of the health of the salt marsh habitat (Silliman & Zieman, 2001). In areas dominated by short-to-intermediate form *Spartina alterniflora*, the species has been noted to occur in densities of at least 100 individuals/m<sup>2</sup> (Silliman & Zieman 2001; Stagg & Mendelssohn 2012). *L. irrorata* is a rasping detritivore/herbivore specialist, feeding on organic matter on the marsh surface during low tide and ascending the *Spartina* stems to feed upon standing-dead *Spartina* and its associated microbial assemblages as the tide rises (Silliman & Zieman 2001). As a detritivore, *L. irrorata* influences nutrient dynamics by expediting the decomposition of *Spartina alterniflora* and serves as an important link between primary and secondary production (Stagg & Mendelssohn 2012). The presence of *S. alterniflora* has been directly linked to increased abundance, growth and survival of *L. irrorata* in marshes (Kiehn & Morris 2009; Stagg & Mendelssohn 2012).

Fiddler crabs (Krebs & Burns, 1977; Burns & Teal 1979; Burger *et al.*, 1991; Teal *et al.*, 1992; Culbertson *et al.*, 2007; Morris *et al.* 2015) and the marsh periwinkle (Hershner & Moore 1977; Hershner & Lake, 1980; Lee *et al.*, 1981; Bennett *et al.*, 1999; Garner *et al.* 2015) both can be severely impacted by oil spills, with direct effects including direct mortality, reduced population densities, and sublethal effects,

e.g. behavior alterations and reproduction effects. Pennings *et al.* (in review), using only sites without substantial oiling from this and other studies, evaluated the potential impact of the *DWH* oil spill on recruitment of *L. irrorata* in Louisiana comparing the proportion of subadults that would have recruited before the oil spill (pre-2011) to the proportion in 2011 and the proportion occurring in subsequent years to evaluate recovery. They found that the proportion of subadults in *L. irrorata* populations in Louisiana was reduced by 90% in 2011 suggesting recruitment failure in 2010. Effects of oiling can occur during the portion of these species life spent on the marsh platform or, as indicated by the Pennings *et al.* (in review) study, when they release fertilized eggs into the water and expect return of larvae to the marsh shoreline. Recovery time following oil-induced impacts for these organisms can vary substantially based on a variety of oiling and habitat conditions, from a year to several decades (Culbertson *et al.*, 2007).

In this study, we present the results of an investigation of the effects of the *DWH* oil spill on fiddler crab and marsh periwinkle at different oiling levels between 30 and 54 months after the spill. As has been indicated, Lin *et al.* (unpublished data) studied the response of the plant community including the dominant salt marsh plants *S. alterniflora* and *J. roemerianus* in the same sampling stations as used for this investigation.

## Methods

The 21 shoreline sampling stations were established in 2011 within an approximate 8 km by 5 km area in Wilkinson Bay and Bay Jimmy in northern Barataria Bay, Louisiana, USA (between coordinates N 29.44060° - 29.47459°, W 89.88492° - 89.94647°) that had been impacted to varying degrees by the *DWH* oil spill (Fig. 1). Sampling stations were randomly selected using SCAT data and our own field observations, and included seven locations each that received either no oiling (i.e., reference) (RF), moderate oiling (MD), or heavy oiling (HV). Sampling occurred at various occasions (generally bi-annually) from 30 months



(November 2012) after the spill to 54 months (October 2014) after the spill. Sampling was generally in the spring (April - June) and fall months (September – November). Total petroleum hydrocarbon and plant community data were collected at the time of sampling (Lin *et al.*, unpublished data).

## Total Petroleum Hydrocarbon (TPH) analysis

The 0–2 cm surface soils were collected from each station on each collection date, transported to the Louisiana State University laboratory on ice, extracted with dichloromethane (DCM), and analyzed gravimetrically (Lin & Mendelssohn, 2012). DCM extracts were transferred to pre-weighed dishes, where the DCM was evaporated, and the unevaporated oil remaining in the dishes was weighed to the nearest 0.0001 g. TPH concentration was calculated and expressed as  $\text{mg g}^{-1}$  dry soil. The results of the analysis for 30 to 54 months after the spill are provided in Figure 8.

## Aboveground Biomass and Stem Density

Plant aboveground biomass and stem density were taken at each station during each sampling within a haphazardly located 0.25 m<sup>2</sup> quadrat. All plants rooted within the quadrat were then clipped to the ground surface and separated into live and dead components by species. Stem density was determined by counting the number of intact living stems for each dominant plant species. All aboveground biomass was then dried to a constant mass at 60°C and weighed. Parameters reported included total live and dead above ground biomass and above ground biomass and stem density for each of the dominant species (*Spartina alterniflora* and *Juncus roemerianus*).

## Macroinvertebrate sampling

Sampling included fiddler crab (*Uca*) burrow density and size and marsh periwinkle (*Littoraria irrorata*) density and size for both components of the study. Sampling was conducted using three 0.25 m<sup>2</sup> quadrats placed at random approximately 1 m inward from the marsh edge. Data was converted to m<sup>-2</sup> basis for analysis. Each quadrat was sampled for both *L. irrorata* abundance and *Uca* burrow abundance. *L. irrorata* were found on the marsh surface or attached to the vegetation within the plot; juvenile *L. irrorata* were often found hiding in the leaf bracts of *Spartina alterniflora*. *L. irrorata* shell length and *Uca* burrow size (diameter) were measured to the nearest millimeter using calipers and a transparent ruler, respectively. Shell lengths were taken as a total height, a measurement from the base of the aperture to the top point of the shell. Once measured, *L. irrorata* were returned to the marsh within their specific sampled quadrat. As fiddler crabs are mobile and difficult to quantify within a certain area, their burrows were used as a proxy for their abundance on the marsh. We were not able to measure crab burrows at two sampling occasions (36 and 42 months after the spill) because water was present on the marsh platform, covering the marsh sediment in the sampling area. Photos were taken of the general site setting and each quadrat, and general notes were made concerning site flora and fauna composition and characteristics. *Uca* adults were identified to species level and released if found in the sampling site vicinity.

Measured *L. irrorata* were grouped into putative age classes based on their size: 0-6 mm were classified as juveniles, 7-13 mm were classified as sub-adults, and 14+ mm were classified as adults (Hamilton, 1978; Stagg & Mendelssohn, 2012). Sizes were examined as a percentage of the total abundance for each site type (RF, MD, and HV; Figs. 5-7). Size frequency distributions of *L. irrorata* are often bi- or tri-modal with the modes representing age class (Hamilton, 1978; Zengel *et al.*, 2014). The

largest snails in most populations range from 20 – 26 mm in shell length (Emerson and Jacobson, 1976; Morris, 1975), though the largest reported size is 32 mm (Kaplan, 1988). Based on growth estimates (Stiven and Hunter, 1976), juveniles would be <1 year of age, subadults would average about 1 year of age, and adults would be 2 years and older in age. These designations were used in analyses to examine life stage-specific parameters of *L. irrorata* abundance and distribution throughout the study. Evaluating size distribution by comparing size frequency histograms over time is helpful in determining how populations were impacted and tracing recovery trajectories (Zengel *et al.*, 2014; Zengel *et al.*, 2015; Pennings *et al.*, in review).

## Statistical Analysis

All statistical analyses were conducted using SAS (Statistical Analysis Systems, version 9.2, SAS Institute, Cary, NC). *L. irrorata* abundance data were tested for normality and found to be skewed [Shapiro-Wilk Test ( $\alpha = 0.05$ )]. Several transformations were tested and the natural log transformation was the best fit; therefore, all *L. irrorata* abundance data were transformed as the natural log prior to analysis. We used repeated measures, two-way mixed-model Analysis of Variance (ANOVA) with oiling level, sampling period, and their interactions to test for variation in *L. irrorata* abundance. We used Mauchly's Test of Sphericity to examine the form of the common covariance on the assumptions of the repeated measures ANOVA and found that the data do not meet sphericity assumptions ( $\text{Pr} > \text{ChiSq} = 0.0057$ ). An adjusted Tukey's test was used to determine differences between *L. irrorata* abundances and oiling level at each sampling period.

We tested the *L. irrorata* shell size data for normality and found these data to be skewed [Shapiro-Wilk Test ( $\alpha = 0.05$ )]. All attempted transformations (log, natural log, square root) were unsuccessful.

Kruskal-Wallis test was therefore used to examine shell size by oiling level and sampling period, e.g. comparing shell size for the RF sites by sampling period.

The nonparametric Kolmogorov-Smirnov two-sample test was used to compare the total abundance (total, adult, and subadult) and average shell length data between oiling levels across all sampling periods with the null hypothesis that the two oiling levels have the same distribution.

*Uca* burrow abundance data were tested for normality and found to be skewed [Shapiro-Wilk Test ( $\alpha = 0.05$ )]. Transformations and removal of the 36 and 40 month data sets which were mainly zeros (periods of high water) were unable to normalize data. Oiling levels and abundance of burrows were therefore tested using Kruskal-Wallis/Mann-Whitney (non-parametric) tests, both leaving in the 36 and 40 month datasets and removing those datasets from the analysis; however, no significant results were found.

Correlations were performed, using Kendall's Tau non-parametric rank correlations with dependence on variables, with oiling level and sampling period to 48 months after the spill because of the lack of availability of plant and TPH data at the time of the analysis; *L. irrorata* abundance; *Uca* burrow density; *S. alterniflora* stem density and biomass; *J. roemerianus* stem density and biomass; total live biomass; total dead biomass; and TPH.

Significant difference was defined as  $p \leq 0.05$ . P-values are reported to 2 decimal places and, in cases where  $p = 0.00$ , the p-value is reported as  $p < 0.01$ . All error terms are expressed in standard error (SE). Raw data can be found at the following link: [doi:10.7266/N7FF3Q9S](https://doi.org/10.7266/N7FF3Q9S).

## Results

### *Littoraria irrorata* abundance

*L. irrorata* abundance varied among oiling levels when data were combined across all sampling periods and size classes. *L. irrorata* were found to have the highest average density at MD (mean = 92.5 ind m<sup>-2</sup>, n=3,888) compared to RF (mean = 38.1 ind m<sup>-2</sup>, n =1604, Kolmogorov-Smirnov test,  $p < 0.01$ ) and HV sites (mean = 33.5 ind m<sup>-2</sup>, n = 1404, Kolmogorov-Smirnov test,  $p < 0.01$ ). RF sites were also found to be statistically different than HV sites (Kolmogorov-Smirnov test,  $p = 0.02$ ). This same trend was found within adults as both RF and MD differed from HV sites (Kolmogorov-Smirnov test,  $p = 0.02$  and  $<0.01$ , respectively, Fig. 2). No difference was found among oiling levels for subadults. Juveniles were cryptic and inconsistent in their distribution, and there was no significant difference between oiling levels and abundance of juveniles. *L. irrorata* abundance also exhibited differences between size classes at all of the oiling levels. Adults are significantly more abundant than both sub-adults (Kolmogorov-Smirnov test,  $p < 0.01$ ) and juveniles (Kolmogorov-Smirnov test,  $p < 0.01$ ), while sub-adults are significantly more abundant than juveniles (Kolmogorov-Smirnov test,  $p < 0.01$ ).

When sampling period and oiling level were analyzed together with ANOVA, oiling level had a significant effect on *L. irrorata* total abundance ( $n = 7$  and  $p < 0.01$ ), but the sampling period and interaction effect were not significant (Fig. 3). We examined within-subjects main effects tests to examine changes in snail abundance over time with the null hypothesis that mean abundance did not change over time. We rejected the null hypothesis using Wilk' test ( $Pr>F = 0.0139$ ) and concluded that abundance varied over time. We used the same method to test the interaction between time and oiling level. We again rejected the null hypothesis and concluded that change in mean abundance across time depended on oiling level ( $Pr>F = 0.0037$ ). Significant differences in *L. irrorata* total density were noted between oiling levels at different sampling periods. Significant differences existed between both MD (Tukey's test,  $p = 0.02$ ) and RF (Tukey's test,  $p < 0.01$ ) and HV sites at 36 months after the spill; between the MD and RF sites (Tukey's test,  $p = 0.05$ ) at 40 months after the spill; and MD and RF (Tukey's test,  $p = 0.05$ ) and HV sites at 36 months after the spill.

249

## 250 ***Littoraria irrorata* shell size**

251 *L. irrorata* body size also varied among oiling levels when data were combined across all sampling  
252 periods. *L. irrorata* average shell length was significantly less at HV sites (mean = 15.44 mm) when  
253 compared to both MD sites (mean = 17.37, Kolmogorov-Smirnov test,  $p < 0.01$ ; Fig. 4) and RF sites (mean  
254 = 16.62 mm, Kolmogorov-Smirnov test,  $p < 0.01$ ). RF and MD sites showed no statistical difference in shell  
255 length (Kolmogorov-Smirnov test,  $p = 0.43$ ).

256 *L. irrorata* shell length was also used to examine size-frequency distributions at all oiling levels. RF  
257 and MD sites exhibited similar proportions of juveniles, subadults, and adults, while the HV sites contained  
258 a lower percentage of adults, in particular, large adults. This low proportion of adults at the HV sites likely  
259 corresponded to the direct mortality of all size categories of *L. irrorata* at the HV sites due to the oiling in  
260 2010. As indicated in methods, adults are two years or older; therefore, even at 30 months, smaller adults  
261 would just be entering the populations and replacing these losses at the HV sites.

262 The null hypothesis of no difference in shell length across sampling periods was rejected (Kruskal-  
263 Wallis test,  $p < 0.01$ ) indicating that there was a difference in shell size among oiling levels (figure 8).  
264 Periwinkles in all oiling categories averaged less than 15 mm in shell size 30 months after the spill. The  
265 HV sites continued to average less than 15 mm in shell size up to 48 months after the spill; whereas, the  
266 RF and MD sites achieve an average shell size greater than 17 mm at 48 months after the spill.

267

## 268 ***Uca* burrow density and size**

269 All of the fiddler crabs collected during all sampling periods were identified as *Uca longisignalis*.  
270 No significant difference was found between oiling level and *Uca* burrow density. Because no significant  
271 difference was found with density, burrow size was not tested.

## Correlations

With all oiling levels and all stations, *L. irrorata* abundance was positively correlated with most of the plant parameters including *S. alterniflora* above ground biomass ( $p = 0.05$ ); *J. roemerianus* above ground biomass ( $p < 0.01$ ) and stem density ( $p < 0.01$ ); and total live ( $p < 0.01$ ) and dead above ground biomass ( $p = 0.06$ ). *Uca* burrow density was negatively correlated with *S. alterniflora* stem density ( $p = 0.03$ ).

For the RF sites, the only significant correlation occurred at 30 months after the spill between *L. irrorata* abundance and *Uca* burrow density ( $p = 0.04$ ). The MD sites showed a significant positive correlation between *L. irrorata* abundance and *J. roemerianus* above ground biomass ( $p = 0.02$ ) and stem density ( $p = 0.03$ ) at 40 months after the spill; a significant negative correlation between *L. irrorata* and *Uca* burrow density ( $p = 0.01$ ) 42 months after the spill; and a significant negative correlation with *S. alterniflora* stem density ( $p = 0.03$ ) and positive correlation with *J. roemerianus* stem density ( $p = 0.02$ ) at 48 months after the spill. The HV sites showed no significant correlations.

TPH did not correlate with either *L. irrorata* or *Uca* abundance; however, there was a positive significant correlation between TPH and *S. alterniflora* stem density ( $p = 0.04$ , all data;  $p < 0.01$ , HV sites). There were significant negative correlations between *J. roemerianus* above ground biomass ( $p < 0.01$ ) and stem density ( $p < 0.01$ ) and total live and dead biomass (both,  $p < 0.01$ ).

## Discussion

### *Littoraria irrorata*

As stated in the Results, *L. irrorata* were found to have the highest average density at MD compared to RF and HV sites. RF sites were also found to be statistically different than HV sites. This same

trend was found within adults as both RF and MD differed from HV sites. The reduced abundance at HV compared to MD was due to the oiling, either by direct mortality or effects on recruitment by the loss of vegetation at the HV sites. Many studies (Kiehn & Morris, 2009; Silliman & Zieman 2001; Silliman & Bertness 2002; Silliman & Newell, 2003; Silliman *et al.*, 2005. Kiehn & Morris (2009) have noted that marsh periwinkle density is positively correlated with *S. alterniflora* stem density. Stagg & Mendelssohn (2012), in a study of marshes restored using sediment from dredging operations, found that *L. irrorata* growth, survival, and productivity were positively correlated to increasing *S. alterniflora* canopy cover. High levels of *J. roemerianus*, however, may be detrimental to *L. irrorata* density. Alber *et al.* (2008) noted that densities of *L. irrorata* are generally low on *J. roemerianus* in their analysis of the possible reasons of saltmarsh dieback. There may be a limit to the amount of *J. roemerianus* (biomass or stem density) within the marsh that is beneficial compared to the amount that suppresses the *L. irrorata* population density. Above ground biomass and stem density of *J. roemerianus* was equal to or greater at times than *S. alterniflora* density at the RF stations (Lin *et al.*, unpublished data), potentially limiting the abundance of *L. irrorata*. *J. roemerianus* has been shown to be more sensitive to oiling than *S. alterniflora* (Lin *et al.*, unpublished data, Lin & Mendelssohn, 2012), and its density and biomass was expectedly suppressed at the moderately and HV stations (Lin *et al.*, unpublished data).

Location may have also contributed to lower *L. irrorata* density at the RF sites. All of the RF sites are located northwest of the moderately and HV stations in a cove that is connected to larger marsh islands. This location could have a larger predator population than the smaller marsh islands where the other stations are located. The location is also further north into the estuary and may be influenced by freshwater, potentially altering the recruitment of *L. irrorata* and/or the population dynamics of *J. roemerianus* and *S. alterniflora* (Pennings *et al.*, 2005). Alternatively, abundances in the MD sites may have been higher than the RF sites because of an elevated stem density of *S. alterniflora* (Fleeger *et al.*,



2015). Increased stem density at the MD sites may have been a response by *S. alterniflora* to reduced competition from *J. roemerianus* or a compensatory response associated with oiling.

Although the density of *L. irrorata* was low at RF sites, length-frequency data are similar to those found in other gulf coast areas not impacted by oiling (Pennings *et al.*, in review), suggesting that these data adequately represent a true reference population for comparison with oiled sites. Average *L. irrorata* shell lengths (Fig. 4) of *L. irrorata* population at the MD and RF stations were equivalent indicating that recovery at MD sites was achieved within the time period of this study. As an example of recovery of lightly oiled areas, McCall and Pennings (2012) estimated *L. irrorata* density, not shell length frequency, at oiled and control sites in Louisiana and Mississippi four and 16 months (August 2010 and 2011) after the DWH spill. The sampling sites were located 1-2 m behind the HV zone in areas dominated by *Spartina alterniflora*, and often partially coated with oil, with a very light (sheen) on the soil surface. McCall and Pennings (2012) reported no difference in the population density at 4 or 16 months after the spill.

A similar comparison of the average shell length (Fig. 4) between the RF and MD stations and the HV stations indicates that body size was much lower at the HV stations and that they did not fully recover within the time period of the study. Shell length also indicates that all of the oiling levels may have initially been affected by oiling within Barataria Bay because the average shell length was initially smaller than that which was eventually achieved at the RF sites (17.71 mm). These data indicate that recovery may have been occurring at the RF and MD sites approximately 36 months after the spill and, at the HV sites, 48 months after the spill. Other studies have investigated more heavily-oiled sites associated with the DWH spill using population density. Silliman *et al.* (2012) sampled *L. irrorata* density at three reference and three impacted sites, approximately 3 m from the shoreline, in Barataria Bay approximately 6 months (October 2012) after the DWH spill. Their group found no live individuals at impacted stations and approximately 50 m<sup>-2</sup> at the reference stations. Note that their reference stations had relatively low

densities (slightly greater than the density at our RF stations) and less than the average density of our MD stations.

In a study of shoreline treatment options on a marsh island in northern Barataria Bay, approximately 1 km south of our study sites, Zengel *et al.* (2015) monitored *L. irrorata* population densities and shell size frequencies in relation to shoreline treatment options (manual, mechanical, and no treatment; see Zengel and Michel (2013) 17 and 29 months (September 2011 and 2012) after the DWH spill. This study provides data at reference and heavily-oiled stations between the time period of the Silliman *et al.* (2012) and our study. They found minimal recovery of *L. irrorata* at treatment sites. An interesting result was that the reference station had less than 50 m<sup>-2</sup> *L. irrorata* in 2011 (17 months after the spill) and greater than 150 m<sup>-2</sup> in 2012 (29 months after the spill). Zengel *et al.* (2015) mentions two potential reasons for the differences in densities between the sampling periods; one, under-sampling the juvenile *L. irrorata* in the first sampling period due to not searching leaf bracts for juveniles and, two, the possible effect of widespread oiling in the water surface within Barataria Bay during the summer of 2010 and the effect that may have had on the larvae of *L. irrorata*. We find the juvenile *L. irrorata* very patchy in their distribution on the marsh platform because of the juveniles' preference, described in Zengel *et al.* (2015) to aggregate and hide in areas on *S. alterniflora*, such as within leaf bracts. We also find the juveniles sporadic between years potentially because of their broadcast life cycle and we do mention below that both *L. irrorata* and *Uca* were affected in their ability to carry out their broadcast life cycle in 2010 throughout northern Barataria Bay because of oil on the water. This caused a potential recovery effect at all of our stations including the RF stations.

The correlation of *L. irrorata* abundance with vegetation parameters is not unexpected. Kiehn & Morris (2009) and Stagg & Mendelssohn (2012) found similar positive correlations between *S. alterniflora* health (stem density, canopy cover) and *L. irrorata* health (density, growth, survival, productivity).

Additionally, Fleeger et al. (2015) found that the infauna at our study sites followed the recovery of *S. alterniflora*. Of particular interest is the correlation of *L. irrorata* with *J. roemerianus* parameters (above ground biomass, stem density, and total live and dead above ground biomass). This may indicate the importance of *J. roemerianus* to the recovery of *L. irrorata* within the marsh. Hughes (2012) describes the potential importance of *J. roemerianus* as alternative refuge for *L. irrorata*; as described above, however, Alber et al. (2008) note that the densities of *L. irrorata* on *J. roemerianus* is generally low. We, therefore, may be underestimating the importance of *J. roemerianus* within a diverse marsh system. The correlation of *L. irrorata* and *J. roemerianus* at 40 months (above ground biomass and stem density) and 48 months (stem density) at the MD stations, however, may indicate the coincidental recovery of the two species with the reduction of oil at those stations (Figure 9).

## *Uca*

This study started 30 months after the *DWH* oil spill, and our data clearly show that that any effect of the spill on fiddler crab, *Uca*, density passed prior to the beginning of this effort. We found that average density of crab burrows within the area of our sampling (approximately 1 m from the shoreline edge) was low, between 5 and 10 burrows m<sup>-2</sup>. This is not surprising, as Mouton & Felder (1996) noted the variation in burrow density of *U. longisignalis* along a 15-m transect from the edge into the marsh. The burrow density was lowest near the marsh edge and highest in the middle and upper reaches of the transect with no marked change in elevation noted beyond 3 m from the water's edge. Several studies have found similar results. Silliman et al. (2012) sampled *Uca* sp. crab burrow density at the three reference and three impacted sites described above and found no difference between the density of crab burrows (approximately 10 m<sup>-2</sup>) at the two site types. McCall & Pennings (2012) sampled *Uca* burrows at the sampling sites described above. Because they were sampling further back in the marsh, McCall &

Pennings (2012) reported greater numbers of burrows (approximately 20 m<sup>-2</sup>, see Mouton & Felder 1996 above) and found a significantly lower density of burrows at the oiled sites 4 months after the spill and no difference at 16 months after the spill.

Zengel *et al.* (2015) monitored *Uca* burrow densities in relation to shoreline treatment options after the DWH spill (see above). *Uca* burrow densities in heavily-oiled and treated plots were similar to reference at both 17 and 29 months; however, the heavily-oiled sites that were not treated were significantly different at 17 months, but similar at 29 months. Zengel *et al.* (2014) revisited the treatment sites at 41 months after the spill (September 2013) and found significant differences between the oiled control sites (no treatment) relative to the references sites. Maximum average burrow density was approximately 10-15 burrows m<sup>-2</sup> similar to densities in our study. We also identified only *Uca longisignalis* within our study sites. The study at the treatment sites (Zengel *et al.*, 2014; Zengel *et al.*, 2015) identified *U. spinicarpa* occurring at the heavily-oiled plots where there was reduced vegetation coupled with areas of surface oil residue overlaid with thin algal mats and clay-like sediments.

Morris *et al.* (2015) conducted studies on adult male and female *Uca longisignalis* placing the individuals in Total Polycyclic Aromatic Hydrocarbon (TPAH)<sub>50</sub> concentrations from 0.07 (reference sediment) to 26 mg/kg in the upper 2 cm of sediment. During the 10 day exposure period several females became gravid. The gravid females were removed after 10 days and placed in clean water until the embryos hatched. The zoea were collected, held in clean water, and exposed to varying levels of ambient sunlight. They found substantial toxicity (calculated LC<sub>20</sub> value of 0.62 mg/kg TPAH<sub>50</sub> in the upper 2 cm of sediment) to *Uca* zoea at relatively low concentrations of oil in the sediments. Fig. 9 provides the total petroleum hydrocarbon concentrations from 30 to 54 months after the spill at the 21 stations from this study. It is difficult to compare the two hydrocarbon parameters and much of the aromatic portions of the oil have weathered reducing the toxicity; however, the levels were found to be significant higher at the HV sites compared to the RF and MD sites (Lin *et al.* unpublished data).

The negative correlations between *Uca* sp. burrow density and *S. alterniflora* stem density (all oiling levels /all stations) and *L. irrorata* (MD stations 42 months after the spill) may be indicative of the conflict between the burrowing and feeding activity of the fiddler crab in and on the marsh and the density of *S. alterniflora*. Mouton & Felder (1996) indicate that *U. longisignalis* burrow density increased landward of the 4-m into the marsh where the vegetation became more open and scattered. They report the maximum density of burrows near the 8-m to 12-m marks into the marsh suggesting a potential relationship between tidal flooding and the deposition of wrack contributing to food resources.

The extent of shoreline oiling within northern Barataria Bay (Zengel & Michel 2013) and the extent and duration of oil in and on the water indicated in the Environmental Response Management Application (ERMA) (<http://gomex.erma.noaa.gov/erma>) would indicate that the populations of *L. irrorata* and *Uca* were affected in their ability to carry out their life cycle on the marsh (Morris *et al.* 2015) and in the water column at all stations including the reference stations for some period following the entry of oil into the region (Fodrie *et al.*, 2014; Pennings *et al.*, unpublished data). This study found that the dominant fiddler crab within these marshes, *U. longisignalis*, recovered at all oil levels prior to sampling at 30 months after the DWH spill. There are indications that *L. irrorata* started recovery at the MD stations at 40 months after the spill and population density is beginning to demonstrate seasonal cycling. Recovery, however, has not occurred at the HV stations.

## References

- Alber, M., E.M. Swenson, S.C. Adamowicz, I.A. Mendelssohn. 2008. Salt marsh dieback: an overview of recent events in the US. *Estuarine, Coastal and Shelf Science*. 80: 1-11.  
doi:10.1016/j.ecss.2008.08.009.
- Alexander, S.K., J. W. Webb. 1987. Relationship of *Spartina alterniflora* growth to sediment oil content following an oil spill. *Proceeding of the 1987 Oil Spill Conference*, pp. 445-449.
- Baca, B.J., T.E. Lankford and E.R. Gundlach. 1987. Recovery of Brittany coastal marshes in the eight years following the Amoco Cadiz incident. *Proceedings of the International Oil Spill Conference 1987*: 459-464.
- Bennett, A., T.S. Bianchi, J.C. Means, K.R. Carman. 1999. The effects of polycyclic aromatic hydrocarbon contamination and grazing on the abundance and composition of microphytobenthos in salt marsh sediments (Pass Fourchon, Louisiana) I: A microcosm experiment. *Journal of Experimental Marine Biology and Ecology* 242, 1-20.
- Burger, J., J. Brzorad, M. Gochfeld. 1991. Immediate effects of an oil spill on behavior of fiddler crabs (*Uca pugnax*). *Arch. Environ. Contam. Toxicol.* 20, 404-409.
- Burger, J., J. Brzorad, M. Gochfeld. 1992. Effects of an oil spill on emergence and mortality in fiddler crabs *Uca pugnax*. *Environ. Monit, Assess.* 22(2): 107-115.
- Burger, J., and M. Gochfeld. 1992. Effects of washing fiddler crabs (*Uca pugnax*) following and oil spill. *Environmental Pollution* 77(1) 15-22.
- Burns, K.A., J.M. Teal. 1979. The West Falmouth Oil Spill: hydrocarbons in the salt marsh ecosystem. *Estuarine and Coastal Marine Science* 8,349-360.
- Culbertson, J.B., I. Valiela, E.E. Peacock, C.M. Reddy, A. Carter. 2007. Long-term biological effects of petroleum residues on fiddler crabs in salt marshes. *Marine Pollution Bulletin* 54, 955–962.

- 452 Culbertson, J.B., I. Valiela, M. Pickart, E.E. Peacock, C.M. Reddy. 2008. Long-term consequences of  
453 residual petroleum on salt marsh grass. *Journal of App. Ecology* 45, 1284-1292 (2008).
- 454 DeLaune, R.D., S. R. Pezeshki, A. Jugsujinda, C. W. Lindau. 2003. Sensitivity of US Gulf of Mexico coastal  
455 marsh vegetation to crude oil: Comparison of greenhouse and field responses. *Aquatic Ecology*  
456 37, 351-360 (2003).
- 457 Emerson, W.K., M.K. Jacobson. 1976. The American Museum of Natural History Guide to Shells. Alfred A.  
458 Knopf, New York. 482 p.
- 459 Fleeger, J.W., K.R. Carman, M.R. Riggio, I.A. Mendelssohn, Q. Lin, A. Hou, D.R. Deis, S. Zengel. 2015.  
460 Recovery of saltmarsh benthic microalgae and meiofauna from the *Deepwater Horizon* oil spill  
461 linked to recovery of *Spartina alterniflora*. *Marine Ecology Progress Series*, Vol. 536: 39–54.
- 462 Grimes, B.H., M.T. Huish, J.H. Kerby, D. Moran. 1989. Species Profiles: Life Histories and Environmental  
463 Requirements of Coastal Fishes and Invertebrates (Mid-Atlantic). Atlantic Marsh Fiddler.  
464 Biological Rept. North Carolina Cooperative Fishery Research Unit Raleigh. 26p.
- 465 Hershner, C. and K. Moore. 1977. Effects of the Chesapeake Bay oil spill on salt marshes of the lower  
466 bay, In: *Proceedings of the International Oil Spill Conference*, American Petroleum Institute,  
467 Washington, D.C., 529-533.
- 468 Hamilton, P.V. 1978. Intertidal distribution and long-term movements of *Littoraria irrorata* (Mollusca:  
469 Gastropoda). *Marine Biology* 46:49–58.
- 470 Hershner, C. and J. Lake. 1980. Effects of chronic oil pollution on a salt marsh grass community. *Marine*  
471 *Biology* 56, 163-173.
- 472 Hester, M.W., I.A. Mendelssohn. 2000. Long-term recovery of a Louisiana brackish marsh plant  
473 community from oil-spill impact: vegetation response and mitigating effects of marsh surface  
474 elevation. *Marine Environmental Research* 49, 233-254.

- 475 Hoff, R.Z., G. Shigenaka, C.B. Henry 1993. Salt marsh recovery from a crude oil spill: vegetation, oil  
476 weathering, and response. *Proceeding of the 1993 Oil Spill Conference*, American Petroleum  
477 Institute, Washington, D.C., pp. 307-311.
- 478 Hughes, A.R. 2012. A neighboring plant species creates associational refuge for consumer and host.  
479 *Ecology*, 93(6), 2012, pp. 1411–1420
- 480 Kaplan, E. H. 1988. A Field Guide to Southeastern and Caribbean Seashores: Cape Hatteras to the Gulf  
481 Coast, Florida, and the Caribbean. Houghton Mifflin, Boston, MA.
- 482 Kiehn, W.M. and J.T. Morris. 2009. Relationships between *Spartina alterniflora* and *Littoraria irrorata* in a  
483 South Carolina salt marsh. *Wetlands* 29(3): 818-825.
- 484 Krebs, C.T., K.A. Burns. 1977. Long-Term Effects of an oil spill on populations of the salt marsh crab *Uca*  
485 *pugnax*. *Science* 197, 484-487.
- 486 Lee, R.F. B. Dornseif, F. Gonsoulin, K. Tenore, and R. Hanson. 1981. Fate and effects of a heavy fuel oil  
487 spill on a Georgia salt marsh. *Marine Environmental Research* 5,125-143.
- 488 Lin, Q. and I. A. Mendelssohn. 1996. A comparative investigation of the effects of Louisiana crude oil on  
489 the vegetation of fresh, brackish, and salt marsh. *Marine Pollution Bulletin* 32, 202-209 (1996).
- 490 Lin,Q. and I. A. Mendelssohn. 1998. The Combined effects of phytoremediation and biostimulation in  
491 enhancing habitat restoration and oil degradation of petroleum contaminated wetlands.  
492 *Ecological Engineering* 10, 263-274.
- 493 Lin, Q. and I. A. Mendelssohn. 2008. Determining tolerance limits for restoration and phytoremediation  
494 with *Spartina patens* in crude oil-contaminated coastal wetlands in greenhouse. *Archives of*  
495 *Agronomy and Soil Science* 54, 681-690.
- 496 Lin, Q. and I. A. Mendelssohn. 2009. Potential of restoration and phytoremediation with *Juncus*  
497 *roemerianus* for diesel-contaminated coastal wetlands. *Ecological Engineering* 35, 85-91.



- 498 Lin, Q. and I. A. Mendelssohn. 2012. Impacts and Recovery of the Deepwater Horizon Oil Spill on  
499 Vegetative Structure and Function of Coastal Salt Marsh in the Northern Gulf of Mexico.  
500 *Environmental Sciences and Technology*,46, 3737-3743.
- 501 Lin, Q., I. A. Mendelssohn, M.T. Suidan, K.Lee, A.D. Venosa. 2002. The dose-response relationship  
502 between No. 2 fuel oil and the growth of the salt marsh grass, *Spartina alterniflora*. *Marine*  
503 *Pollution Bulletin*, 44, 897-902.
- 504 Lin, Q., I.A. Mendelssohn, S. Graham, A. Hou, J.W. Fleeger, D.R. Deis. Unpublished data. Long-Term  
505 Recovery of Structure, Function, and Sustainability of Louisiana Coastal Salt Marshes Impacted  
506 by the *Deepwater Horizon* Oil Spill. *Environmental Science & Technology*.
- 507 Malakoff, D. 2015. After geoscientists joust, judge rules BP Gulf spill totaled 3.19 million barrels of oil.  
508 *Science*, doi: 10.1126/science.aaa6358. <http://news.sciencemag.org/environment/2015/01>.
- 509 McCall, B.D. and S.C. Pennings. 2012. Disturbance and recovery of salt marsh arthropod communities  
510 following BP Deepwater Horizon oil spill. *PLOS One* 7(3): e32735.
- 511 Mendelssohn, I.A., M. W. Hester, C. Sasser, M. Fischel. 1990. The effect of Louisiana crude oil discharge  
512 from a pipeline break on the vegetation of a southeast Louisiana brackish marsh. *Oil Chem.*  
513 *Pollut.*, 7, 1-15 (1990).
- 514 Mendelssohn, I. A., G. L. Andersen, D. Baltz, R. Caffey, K. R. Carman, et al. 2012. Oil impacts to coastal  
515 wetlands: Implications for the Mississippi river delta ecosystem after the Deepwater Horizon oil  
516 spill. *Bioscience* 62, 562-574.
- 517 Michel, J., E.H. Owens, S. Zengel, A. Graham, Z. Nixon, T. Allard, W. Holton, P.D. Reimer, A. Lamarche, M.  
518 White, N. Rutherford, C. Childs, G. Mauseth, G. Challenger and E. Taylor. 2013. Extent and degree  
519 of shoreline oiling: Deepwater Horizon oil spill, Gulf of Mexico, USA. *PLOS One* 8(6): e65087.
- 520 Michel, J., N. Rutherford 2014. Impacts, recovery rates, and treatment options for spilled oil in marshes.  
521 *Marine Pollution Bulletin* 82:19-25Q.

- 522 Montague, C.L. 1980. A natural history of temperate western Atlantic fiddler crabs (genus *Uca*) with  
523 reference to their impact on the salt marsh. *Contributions in Marine Science* 23, 25-55 (1980).
- 524 Morris, J.M., M.O. Krasnec, M.W. Carney, H.P. Forth, C.R. Lay, I. Lipton, A.K. McFadden, R. Takeshita, D.  
525 Cacela, J.V. Holmes, J. Lipton. 2015. *Deepwater Horizon* Oil Spill Natural Resource Damage  
526 Assessment Comprehensive Toxicity Testing Program: Overview, Methods, and Results.  
527 *Prepared for:* National Oceanic and Atmospheric Administration, Assessment and Restoration  
528 Division. 805 p.
- 529 Morris, P.A. 1975. A Field Guide to Shells of the Atlantic and Gulf Coasts and the West Indies. Houghton  
530 Mifflin Company, Boston. 330 p.
- 531 Mouton, E.C. and D.L. Felder. 1996. Burrow distributions and population estimates for the fiddler  
532 crabs *Uca spinicarpa* and *Uca longisignalis* in Gulf of Mexico salt marsh. *Estuaries* 19(1):  
533 51-61.
- 534 Pennings, S.C., S. Zengel, J. Oehrig, M. Alber, T.D. Bishop, D.R. Deis, D. Devlin, A.R. Hughes, J.J.  
535 Hutchens, Jr., W.M. Kiehn, C.R. McFarlin, C.L. Montague, S. Powers, C.E. Proffitt, N.  
536 Rutherford, C.L. Stagg, K. Walters. Unpublished data. Marine ecoregion and the  
537 *Deepwater Horizon* oil spill affect recruitment and population structure of a salt marsh  
538 snail.
- 539 Pennings, S.C., M. Grant, M.D. Bertness. 2005. Plant zonation in low-latitude salt marshes: disentangling  
540 the roles of flooding, salinity and competition. *Journal of Ecology*, Vol. 93 (1), pages 159–167.
- 541 Pezeshki, S. R., M. W. Hester, Q. Lin, J. A. Nyman. 2000. The effects of oil spill and clean-up on dominant  
542 US Gulf coast marsh macrophytes: a review. *Environmental Pollution* 108, 129-139.
- 543 Silliman, B. R., and S. Y. Newell. 2003. Fungal farming in a snail. *Proceedings of the National Academy of*  
544 *Sciences, USA*. 100:15643–15648.

- 545 Silliman, B. R., M. D. Bertness. 2002. A trophic cascade regulates salt marsh primary production.  
546 *Proceedings of the National Academy of Sciences*. 99:10500–10505
- 547 Silliman, B. R., J. van de Koppel, M. D. Bertness, L. E. Stanton, and I. A. Mendelssohn. 2005. Drought,  
548 snails, and large-scale die-off of Southern U.S. salt marshes. *Science* 310:1803–1806.
- 549 Silliman, B.R., J. van de Koppel, M.W. McCoy, J. Diller, G.N. Kasozi, K. Earl, P.N. Adams , A.R.  
550 Zimmerman. 2012. Degradation and resilience in Louisiana salt marshes after the BP-Deepwater  
551 Horizon oil spill. *Proceedings of the National Academy of Sciences of the United States of*  
552 *America* 109:11234-11239.
- 553 Silliman, B.R., Zieman 2001. Top-down control of *Spartina alterniflora* production by periwinkle grazing  
554 in a Virginia salt marsh. *Ecology* 82:2830–2845.
- 555 Stagg, C.L. and I.A. Mendelssohn. 2012. *Littoraria irrorata* growth and survival in a sediment- restored  
556 salt marsh. *Wetlands* 32: 643-652.
- 557 Stiven, A. E. and J. T. Hunter. 1976. Growth and mortality of *Littorina irrorata* Say in three North Carolina  
558 marshes. *Chesapeake Science* **17**:168-176.
- 559 Teal, J.M., J.W. Farrington, K.A. Burns, J.J. Stegeman, B.W. Tripp. 1992. The West Falmouth oil spill after  
560 20 years: fate of fuel oil compounds and effects on animals. *Marine Pollution Bulletin* 24, 607-  
561 614.
- 562 Whitehead, A., B. Dubansky, C. Bodinier, T.I. Garcia, S. Miles, et al., Genomic and physiological footprint  
563 of the *Deepwater Horizon* oil spill on resident marsh fishes. *PNRS*, doi:  
564 [10.1073/pnas.1109545108](https://doi.org/10.1073/pnas.1109545108). (2011).
- 565 Zengel, S. and J. Michel. 2013. Deepwater Horizon oil spill: salt marsh oiling conditions, treatment  
566 testing, and treatment history in northern Barataria Bay, Louisiana. (Interim Report October  
567 2011). *U.S. Dept. of Commerce, NOAA Technical Memorandum NOS OR&R 42*. Seattle, WA:

568 Emergency Response Division, NOAA. 74 pp.

569 [http://response.restoration.noaa.gov/deepwater\\_horizon](http://response.restoration.noaa.gov/deepwater_horizon)

570 Zengel, S., N. Rutherford, B. Bernik, Z. Nixon, J. Michel. 2014. Salt marsh remediation and the

571 *Deepwater Horizon* oil spill, the role of planting in vegetation and macroinvertebrate recovery.

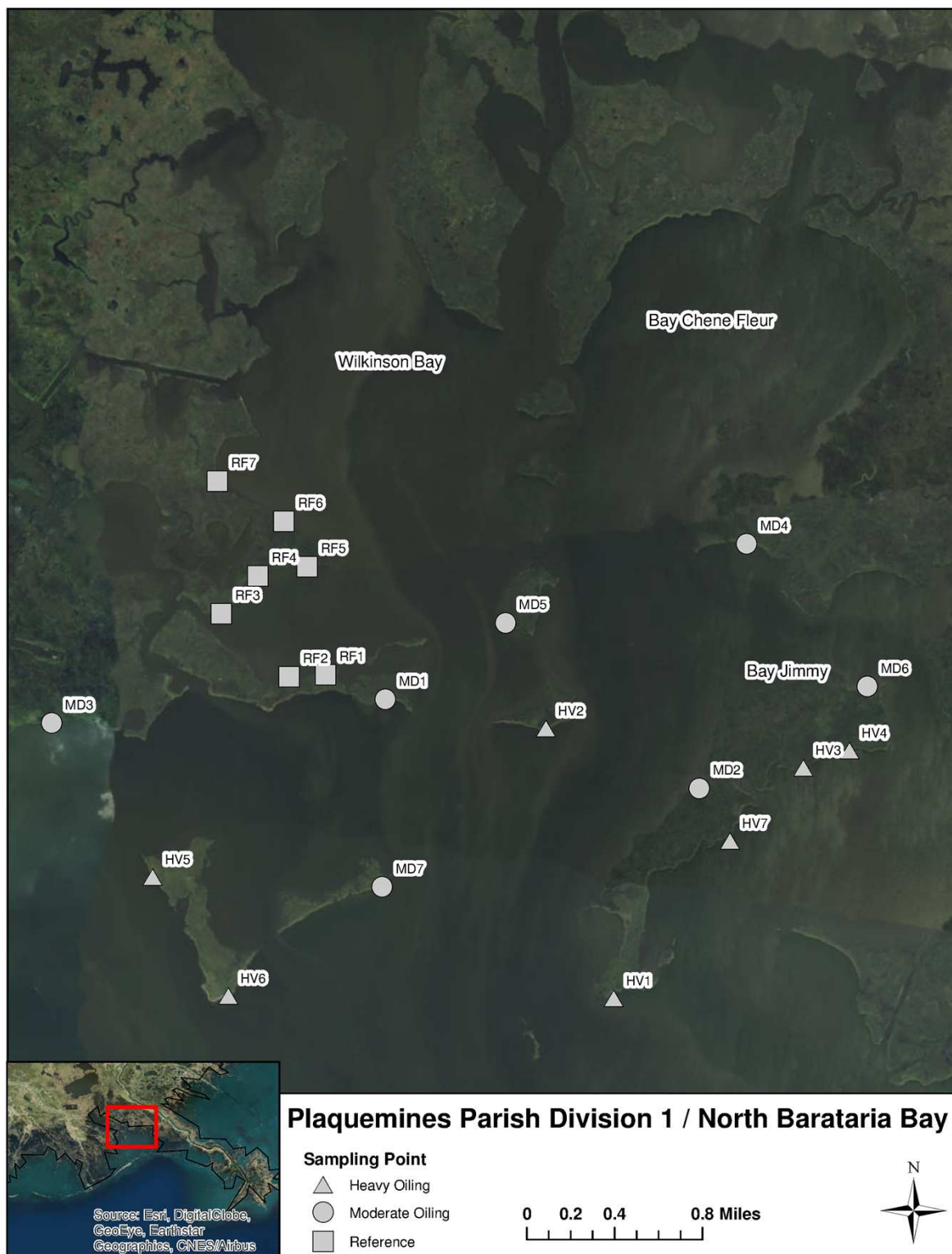
572 *Proceedings of the 2014 International Oil Spill Conference*, pp 1985-1999.

573 Zengel, S., B.M. Bernik, N. Rutherford and Z. Nixon, J. Michel. 2015. Heavily Oiled Salt Marsh following

574 the Deepwater Horizon Oil Spill, Ecological Comparisons of Shoreline Cleanup Treatments and

575 Recovery. PLoS ONE 10(7): e0132324. doi:10.1371/journal.pone. 66p.

576



577

578 Fig. 1. Stations locations within Barataria Bay, LA.



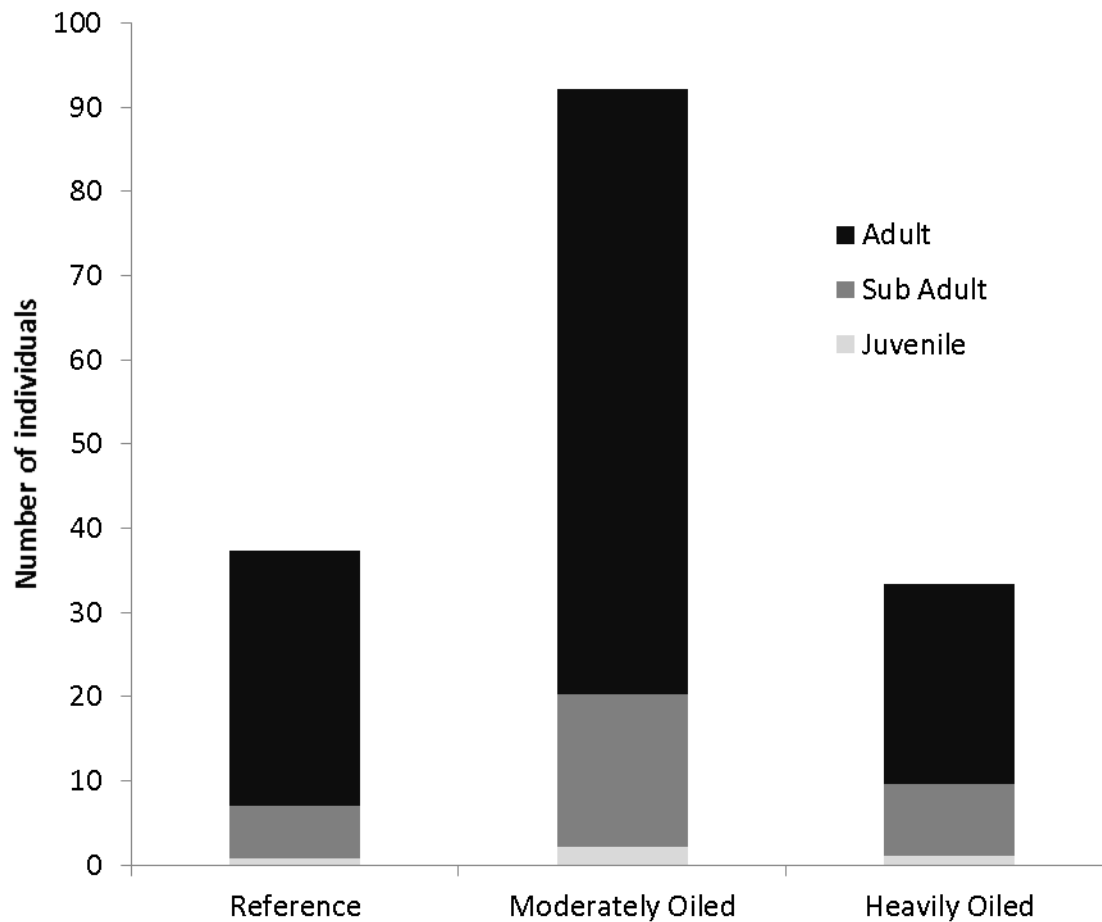
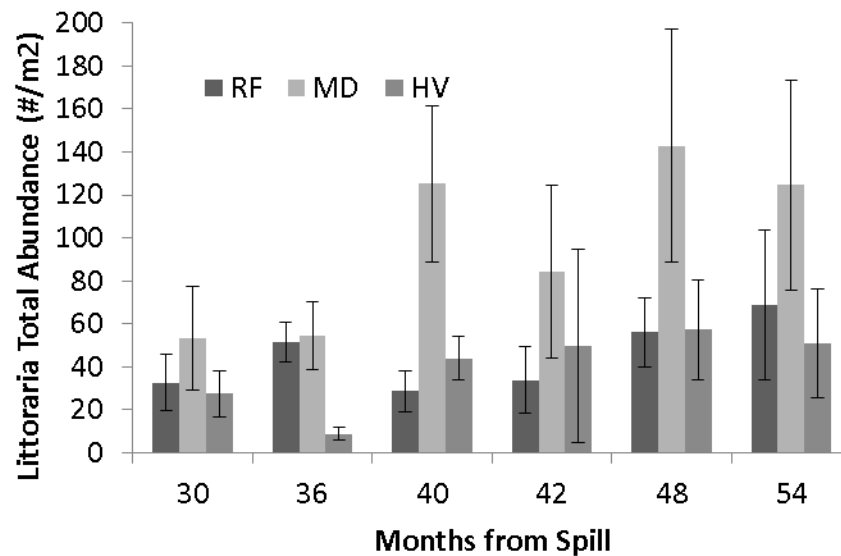


Fig. 2. Mean *Littoraria* abundance between oiling levels across all sampling times and within various size classes. X-axis = site type, Y-axis = average density/m<sup>2</sup>. Black = adults, grey = sub-adults, and light grey = juveniles.



586

587 Fig. 3. *Littoraria irrorata* total density  $m^{-2}$  30 to 54 months after the *DWH* oil spill. Values are means ( $\pm$ SE;  
 588  $n=7$ ) for reference (RF), moderately-oiled (MD), and heavily-oiled (HV) shoreline marshes of northern  
 589 Barataria Bay, LA, USA.

590



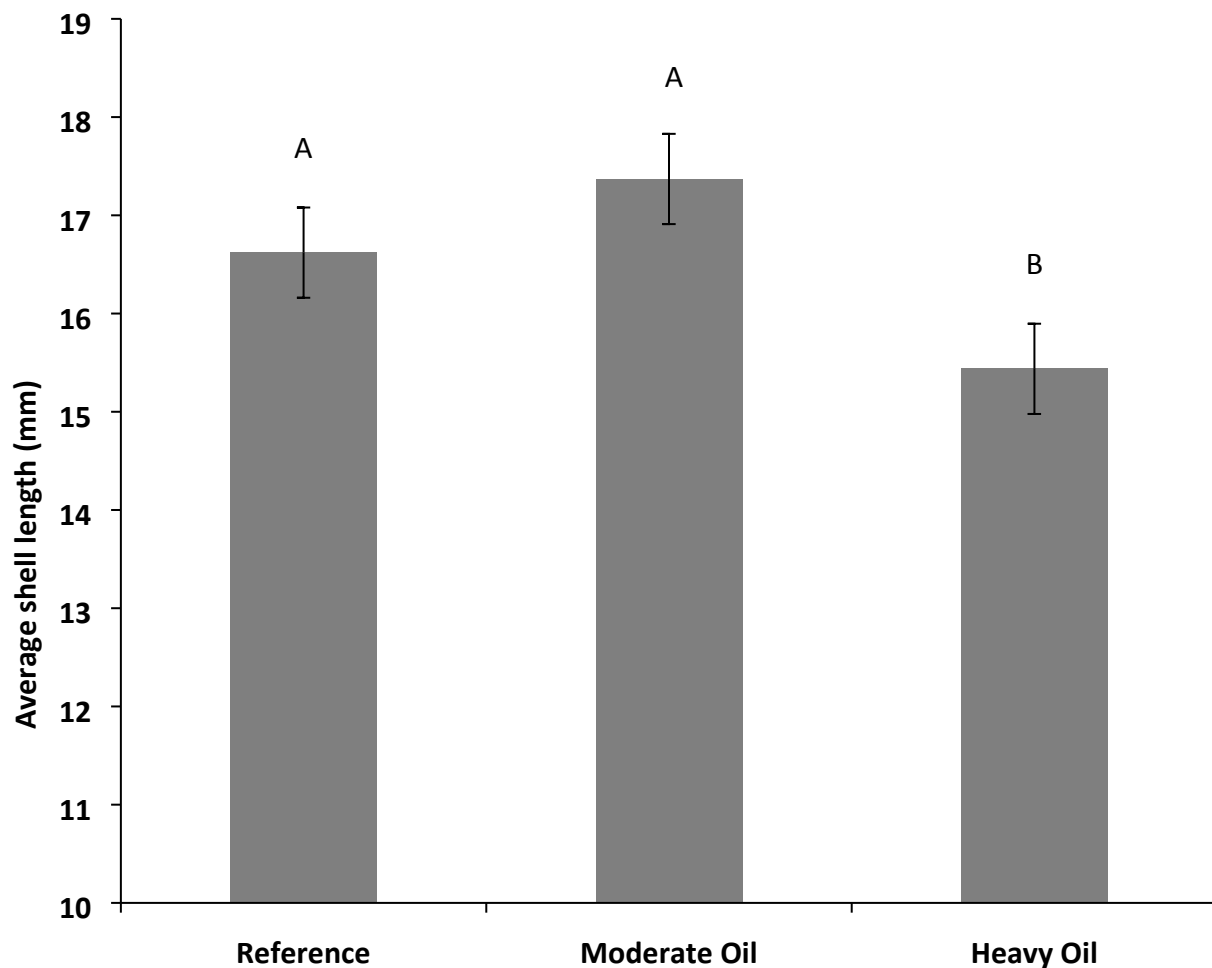
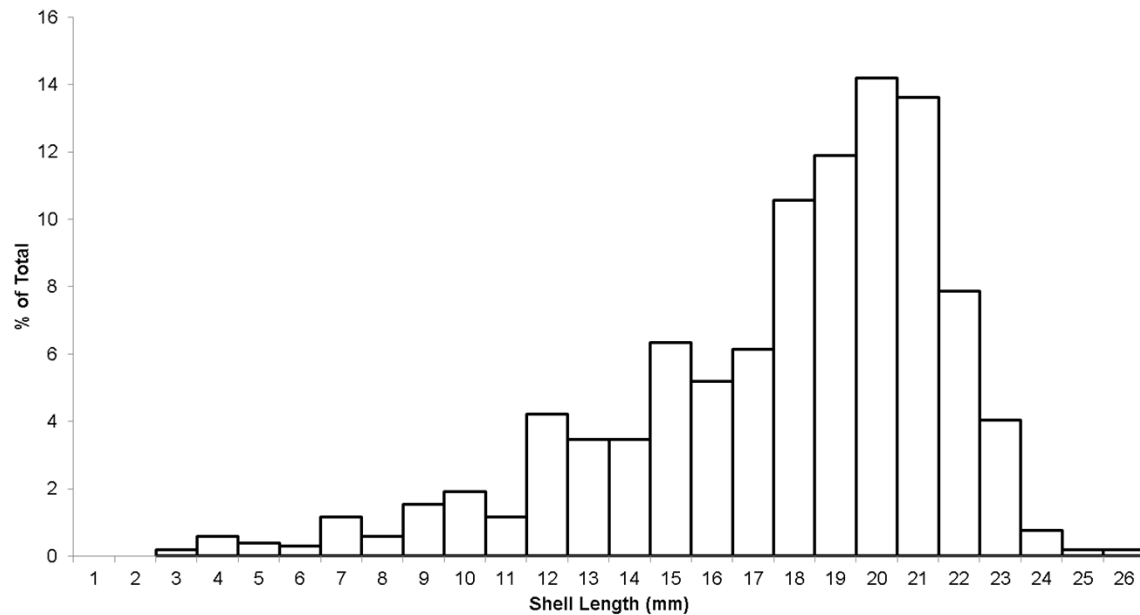


Fig. 4. Average *Littoraria* shell length by sampling site type across all sampling times. X-axis = site type, Y-axis = average shell length (mm). Heavily-oiled (HV) sites had significantly smaller *Littoraria* than reference (RF) or moderately-oiled (MD) sites.



595

596

Fig. 5. Reference (RF) sites Littoraria size distribution by percentage of total abundance. X-axis = shell length in mm, Y-axis = percentage of total abundance represented by each specific shell length across all sampling times.

598

599

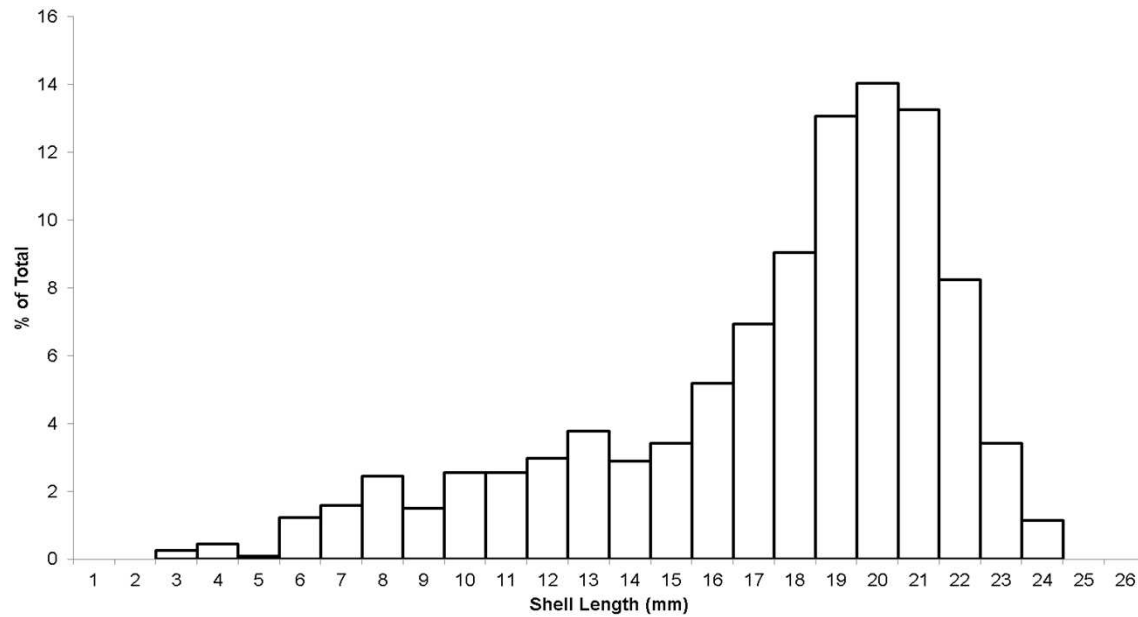
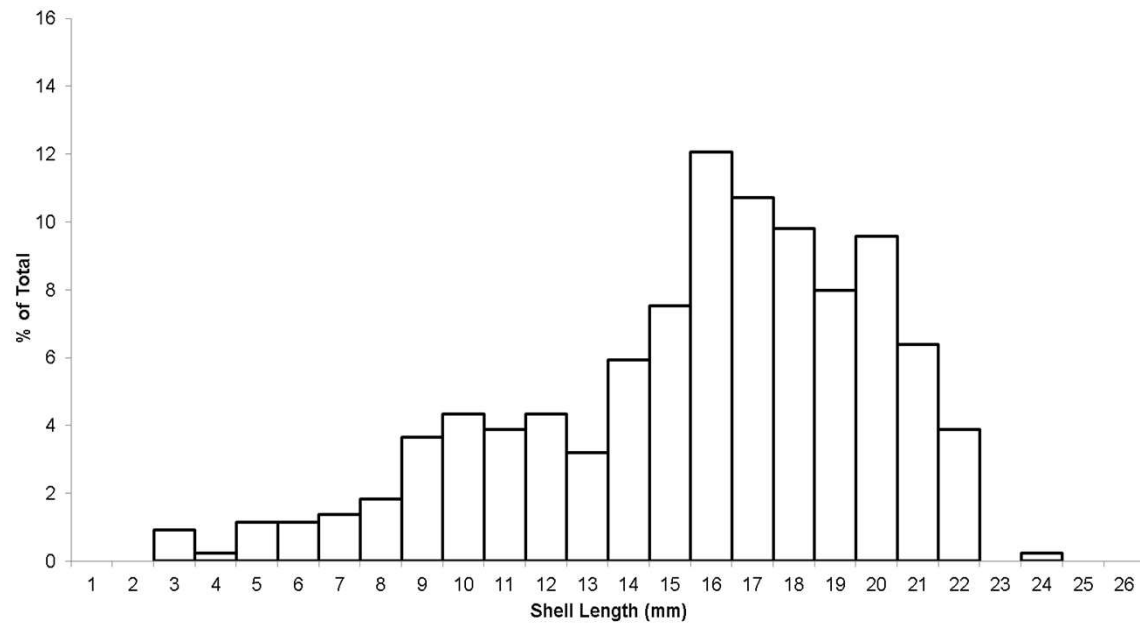


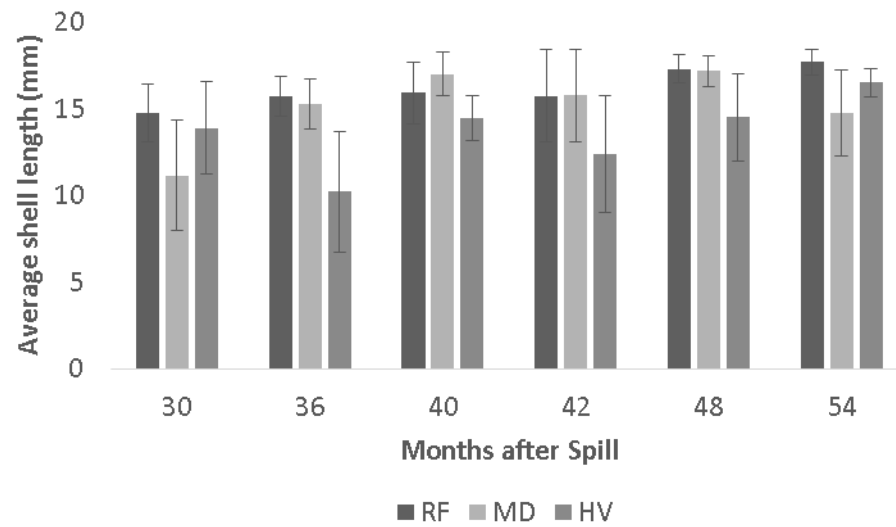
Fig. 6. Moderately-oiled (MD) sites *Littoraria* size distribution by percentage of total abundance. X-axis = shell length in mm, Y-axis = percentage of total abundance represented by each specific shell length across all sampling times.



605

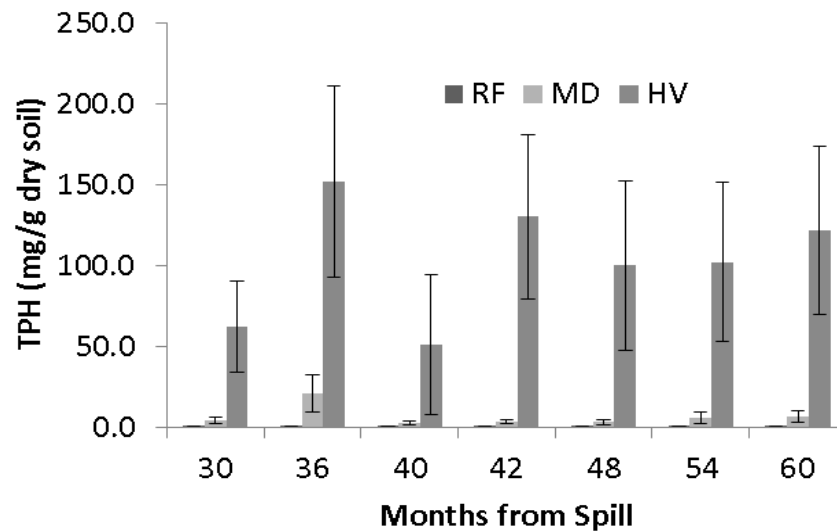
606 Fig. 7. Heavily-oiled (HV) sites *Littoraria* size distribution as percentage of total abundance. X-axis = shell  
 607 length in mm, Y-axis = percentage of total abundance represented by each specific shell length across all  
 608 sampling times.

609



610

611 Figure 8. Average shell length (mm) for each oiling level within each sampling period after the spill.



612

613 Fig. 9. Surface soil (0-2 cm) total petroleum hydrocarbon concentrations 30 to 54 months after the *DWH*  
 614 oil spill. Values are means ( $\pm$ SE; n=7) for reference (RF), moderately-oiled (MD), and heavily-oiled (HV)  
 615 shoreline marshes of northern Barataria Bay, LA, USA (from Lin *et al.* unpublished data).

616

617