Marine Pollution Bulletin 110 (2016) 316–323

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Islands in the oil: Quantifying salt marsh shoreline erosion after the Deepwater Horizon oiling

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A R T I C L E I N F O

Article history:
Received 26 June 2015
Received in revised form 21 May 2016
Accepted 12 June 2016
Available online 24 June 2016

Keywords:
Wetland
Erosion
Macondo oil spill
Louisiana
Rookery habitat

A B S T R A C T

Qualitative inferences and sparse bay-wide measurements suggest that shoreline erosion increased after the 2010 BP Deepwater Horizon (DWH) disaster, but quantifying the impacts has been elusive at the landscape scale. We quantified the shoreline erosion of 46 islands for before and after the DWH oil spill to determine how much shoreline was lost, if the losses were temporary, and if recovery/restoration occurred. The erosion rates at the oiled islands increased to 275% in the first six months after the oiling, were 200% of that of the unoiled islands for the first 2.5 years after the oiling, and twelve times the average land loss in the deltaic plain of 0.4% y^{-1} from 1988 to 2011. These results support the hypothesis that oiling compromised the belowground biomass of the emergent vegetation. The islands are, in effect, sentinels of marsh stability already in decline before the oil spill.

1. Introduction

The 20 April, 2010 Deepwater Horizon (DWH) oil spill 66 km from the Louisiana coastline was momentous in terms of its prolonged release, the wetland area covered with oil, multiple oilings (Boufadel et al., 2014), and the anticipated ecological effects. This accident released approximately 4.9 million barrels of oil and gas hydrate, of which 0.8 million barrels were collected before the remaining 4.1 million barrels dispersed into the Gulf of Mexico over 87 days (McNutt et al., 2012). It was seven times larger than the 1989 Exxon Valdez oil spill in Alaska, and the largest in US history. The oil released reached about 1055 km of Louisiana's shoreline (348 km of ≥heavier oiling), equaling about 65% of the total oiled shoreline in the Gulf of Mexico and 95% of the oil wetland area (Nixon et 2016). Oil arrived at the beach mid-May (Boufadel et al., 2014), inside the estuaries in significant quantities in early June 2010 (NOAA 2013), and observations from two different shoreline erosion studies started months afterwards. McClenachan et al.'s (2013) measurements at 30 shoreline locations within a few km of each other began in November 2010, and Silliman et al.'s (2012) shoreline erosion measurements for three oiled stations and three control sites started in October 2010. Both studies, therefore, had no baseline measurements of shoreline changes before the spill or in the first few months afterwards when oil toxicity might be the highest. Quantifying how the salt marsh system immediately responded to the oiling over a broader geographical area and at many more sites may inform our understanding of the influence of multiple stressors on these systems occurring before sea level rise begins its anticipated strong acceleration (Strauss, 2013).

There are hundreds of islands with salt marsh vegetation in Louisiana estuaries that could be used for this purpose (Fig. 1). Some islands were oiled in 2010 and some were not, thus offering a potential natural laboratory to test for differences in shoreline erosion rates, and to indirectly test for evidence of recovery from shoreline erosion. We measured the changes in island widths and lengths from 1989 to 2012 for a suite of islands that were, or were not, oiled by the 2010 DWH oil spill. We used these data to address three questions: 1) how much faster is the shoreline retreat when these marshes are oiled? 2) how long do their effects last?, and, 3) is there recovery?

2. Materials and methods

2.1. Landscape background

About 23% Louisiana's coastal wetlands converted to open water from the 1930s to 2014, with a peak loss rate of 0.86% y^{-1} (12,700 ha y^{-1}) from 1958 to 1975, and then a decline to a few km^2 y^{-1} and even recent stasis in some areas (Couvillion et al., 2011). These land-to-water conversions are tightly coupled in time and space with, and many conclude are driven by, the amount of dredging done to create channels through the marsh (Turner, 2011). The plants survive within a tidal amplitude of 10 to 20 cm (McKee and Patrick, 1988), and require favorable soil oxidation gradients (Mendelssohn et al., 1981). The predominately organic surface soils originating from these plants overlie layers of sand, silts and clays far below the root zone, and deposited hundreds, if not thousands of years ago. The roots and rhizomes

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http://dx.doi.org/10.1016/j.marpolbul.2016.06.046

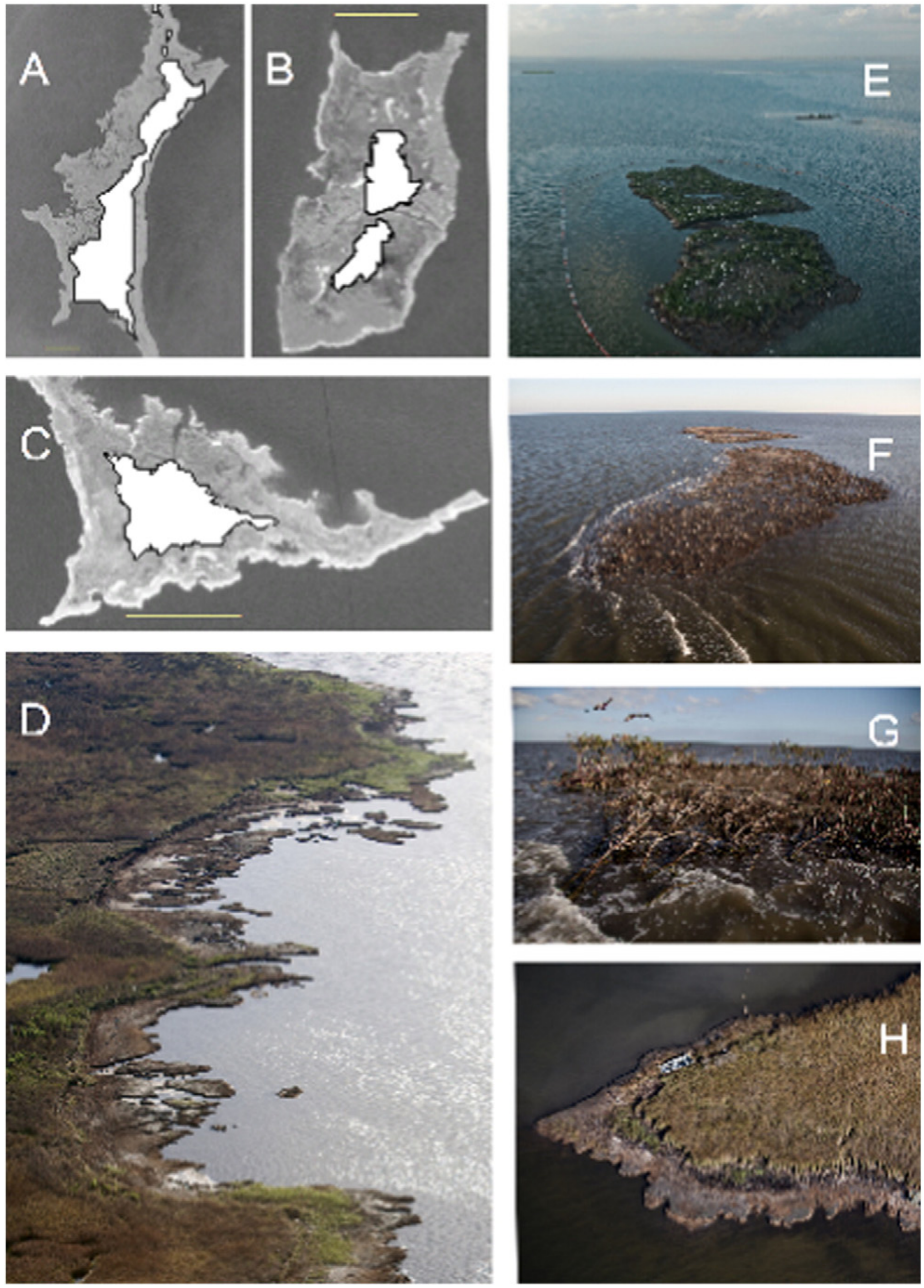

Fig. 1. Photo mosaic of oiled islands. (A–C). Examples of changes in the size and shape of three islands from January 1998 to May 2010, before the oiling of the marshes in south Louisiana. The yellow bar is 100 m long. (A) site 3.16; (B) Cat Island, location of a Pelican rookery before 2010 (located north of Grand Terre, LA, USA); (C) site 1.6; (D) Bay Jimmy on 14 December 2010. The brown fringes of the land-water interface are oiled marshes; (E) Cat Island on 12 June 2010. An oil boom surrounds the island; (F and G) Cat Island on 7 May 2012. The pelicans are the white specks in F, and seen flying in G; (H) Bay Jimmy on 14 December 2010. The brown edge is the oiled marsh platform that has lost vegetation and subsided about 10 cm. Photo credits: photos D, G, H by Tyrone Turner/www.tyronefoto.com; photo E by Joel Sartore/www.joelsartore.com.

within the modern organic layer helps to hold the overlying marsh soil together.

The vegetation on all islands examined is primarily salt marsh dominated by *Spartina alterniflora*, with some patches of *Spartina patens* and *Schoenoplectus americanus*, *Phragmites australis* and the occasional black mangrove, *Avicennia germinans*. A few islands are important bird rookeries for several species including the brown pelican (*Pelecanus occidentalis* Linnaeus, 1766), the Louisiana State Bird that was reintroduced to Louisiana after near-extinction from pesticide poisoning, and taken off the threatened and endangered species list in 2009.

2.2. Aerial imagery

We used readily-accessible and credentialed aerial imagery to measure the width (east to west) and length (north to south) of 46 islands (SI Fig. 1) starting in 1989 and 1990 to the end of 2012. The islands were chosen before being designated as oiled or not as a result of oil from the April 2010 DWH disaster. The oil reached the mainland for the first time in late June 2010. This was an intentional order of analysis meant to minimize subjectivity. Islands intermingled among a substantial marsh or with dredged channels were excluded, as were island images containing a seam between two photos.

The photographs are in the publicly-accessible software GoogleEarth (vers. 7.1.1.1888; accessed from August to October 2013 and June 2014). The photos come from various US agencies that routinely process satellite and aerial imagery that is publicly available (e.g. US Geological Survey, US National Aeronautics and Space Administration, and the US Department of Agriculture). The quality of the aerial photos is sufficient to determine the presence of oil booms, pelicans on roosts, and recreational fishing boats. The software facilitates easy movements from one photo to another while maintaining a vertical position. We used the linear measurement tool to measure the east-to-west and north-to-south dimension of a geomorphic feature of each island appearing in all photos. Not all photos are precisely and accurately registered with each other, and so a measurement was positioned in reference to an anchor point (the geomorphic feature) that remains in each photo. It was important to find the same recognizable feature from the oldest to the latest photographs, and to use only those islands that remained intact as one island for the entire measurement interval. The recognizable features used might be an indentation or promontory whose basic shape is maintained through the entire measurement interval. The marsh-water edge was easily identified because these marshes have emergent vegetation up to the shoreline that is at least 50 cm high at all times. The marsh is at the upper-level of a 20 to 30 cm tidal range. We only excluded measurements if there was cloud interference (poor visibility).

We evaluated the precision of these images by making measurements of the distance between fixed points on structures for six different locations within the Barataria and Terrebonne Bay estuaries. Seven to twelve different photographs were used for each site, with each representing a different photograph date. The standard error of the mean (SEM) for the widths of the fixed structures ranged from 0.09 to 0.33 m, with a coefficient of variation of 0.22 to 0.89% (Table S1).

Some aerial images were of insufficient quality or had an incorrect date. The earliest photo for the 46 islands was from 1989, and the latest from November 2012 (the mean (μ) time interval 15.98 ± 0.56 y; $\mu \pm 1$ SEM). There were a minimum of five and a maximum of nine aerial images taken before the marsh was oiled that we used to construct an erosion rate ending in May 2010 for each island.

A simple linear regression of the relationship of island width (m) and time (days) was made for each island before May 2010, and used only if the coefficient of determination (R^2) was >0.90. This pre-oiling erosion rate is the 'baseline erosion rate'. The first interval after the DWH disaster ($\mu = 2010.21 \pm 0.08$ yr; $n = 46$) used photographs from April or May 2010, and ended in December 2010 ($\mu = 2010.99 \pm 0.0001$; $n = 45$). The second interval continued until at least April, and sometimes to August 2011 ($\mu = 2011.42 \pm 0.03$ y; $n = 32$). The third interval ended in November or December 2012 ($\mu = 2012.83 \pm 0.001$; $n = 46$). The data are in Tables S1 and S2.

The erosion rates were then calculated for five intervals: (1) the 2010 initial po*st*-oiling interval up to December 2010, (2) the early 2011 interval from December 2010 to spring/summer 2011, (3) from the beginning of the first interval to the end of the second (spring/summer 2011), (4) from spring/summer 2011 to the end of 2012, and (5) from May 2010 to the end of 2012. The po*st*-oiling erosion rates were normalized to the pre-spill erosion rate by dividing by the baseline erosion rate to obtain relative rates for each site for all intervals. Examples of the changes from circa 1990 to 2012 are in Figs. 1A, B and C.

We recognize that we are measuring island length and width (erosion), but not island area. Using these photographic sources to measure island dimensions may be novel, and it does have the advantages of being easily accessed and inexpensive, and the accurate results are replicable. The large number of different islands ($n = 46$) spread over 60 km distance, and the multiple photographic dates used, made this an analysis that was more spatially and temporally expansive than any previous analysis of the impact of marsh oiling that we are aware of. It is a landscape-scale approach with quantitative results.

2.3. Categories of oiling

We used two categories of exposure to the 2010 BP Macondo oil: oiled and unoiled. The definition was based on the multi-agency damage assessment operations called SCAT (Shoreline Cleanup Assessment Technique, Michel et al., 2013). The SCAT assessment has six color-coded categories to indicate the degree of oiling of the shoreline: red, orange, yellow, green, light green, and blue that are equivalent to heavy, moderate, light, very light, trace and no oil, respectively (NOAA, 2013). Sites were considered 'oiled' for this analysis if the shoreline included red, orange or yellow assessments. Sites coded with only green or light green were not included in the analysis. Shorelines with portions coded as green or light green were not used if they were mixed with shorelines with blue assessments. Island shorelines coded with *only* blue were defined as 'unoiled'. Island shorelines without a color classification were not used.

2.4. Sampling design and analysis

We compared the erosion of oiled and unoiled islands before and after the 2010 DWH oil spill. One of the reasons for this sampling design is that we were concerned that the final distribution of oil might be directly related to variance in the shoreline erosion rates occurring before the spill. This might be because the movement of Macondo oil from 100 km offshore and into these estuarine systems was strongly influenced by currents, as is the erosion of estuarine islands. But the erosion of the islands occurs over years and in all months, whereas the oil transport from this particular spill was largely episodic (there was re-oiling, however) with a peak in August 2013 (Michel et al., 2013; Zengel et al., 2015). Erosion is a continuing process, therefore, whereas the oiling was not. The oiling was also patchy among islands – there were heavily oiled islands that were sometimes within a km of an island that was not categorized as oiled (see, for example, the SCAT maps discussed by Michel et al., 2013). For example, we previously studied some heavily-oiled shorelines in Bay Batiste that had no oil 20 m further down the shoreline (McClenachan et al., 2013). We therefore chose a sampling design using a large number of sampling sites because of the patchy distribution of the oil in space and the multiple stressors possibly affecting shoreline erosion, some of which might be co-related. We were able to test for differences in erosion rates of oiled and unoiled islands before the DWH oil spill, and we used these results to test for the effect of oiling on shoreline erosion over time.

We investigated if the 22 oiled and 24 unoiled islands were of similar width and length, if they eroded at the same rates before spring 2010, and if they had the same anticipated longevity. We normalized the erosion rate of each island for each interval after the oil spill by comparing the erosion rate for the interval (m y^{-1}) to the baseline value (m y^{-1}) from before the oil spill and used a log transformation to test differences. We used an unpaired Student's *t*-test to test for absolute differences between the two data sets within the time period. The statistical measure of variance is the $\mu \pm 1$ SEM, unless otherwise indicated.

We used the average increase in shoreline erosion from oiling (described above) and the SCAT surveys of oiled shoreline length in Louisiana (Nixon et al., 2016) to estimate a cumulative marsh-to-water conversion for the 2.5 years after May 2010.

In brief, then, we used verified aerial imagery to study island erosion in four directions for before and after an apparent stressor. The data were divided into the islands that were oiled or not using oiling categories defined by others, so there was an independent and binary category definition established. We are report herein on the differences found in erosion between oiled and unoiled islands.

3. Results

3.1. Erosion rates

The width, length and erosion rate of the 22 oiled and 24 unoiled islands located in three estuaries were similar at the end of the baseline measurement period that ended before oiling began (Fig. 2). The width of the oiled and unoiled islands was 274.3 ± 57.6 and 238.5 ± 49.7 m, respectively, and the average island length (north to south axis) of the same oiled and unoiled islands was 377.6 ± 60.3 and 513.6 ± 71.2 m, respectively (Figs. 2B and E). There was no statistically-significant difference in the erosion rate of oiled and unoiled islands before the 2010 oil spill (the erosion rate along the east-to-west axis for the oiled and unoiled sites was 5.2 ± 0.68 and 5.2 ± 0.55 m y^{-1}, respectively, and 8.1 ± 1.4 and 5.2 ± 0.85 m y^{-1}, respectively, along the north-to-south axis; Figs. 2A and D). If the islands eroded at these rates in the future, then the average longevity of the oiled and unoiled islands would be 61.1 ± 10.5 and 54.9 ± 10.8 years, respectively (Figs. 2C and F), equivalent to an average linear contraction rate of 1.8 and 2.2% y^{-1}, respectively. There was no difference in the erosion rates (width or length) for islands in the three estuaries (Fig. S2). These results form the basis for evaluating changes in island morphology after the oil from the Macondo oil spill entered estuarine waters.

The relative erosion rate of island width and length during the first 8 to 12 months after spring 2010 was significantly higher for the oiled islands compared to the unoiled islands (Figs. 3B and C; intervals a, b, and c). The relative erosion rate for oiled islands from spring 2010 (before the spill) to the end of 2010 (after the spill) was 250% and 319% of the 16 year long baseline value for the width and length, respectively. The average erosion rate of island width at oiled sites declined to 248% of the unoiled sites within the first 5 months of 2011, and was 130% from spring 2011 to the end of 2012 (Fig. 3B). The enhanced erosion rate for oiled island width was 170% that of the unoiled sites over the 2.5 years after the oil spill. The erosion of the island length over the 2.5 years after the oil spill was 247% of that on the unoiled islands (Fig. 3C, interval d), but lower ($p > 0.05$) for the interval 'b' after oiling.

The general pattern is, therefore, that island erosion rose to 275% of the pre-oiling rate during the first 8 months after the DWH oil spill, was 200+% higher for the first 12 months, and slowed to the point where there was no detectable difference in the erosion rates for the 1.5 years after the spill. The average erosion rate for the oiled marshes was around 200% of that for the unoiled islands over the entire 2.5 year record. The average enhanced erosion in island width and length was 3.07 m y^{-1} from May 2010 to December 2012, equaling an average shoreline erosion rate that was 1.54 m y^{-1} higher at the oiled islands compared to of the unoiled islands.

4. Discussion and conclusion

4.1. Cumulative shoreline erosion

The summary observation is that the average oiled and unoiled island width, length and erosion rate before the oil spill were similar, and that the result of the oil spill was about 275% of the erosion rate of the oiled islands in the first 8 to 12 months after the oil spill, compared to the unoiled islands. The average enhanced erosion rate for the oiled islands declined thereafter, and was about twice the average erosion rate of the unoiled sites over 2.5 years. There was no evidence of reversal in the aggregate, although temporary re-vegetation may have taken place between the photographed intervals at individual islands.

The results of our analysis of shoreline erosion for 46 islands are similar to the analysis of erosion rates from a study by Silliman et al. (2012) who started with 3 oiled and 3 unoiled marshes in eastern Barataria Bay. They measured a doubling in the shoreline erosion rate in months 8 through 15 after the oil spill (equivalent to 1.5 m y^{-1}), and no difference between control and oiled sites after 15 months. The choice of sampling sites was plagued by the possible consequences of oiling on all but one of their six sites after the site classification was made and the study began (NOAA, 2013; Khanna et al., 2013), and that two of the three sites defined as 'oiled sites' overlap the area of an intensive field experiment about the efficacy of post-spill mitigation approaches (Zengel and

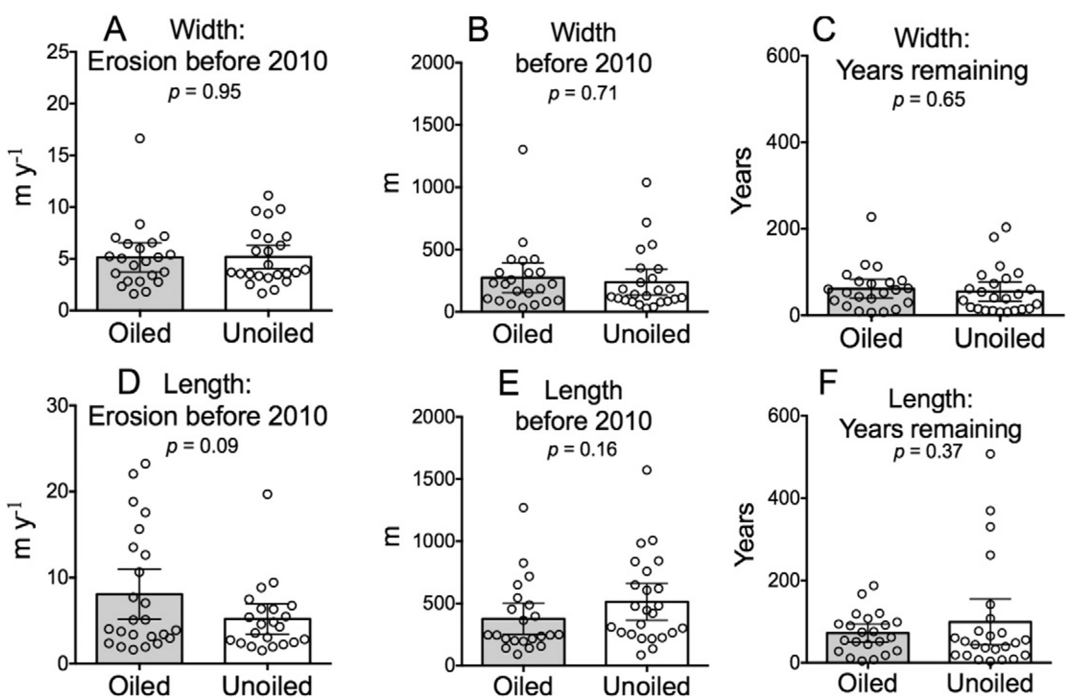

Fig. 2. Comparisons of the average erosion rates (m yr^{-1}; A and D), island transect width and length (m; B and E, respectively), and implied lifetime remaining (years; C and F), for 22 oiled and 24 unoiled islands. The data are the mean, 95% confidence interval, and the individual data (open circles). The p values are for a Student's t-test for differences between the two groups, where a p value ≤ 0.05 is significant.

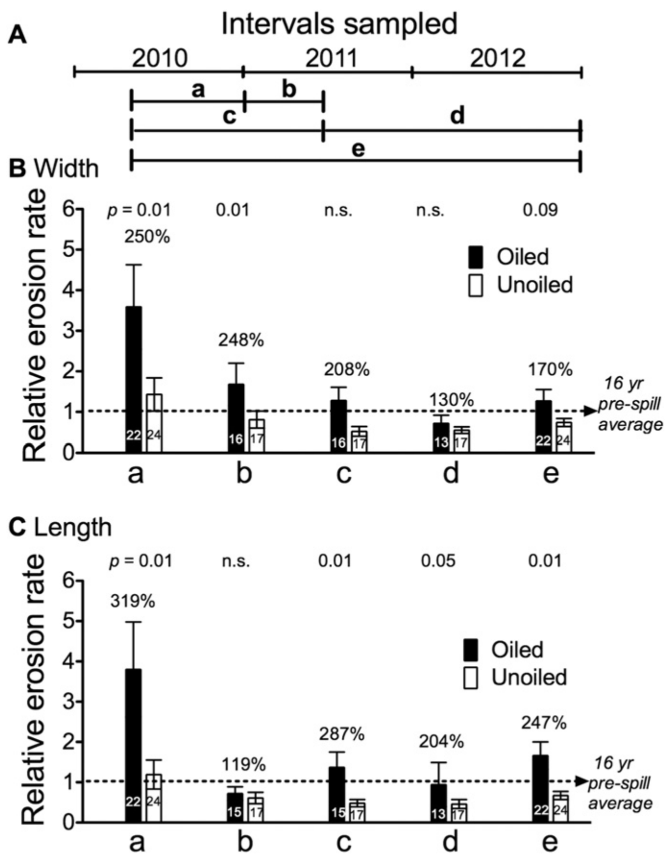

Fig. 3. Comparisons of the relative erosion rates of island width and length in five different time periods for oiled and unoiled islands. A. The intervals sampled (a, b, c, d and e). B. The data for the five sampling intervals for island width. C. The data for the five sampling intervals for island length. The data are the range and standard error of the mean for the island transect width and length (m) for values normalized relative to 16 years before the DWH disaster where 1 = the baseline rate. The *p* value is for a Student's *t*-test for differences between the two groups in each interval sampled. The percent of the erosion rate at oiled and unoiled sites for each interval is also given. The sample number (n) is in each bar. The overall average increase of the erosion width and length over 2.75 years is 224% in the oiled sites compared to the unoiled sites.

Michel, 2013). In spite of these potential problems, we can confirm the broad outlines of their results with this larger sample size.

Recently reported measurements by Zengel et al. (2015) focused in a portion of our study area. They reported similar erosion rates for heavily-oiled wetlands as we did compared to reference wetlands. The shoreline retreat rates in the reference plots were very similar to previously published rates of 0.8–1.3 m yr^{-1} for this portion of Barataria Bay, but was 2–3 times greater in the heavily oiled plots compared to the reference plots in 2011 and 2012.

The resistance to erosion offered by plants at seashore, riverbank, and hillside is presumed to be related to the root-enhanced structural strengthening of soil (Van Eerdt, 1985; Michel and Kirchner, 2002; Feagin et al., 2009; Francalanci et al., 2013; Wu, 2013). An increasing incidence of cantilevered edge profiles after exposure to a strong stressor might suggest that the soil beneath the main rooted layer is what is compromised, not the plant. Wetlands oiled after the DWH oil spill disaster had reduced soil strength at 50 to 100 cm depth, but not above where the majority of large roots are (McClenachan et al., 2013). Further, the overhang width in highly oiled wetlands was greater than in lightly oiled sites (McClenachan et al., 2013). The emergent vegetation at other wetlands exposed to the DWH oil (Fig. 1), however, was killed and there was no overhang – the wetland soil rotted in place, subsided, or sloughed off as a green toupeé (Figs. 1D and H).

4.2. Comparisons with other oiled wetlands

Details about the cause-and-effect relationships between plant physiology and oiling as it affects wetland stability may be sparse, but there is empirical evidence demonstrating consequences that are may be common among these wetlands and others that suggest continued erosion is not reversible without management interventions. For example, the disintegration of root mass after oiling is evident from the decreased trafficability of the wetland (Alexander and Webb, 1985), subsidence or surface erosion, and a deeper cantilevered shoreline profile (McClenachan et al., 2013). Hampson and Moul (1978) reported an erosion rate over three years of 24 times that in the control wetland that they attributed to disintegration of the root system after oiling. The compromises to belowground reserves leads to decreased plant production aboveground and less re-growth for longer than a year (Ferrell et al., 1984). Intentionally dosing a Chesapeake Bay salt marsh with fuel oil from November 1973 to August 1974 caused the loss of *S. alterniflora* vegetation at the shoreline, and an eight to ten cm vertical drop of the remnants (Hershner and Lake (1980). Greenhouse studies creating a dose-response relationship between fuel oil and the growth of *S. alterniflora* demonstrated a stronger effect on the belowground biomass (reduced by 50%) than the aboveground biomass (Lin et al., 2002). Further, the natural recovery from the effects of heavy oiling is not common. A 26-year post-mortem of a fuel oil spill in a Winsor Cove (MA) salt marsh, for example, found that there was still a complete loss of peat and that any plants remaining were anchored to a rocky shore (Hampson, 2000). These results support the idea that the impacts are long-lasting, that prompt remediation is important, and that natural recovery is unlikely.

4.3. Regional significance

Small scale observations of shoreline loss and recovery are not easily converted to a landscape estimate because various influences tilt and nudge the geomorphic constraints on wetland form from one land-water equilibrium position to another. Channel size will match carrying capacity (Allen, 1989), for example, and there are subtle influences of tidal flats on wetland form and wave, wind fetch and currents influencing geomorphic form and function (Fagherazzi and Wiberg, 2009; Marani et al., 2011, Mariotti and Fagherazzi, 2013). These influences confound experimental design.

The relative significance of these shoreline changes can be put within the context of the regional erosion of coastal wetlands. The pre-2010 erosion rate (m y^{-1}) of the oiled and unoiled islands was about 1.5 to 1.6% y^{-1} of the island width and length, respectively. The island shrinkage for newly oiled islands was 4.8% y^{-1} in the first few months, but 2.9% y^{-1} over 2.5 years, which compares to an average contraction of the unoiled islands of about 1.5% y^{-1}. The higher contraction rates of oiled wetland islands occurring in the few months immediately after the oiling are 5.6 times the peak loss rates in coastal wetland that occurred from 1955/6 to 1978 (0.85% y^{-1}; Baumann and Turner, 1990), and twelve times higher than the average loss rates in the deltaic plain of 0.4% y^{-1} from 1988 to 2011 (Couvillion et al., 2011). Wetland losses on island through erosion is considerably faster than on the larger mass of wetlands attached to the mainland in this case.

A cumulative wetland erosion due to the oiling for the measured interval was estimated to be the average increased shoreline erosion rate of the oiled islands (1.54 m y^{-1}) × years elapsed (2.51 y) × the cumulative area of oiled wetland in Louisiana (1055 km; Nixon et al., 2016), equal to 4.1 km^2, or 1.6 km^2 y^{-1}. This amount compares to the 23 km^2 y^{-1} wetland loss in the Barataria and Terrebonne estuaries from 1998 to 2011 (Couvillion et al., 2011). The estimate of the cumulative wetland-to-water conversion caused by oiling assumes that the island erosion rate applies to all oiled wetlands, which is unlikely because of the variance in the different suite of physical and biological factors affecting island stability. The rate is clearly higher for wetlands on islands, than on the contiguous wetlands, and so this is an overstatement of wetland area lost. In aggregate, though, islands are a sentinel of future losses – being more sensitive to the shoreline stressors than nearby wetlands embedded in a contiguous wetland landscape.

The 50% difference between the erosion rates of oiled and non-oiled islands was not statistically significant for the interval 'd' ($p = 0.13$), which is when Hurricane Isaac made landfall (August 2012). The effects of this hurricane do not appear to have been a significant influence on erosion rates, because shoreline erosion at the unoiled sites remained below the average 16 year pre-spill rates, and were equivalent to the loss rates in the three preceding sampling intervals (a, b and c; Fig. 2).

We could find no recent estimates of wetland dredge-and-fill losses for 2010 to 2013. These permitted losses, largely issued for oil and gas canal construction in wetlands, are casually related to the total wetland losses in time and space (Turner, 1997; Turner, 2009). Estimates of the canal density up to 2001, however, suggest that the area of new canals increased to around 1 $km^2\ y^{-1}$ (Turner, 1997). The indirect losses from a new canal results in 5 times more loss than these direct effect, i.e., a total of around 6 $km^2\ y^{-1}$ (Turner, 1997). This preliminary estimate of shoreline erosion from the DWH disaster in the first 2.5 years after the spill is, therefore, less than from the direct + indirect losses from the annual wetland dredge-and-fill permits issued by the State-Federal permitting programs.

4.4. Future conditions

Petroleum hydrocarbons can persist in coastal wetlands for decades (Teal et al., 1992; Reddy et al., 2002) and the effects on wetlands linger for years, even decades. Reddy et al. (2002), for example, concluded that pristane, phytane and other branched alkanes were still present 30 years after the West Falmouth oil spill (1969, MA) and observed that "hydrocarbon contamination will persist indefinitely in the sedimentary record". The growth, condition index, and filtration rate of the ribbed mussel (*Geukensia demissa*), which is co-located at the base of the dominant emergent salt marsh grass (*S. alterniflora*), was affected by remnants of oil deposited 38 years earlier (Culbertson et al., 2007). Indeed, enhanced erosion continued in 2001 after the Bouchard 65 oil barge release of No. 2 fuel oil in 1974 (Peacock et al., 2007), and the Chalk Point oil spill wetland remained toxic to amphipods after 7 years (Maryland, Michel et al., 2009). These results suggest that the legacy of the oil spill may be a continuing stressor on the emergent plants that have a cascading effect on the interdependent ecosystem structure and functions.

The loss of salt marsh vegetation has, of course, consequences to storm attenuation, shoreline protection and various so-called 'ecosystem services' (Gedan et al., 2011; Shepard et al., 2011). One consequence is the permanent loss of an important limiting habitat for

Fig. 4. One of the last brown pelican nests on the Cat Island rookery, Barataria Bay, LA, USA. The island eroded and lost all of its vegetation after the 2010 oil spill, and pelicans abandoned it in 2012. This photo is offered with the photographer's permission (Tyrone Turner/www.tyronefoto.com).

commercial fisheries species (Turner, 1977) and another is breeding habitat for important migratory birds.

For example, 92 of the 319 nesting seabird colonies on islands in coastal Louisiana are on Barataria and Terrebonne islands (Fontenot et al., 2012), including eleven (11) islands with brown pelican nesting colonies out of the total of 30 sites, of which 10 are not islands. The Cat Island brown pelican rookery, north of Grand Isle, LA, shrank to <2 ha after the 2010 oil spill (Fig. 4) and was abandoned in 2012 when it consisted of only shell hash (Figs. 3F and G).

One potential mitigation intervention to use in for future oiling events might seem to be to fertilize the oiled wetland as a way to speed up the microbial decomposition of the oil, thereby decreasing its toxicity. Venosa et al. (2002) found that this did not enhance bio-decomposition of oil, but did enhance aboveground plant growth. Nutrient additions, however, may be disadvantageous for other reasons. Field observation in Louisiana and Massachusetts peaty wetlands demonstrate that the biomass of salt marsh roots and rhizomes is reduced and that soil strength declines with the addition of nutrients (Deegan et al., 2012; Darby and Turner, 2008a,b; Howes et al., 2010; Kearney et al., 2011; Turner, 2011). Wigand et al. (2015), in contrast, report that +NP amended experimental plots in minerogenic salt marshes of South Carolina (USA) had higher standing stocks of roots and rhizomes (but not fine roots) in the 0 to 20 cm layer.

We conclude that the damage to plants from the oiling demonstrates their significance in reducing shoreline erosion. The losses were immediate, continued for several years, and were irreversible. The oil-spill related losses compromised the longevity of islands within an eroding coast, caused the loss of important bird rookery habitat, and diminished various ecosystem functions, including important ecosystem services of social significance.

Acknowledgments

This analysis was made possible by a grant from the Gulf of Mexico Research Institute funded Coastal Waters Consortium at the Louisiana Universities Marine Consortium. The financial sources had no role in the design or execution of the study, data analysis, decision to publish, or manuscript preparation. We thank D. Daigle for his critical and constructive review of the draft manuscript and thank the anonymous journal reviewers. The data used in this paper are in the Supporting Online Materials and publicly available through the Gulf of Mexico Research Initiative Information & Data Cooperative (GRIIDC) at https://data.gulfresearchinitiative.org (DOI: http://doi.org/10.7266/N72J68WS).

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.marpolbul.2016.06.046.

References

Alexander, S.K., Webb, J.W., 1985. Seasonal response of *Spartina alterniflora* to oil. Proceedings of the 1985 Oil Spill Conference. American Petroleum Institute, Washington, DC, pp. 355–357.

Allen, J.R.L., 1989. Evolution of salt-marsh cliffs in muddy and sandy systems: a qualitative comparison of British west-coast estuaries. Earth Surf. Process. Landf. 14, 85–92.

Baumann, R.H., Turner, R.E., 1990. Direct impacts of outer continental shelf activities on wetland loss in the central Gulf of Mexico. Environ. Geol. Water Res. 15, 189–198.

Boufadel, M.C., Abdollahi-Nasab, A., Geng, X., Galt, J., Torlapati, J., 2014. Simulation of the landfall of the Deepwater Horizon oil on the shorelines of the Gulf of Mexico. Environ. Sci. Technol. 48, 9497–9505.

Couvillion, B.R., Barras, J.A., Steyer, G.D., Sleavin, W., Fischer, M., Beck, H., Trahan, N., Griffin, B., Heckman, D., 2011. Land area change in coastal Louisiana from 1932 to 2010. US Geol. Surv. Sci. Investig. Map 3164 (scale 1:265,000, 12 p. pamphlet).

Culbertson, J.B., Valiela, I., Olsen, Y.S., Reddy, C.M., 2007. Effect of field exposure to 38-year- old residual petroleum hydrocarbons on growth, condition index, and filtration rate of the ribbed mussel, *Geukensia demissa*. Environ. Pollut. 154, 312–319.

Darby, F.A., Turner, R.E., 2008a. Effects of eutrophication to salt marsh roots, rhizomes, and soils. Mar. Ecol. Prog. Ser. 363, 63–70.

Darby, F.A., Turner, R.E., 2008b. Below- and aboveground biomass of *Spartina alterniflora*: response to nutrient addition in a Louisiana salt marsh. Estuar. Coasts 31, 326–334.

Deegan, L.A., Johnson, D.S., Warren, R.S., Peterson, B.J., Fleeger, J.W., Fagherazzi, S., Wollhein, W.M., 2012. Coastal eutrophication as a driver of salt marsh loss. Nature 490, 388–392.

Fagherazzi, S., Wiberg, P.L., 2009. Importance of wind conditions, fetch, and water levels on wave-generated shear stresses in shallow intertidal basins. J. Geophys. Res. 114, F03022.

Feagin, R.A., Lozada-Bernard, S.M., Ravens, T.M., Möller, I., Yearger, K.M., Baird, A.H., 2009. Does vegetation prevent wave erosion of salt marsh edges? Proc. Natl. Acad. Sci. U. S. A. 106, 10109–10113.

Ferrell, R.E., Seneca, E.D., Linthurst, R.A., 1984. The effects of crude oil on the growth of *Spartina alterniflora* Loisel. and *Spartina cynosuroides* (L.) Roth. J. Exp. Mar. Biol. Ecol. 83, 27–39.

Fontenot, W.R., Cardiff, S.W., DeMay, R.A., Dittmann, D.L., Hartley, S., Jeske, C.W., Lorenz, N., Michot, T.C., Purrington, R.D., Seymour, M., Vermillion, W.G., 2012. A Catalog of Louisiana's Nesting Seabird Colonies. Barataria-Terrebonne National Estuary Program, Thibodaux, LA. Report Number 34.

Francalanci, S.M., Bendoni, S., Rinaldi, M., Solari, L., 2013. Ecomorphodynamic evolution of salt marshes: experimental observations of bank retreat processes. Geomorphology 195, 53–65.

Gedan, K.B., Altieri, A.H., Bertness, M.D., 2011. The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. Clim. Chang. 106, 7–29.

Hampson, G.R., 2000. Destruction and recovery of the Winsor Cover, Cataumet, MA salt marsh from a #2 fuel oil spill: a 25 year history. Environ. Cape Cod 3, 32–39.

Hampson, G.R., Moul, E.T., 1978. No. 2 fuel oil spill in Bourne, Massachusetts: immediate assessment of the effects on marine invertebrates and a 3-year study of growth and recovery of a salt marsh. J. Fish. Res. Bd. Canada 35, 731–744.

Hershner, C., Lake, J., 1980. Effects of chronic oil pollution on a salt-marsh grass community. Mar. Biol. 56, 163–173.

Howes, N.C., FitzGerald, D.M., Hughes, Z.J., Georgiou, I.Y., Kulp, M.A., Miner, M.D., Smith, J.M., Barras, J.A., 2010. Hurricane-induced failure of low salinity wetlands. Proc. Natl. Acad. Sci. U. S. A. 107, 14014–14019.

Kearney, M.S., Riter, C.A., Turner, R.E., 2011. Freshwater diversions for marsh restoration in Louisiana: twenty-six years of changing vegetative cover and marsh area. Geophys. Res. Lett. 38, L16405. http://dx.doi.org/10.1029/2011GL047847.

Khanna, S., Maria, J., Santos, J., Ustin, S.L., Koltunov, A., Kokaly, R.F., Roberts, D.A., 2013. Detection of salt marsh vegetation stress and recovery after the Deepwater Horizon oil spill in Barataria Bay, Gulf of Mexico using AVIRIS data. PloS One 8 (11), e78989. http://dx.doi.org/10.1371/journal.pone.0078989.

Lin, Q., Mendelssohn, I.A., Suidan, M.T., Lee, K., Venosa, A.D., 2002. The dose-response relationship between no. 2 fuel oil and the growth of the salt marsh grass, *Spartina alterniflora*. Mar. Pollut. Bull. 44, 897–902.

Marani, M., D'Alpaos, A., Lanzoni, S., Santalucia, M., 2011. Understanding and predicting wave erosion of marsh edges. Geophys. Res. Lett. 38, L21401. http://dx.doi.org/10.1029/2011GL048995.

Mariotti, G., Fagherazzi, S., 2013. Critical width of tidal flats triggers marsh collapse in the absence of sea-level rise. Proc. Natl. Acad. Sci. U. S. A. 110, 5353–5356.

McClenachan, G., Turner, R.E., Tweel, A.W., 2013. Effects of oil on the rate and trajectory of Louisiana marsh shoreline erosion. Environ. Res. Lett. 8, 044030.

McKee, K.L., Patrick Jr., W.H., 1988. The relationship of smooth cordgrass (*Spartina alterniflora*) to tidal datums: a review. Estuaries 11, 143–151.

McNutt, M.K., Camilli, R., Crone, T.J., Guthrie, G.D., Hsieh, P.A., Reyerson, T.B., Savas, O., Shaffer, F., 2012. Review of flow rate estimates of the Deepwater Horizon oil spill. Proc. Natl. Acad. Sci. U. S. A. 109 (50), 20260–20267.

Mendelssohn, I.A., McKee, K.L., Patrick Jr., W.H., 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaptation to anoxia. Science 214, 439–441.

Michel, J., Kirchner, E.R., 2002. Effects of wet meadow riparian vegetation on streambank erosion: part 2. Measurements of vegetated bank strength and consequences for failure mechanics. Earth Surf. Process. Landf. 27, 687–697.

Michel, J., Nixon, A., Dahlin, J., Betenbaugh, D., White, M., Burton, D., Turley, S., 2009. Recovery of interior brackish marshes seven years after the chalk point oil spill. Mar. Pollut. Bull. 58, 995–1006.

Michel, J., Owens, E.H., Zengel, S., Graham, A., Nixon, Z., Allard, T., Horton, W., Reimer, P.D., Lamarche, A., White, M., Rutherford, N., Childs, C., Mauseth, G., Challenger, G., Taylor, E., 2013. Extent and degree of shoreline oiling: Deepwater Horizon oil spill, Gulf of Mexico, USA. PLoS One 8, e65087.

National Oceanic and Atmospheric Administration (NOAA), 2013. ERMA Deepwater Gulf response. (http://gomex.erma.noaa.gov/erma.html#/x=-89.89486&y=29.45903&z=9&layers=10012+1094+2685 and http://gomex.erma.noaa.gov/erma.html#x=89.88671&y=29.50386&z=12&layers=10012 and http://gomex.erma.noaa.gov/erma.html#/x=-90.15175&y=29.47547&z=10&layers=10012+23809, accessed November 2013).

Nixon, Z., Zengel, S., Baker, M., Steinhoff, M., Fricano, G., Rouhani, S., Michel, J., 2016. Shoreline oiling from the Deepwater Horizon oil spill. Mar. Pollut. Bull. 107, 170–178.

Peacock, E.E., Hampson, G., Nelson, R.K., Xu, L., Frysinger, G.S., Gaines, R.B., Farrington, J.W., Tripp, B.W., Reddy, C.M., 2007. The 1974 spill of the Bouchard 65 oil barge: petroleum hydrocarbons persist in Winsor Cove salt marsh sediments. Mar. Pollut. Bull. 54, 2214–2225.

Reddy, C.M., Eglinton, T.I., Hounshell, A., White, H., Xu, L., Gaines, R.B., Frysinger, G.S., 2002. The West Falmouth oil spill: the persistence of petroleum hydrocarbons in marsh sediments. Environ. Sci. Technol. 36, 4754–4760.

Shepard, C.C., Crain, C.M., Beck, M.W., 2011. The protective role of coastal marshes: a systematic review and meta-analysis. Plos One http://dx.doi.org/10.1371/journal.pone.0027374.

Silliman, B.R., van de Koppel, J., McCoy, M.W., Diller, J., Kasozi, G.N., Earl, K., Adams, P.N., Zimmerman, A.R., 2012. Degradation and resilience in Louisiana salt marshes after the BP–Deepwater Horizon oil spill. Proc. Natl. Acad. Sci. U. S. A. 109, 11234–11239.
Strauss, B.H., 2013. Rapid accumulation of committed sea-level rise from global warming. Proc. Natl. Acad. Sci. U. S. A. 110, 13699–13700.
Teal, J.M., Farrington, J.W., Burns, K.A., Stegeman, J.J., Tripp, B.W., Woodin, B., Phinney, C., 1992. Falmouth oil spill after 20 years: fate of fuel oil compounds and effects on animals. Mar. Pollut. Bull. 24, 607–614.
Turner, R.E., 1977. Intertidal vegetation and commercial yields of penaeid shrimp. Trans. Am. Fish. Soc. 106, 411–416.
Turner, R.E., 1997. Wetland loss in the northern Gulf of Mexico: multiple working hypotheses. Estuaries 20, 1–13.
Turner, R.E., 2009. Doubt and the values of an ignorance-based world view for wetland restoration: coastal Louisiana. Estuar. Coasts 32, 1054–1068.
Turner, R.E., 2011. Beneath the wetland canopy: loss of soil marsh strength with increasing nutrient load. Estuar. Coasts 33, 1084–1093.
Van Eerdt, M.M., 1985. Salt marsh stability in the Oosterschelde. Vegetatio 62, 367–373.
Venosa, A.D., Lee, L., Suidan, M.T., Garcia-Blanco, S., Cobanli, S., Moteb, M., Hairnes, J.R., Tremblay, G., Hazelwood, M., 2002. Bioremediation and biorestoration of a crude oil-contaminated freshwater wetland on the St Lawrence River. Bioremediat. J. 6, 261–281.
Wigand, C., Davey, E., Johnson, R., Sundenberg, K., Morris, J.T., Kenny, P., 2015. Nutrient effects on belowground organic matter in a minerogenic salt marsh, North Inlet, SC. Estuar. Coasts 38, 1838–1853. http://dx.doi.org/10.1007/s12237-014-9937-8.
Wu, T.H., 2013. Root reinforcement of soil: review of analytical models, test results, and applications to design. Can. Geotech. J. 50, 259–274.
Zengel, S., Michel, J., 2013. Deepwater Horizon Oil Spill: Salt Marsh Oiling Conditions, Treatment Testing, and Treatment History in Northern Barataria Bay, Louisiana (Interim Report October 2011). U.S. Dept. of Commerce, NOAA Technical Memorandum NOS OR&R 42, Seattle, WAhttp://response.restoration.noaa.gov/deepwater_horizon.
Zengel, S., Bernik, B.M., Rutherford, N., Nixon, Z., Michel, J., 2015. Heavily oiled salt marsh following the Deepwater Horizon oil spill, ecological comparisons of shoreline cleanup treatments and recovery. PLoS One http://dx.doi.org/10.1371/journal.pone.0132324.