4.6 Nearshore Marine Ecosystem

What Is in This Section?

- **Introduction (Section 4.6.1):** What is the Gulf of Mexico nearshore ecosystem and why is it important?
- **Approach to the Assessment (Section 4.6.2):** How did the Trustees assess injury to the nearshore ecosystem?
- **Exposure (Section 4.6.3):** How, and to what extent, were the nearshore habitats and associated species exposed to *Deepwater Horizon* oil (and response activities)?
- **Estuarine Coastal Wetlands Complex Injury** Assessment **(Section 4.6.4):** How were coastal wetlands and associated species affected by *Deepwater Horizon* oil and response activities? What was the magnitude of the injury?
- **Subtidal Oyster Assessment (Section 4.6.5):** How were subtidal oysters affected by *Deepwater Horizon* oil and response activities? What was the magnitude of the injury?
- **Beach Assessment (Section 4.6.6):** How were beaches and associated species affected by *Deepwater Horizon* oil and response activities? What was the magnitude of the injury?
- **Shallow Unvegetated Habitats—Gulf Sturgeon Assessment (Section 4.6.7):** How were shallow unvegetated habitat and Gulf sturgeon species affected by *Deepwater Horizon* oil and response activities? What was the magnitude of the injury?
- **Submerged Aquatic Vegetation Assessment (Section 4.6.8):** How was submerged aquatic vegetation and associated species affected by *Deepwater Horizon* oil and response activities? What was the magnitude of the injury?
- **Conclusions and Key Aspects of the Injury for Restoration Planning (Section 4.6.9):** What are the Trustees' conclusions about injury to nearshore habitats, associated species, ecosystem effects, and restoration considerations?
- **References (Section 4.6.10)**

Executive Summary

The nearshore marine ecosystem of the northern Gulf of Mexico is a vast, biologically diverse collection of interrelated habitats that stretch from Texas to Florida. These nearshore habitats are among the most biologically productive coastal waters in the United States. They provide food, shelter, and nursery grounds for many ecologically and economically important animals that use the Gulf's open waters, including fish, shrimp, shellfish, sea turtles, birds, and mammals. In this way, the nearshore ecosystem fundamentally supports the entire Gulf of Mexico ecosystem (including offshore habitats) and provides myriad services that humans value.

Almost all types of nearshore ecosystem habitats in the northern Gulf of Mexico were oiled and injured as a result of the *Deepwater Horizon* (DWH) oil spill. Oil was observed on more than 1,300 miles (2,113 kilometers) of shorelines from Texas to Florida. By state, Louisiana had the majority of oiled shoreline (approximately 65 percent) and the vast majority of oiled wetland shorelines (95 percent). Most of the observed oiling in the nearshore zone occurred along the shoreline edge. Six hundred miles (965 kilometers) of beaches were oiled, causing ecological injury and affecting human use. The geographic extent of shoreline oiling is the largest of any marine spill globally (Zachary Nixon et al. 2015a).

Oiling caused multiple injuries to marsh habitats, including reductions in aboveground biomass and total plant cover in mainland herbaceous salt marshes, reductions in periwinkle snail abundance, reductions in shrimp and flounder growth rates, reduced reproductive success in forage fish, reduced amphipod survival, and reduced nearshore oyster cover. These injuries were observed over 350 to 721 miles (563 to 1,160 kilometers) of shoreline. Increased erosion of oiled shorelines has also been documented over at least 108 miles (174 kilometers) of coastal wetlands. Additional injuries include:

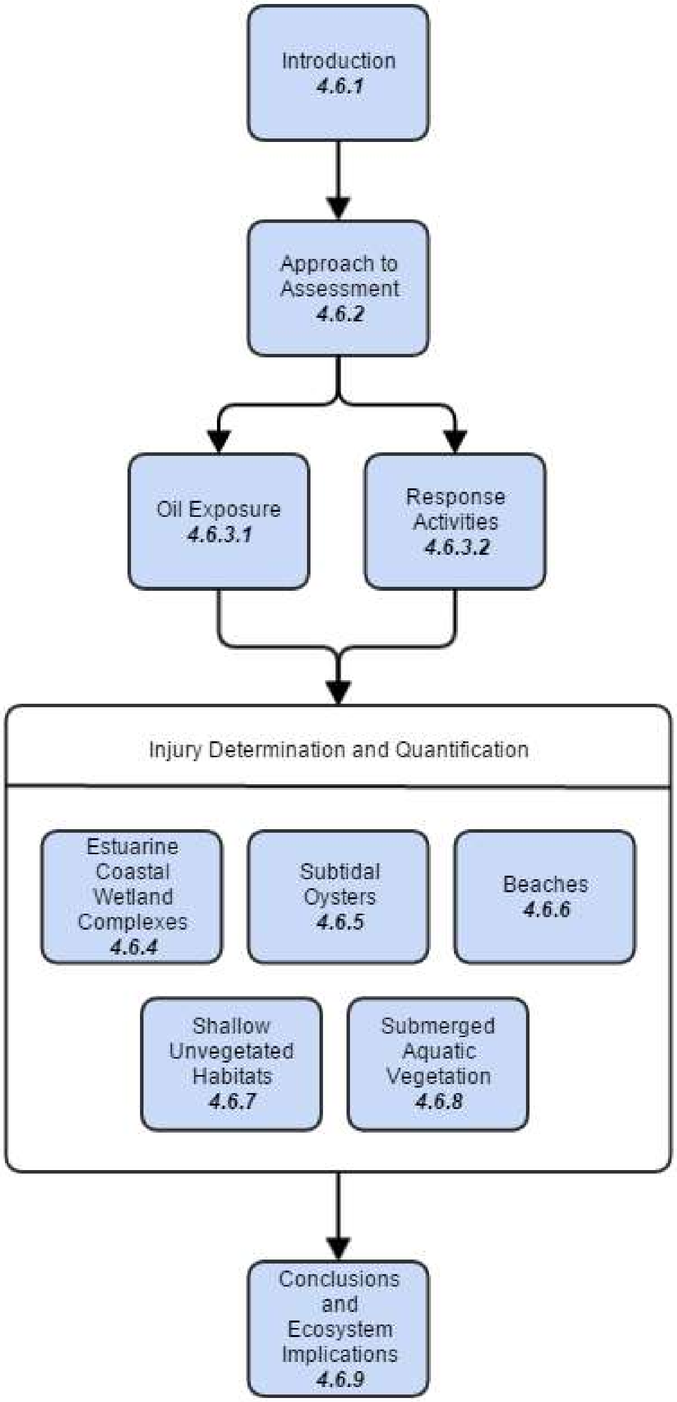
- Billions of subtidal oysters were killed by releases of river water from response actions and—when combined with effects to nearshore oysters from shoreline oiling—exhibit long-term recruitment problems over a large area of the Gulf of Mexico.
- Beach shorelines were affected by oiling and response actions, with the most severe cleanup actions killing all creatures that burrow in beach sand.
- Unvegetated nearshore sediment systems were also affected, as indicated by injury to threatened Gulf sturgeon along shorelines of Louisiana, Mississippi, Alabama, and Florida.
- Submerged aquatic vegetation (SAV) habitats were lost from oiling and from physical disturbance as part of response actions. Chandeleur Islands SAV, which is uniquely valuable in the region, was particularly affected, with more than 270 acres (109 hectares) of seagrass destroyed. Injuries to SAV habitats were also documented within the boundaries of Gulf Islands National Seashore and in Jean Lafitte National Historic Park and Preserve.

Some of these losses are permanent. For example, marsh edge erosion and destruction of nearshore oyster cover can only be addressed through restoration. Subtidal oyster recruitment may slowly recover over time, or the spill-related losses may have been so severe that restoration will be required to initiate recovery. Injuries to marsh flora and fauna will persist until oil concentrations in marsh soils fall below levels that are toxic to the most sensitive prey species and life stages. Populations of long-lived species (e.g., periwinkle snails, sturgeon) will take years to recover normal age/size distributions, even after environmental conditions are no longer toxic. The largest patches of SAV, which spread slowly through rhizomes, will also take decades to recover.

Addressing injuries to these marsh habitats will require special attention. Gulf salt marshes are productive because of their intricate complexity. Sinuous tidal channels that maximize edge habitat provide fauna access to flooded marsh surfaces for refuge and forage and promote rapid growth of juvenile fish and invertebrates of commercial importance. The marsh edge was the most severely oiled and most severely injured, but marsh edge is productive because it is part of a more complex adjacent

system. Nearshore oysters that can stabilize vegetated edge habitats will be vital to compensate for injuries.

The following flow chart provides a road map to Section 4.6 (Nearshore Marine Ecosystem). The chart appears at the start of each subsection with the corresponding subsection box highlighted.

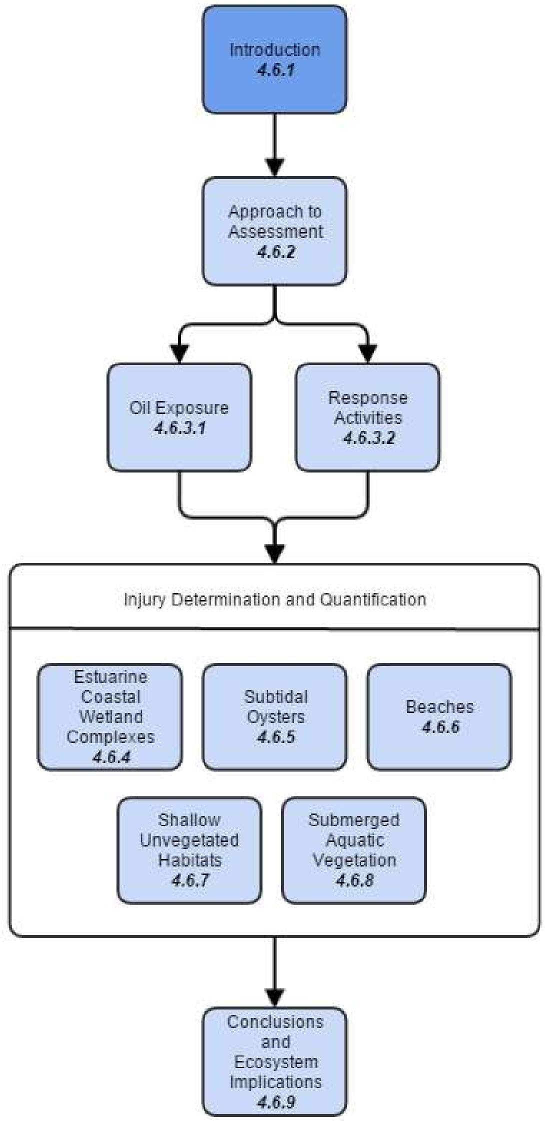


4.6 Executive Summary

4.6.1 Introduction

4.6.1.1 Ecological Description

The nearshore marine ecosystem of the northern Gulf of Mexico is a vast, biologically diverse collection of interrelated habitats that stretch from Texas to Florida. The habitats comprising this ecosystem include marshes, mangroves, beaches and dunes, barrier islands, SAV, oyster reefs, and shallow unvegetated areas. These nearshore habitats are among the most biologically productive coastal waters in the United States. They provide food, shelter, and nursery grounds for many ecologically and economically important animals, including fish, shrimp, shellfish, sea turtles, birds, and mammals. In this way, the nearshore ecosystem fundamentally supports the offshore ecosystem.



4.6.1.1.1 Ecological and Economic Importance

The northern Gulf of Mexico nearshore marine ecosystem provides myriad ecosystem services, including protection of the shoreline from erosion and flooding, feeding and nesting habitat, nutrient cycling, water quality improvement, and carbon sequestration (Mitsch & Gosselink 2007; UNEP 2007). The northern Gulf of Mexico nearshore ecosystem is particularly recognized for its provision of food, refuge, and nursery habitat for commercially important crustacean, fish, and shellfish species (Moody & Aronson 2007). Nearshore ecosystems are among the most ecologically valued in the world in terms of ecosystem services provided per unit area (Costanza et al. 1997; Costanza et al. 2014).

The economic contributions of the northern Gulf of Mexico nearshore marine ecosystem are significant. Many of the region's most important commercial and recreational fisheries include species that spend all or part of their lives in the nearshore environment (Peterson & Turner 1994; Zimmerman et al. 2000). For instance, the nearshore-dependent penaeid shrimp represents the largest northern Gulf of Mexico fishery by revenue. Other economically important nearshore fisheries include blue crabs, oysters, and menhaden (NMFS 2012). The Gulf of Mexico commercial shrimp fishery is critical to the livelihood of coastal fisherman: in 2009, more than 4,700 vessels actively participated in the inshore, nearshore, and offshore segments of the fishery. In 2009, ex-vessel revenue for the Gulf-wide shrimp fishery was $314 million (NMFS 2011). The commercial oyster fishery is also economically valuable: prior to the *Deepwater Horizon* incident, the Gulf of Mexico oyster fishery annual harvest was valued at

approximately $60 million (NMFS 2012), with 69 percent of U.S. oyster landings from the northern Gulf of Mexico (Turner 2006). The nearshore environment serves as a critical habitat in early developmental stages for many economically important finfish species (Able 2005). For U.S. fisheries as a whole, approximately 68 percent of commercial catch and 80 percent of recreational catch is dependent on nearshore environments (Lellis-Dibble et al. 2008).

The nearshore environment provides various other recreational and human use services. In addition to recreational fishing, beach-going has significant economic value in the Gulf states. Coastal wetlands also support birdwatching and hunting. Wetlands and barrier island environments also offer protection from storm events, which has great economic value. The Mississippi River Delta alone is estimated to provide at least $12 to $47 billion annually in ecosystem services associated with hurricane and flood protection, water supply, water quality, recreation, and fisheries (Batker et al. 2014).

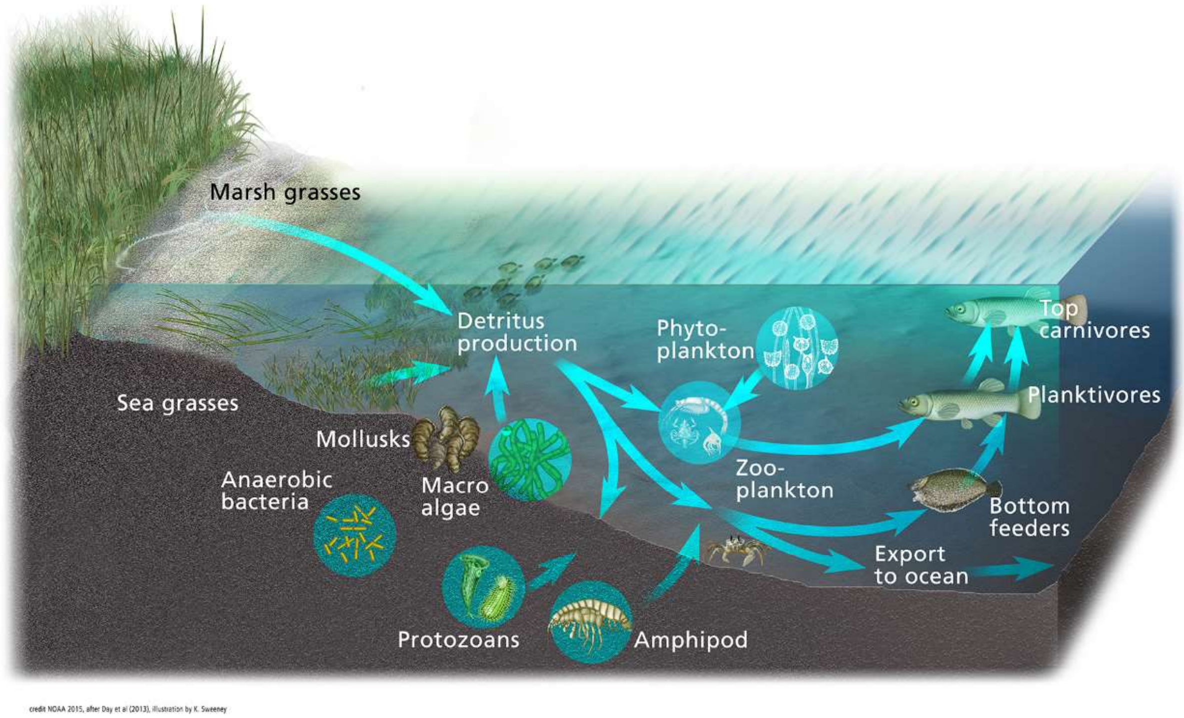
In addition, because of their unique ecological importance, many of the Gulf's habitats are federal trust resources and are protected as national parks, seashores, and wildlife refuges. These federal trust resources include various habitats (e.g., coastal wetlands, marsh, SAV, beaches, sand dunes) that support a diverse array of species. While these habitats also occur at other locations, Congress carefully selected these lands to be conserved as a whole; and these lands typically serve as a foundation of a natural resource conservation system upon which other local efforts are built (National Park Service 2014) (National Wildlife Refuge Administration Act 1966).

4.6.1.1.2 Nearshore Estuary Food Web Dynamics

Nearshore estuarine ecosystems support food webs that tend to be complex. This complexity is a result of the interactions that occur among the different subsystems (e.g., salt marsh, oyster reef; Figure 4.6-1) and series of food webs. An extremely important feature of estuarine food webs is the estuarine bottom:

- Various plants grow in the shallow water sediments (e.g., marsh grasses, SAV, and benthic algae). Decomposing plant material is an important food in estuaries (Mann 1988).
- Food and inorganic nutrients flow from the water column to the bottom, and in the opposite direction.
  - Benthic organisms filter water for food, and some move over and through sediments and take food from the sediment itself.
  - Numerous other organisms also feed on the bottom, including many invertebrates (e.g., shrimp, crab), fish, and birds.
  - The flow of energy from phytoplankton, detritus, and bottom sediments converges upon top carnivores that are generalist feeders on various organisms. These top carnivores include many species of fish (e.g., sea trout, red drum, and flounder), birds (e.g., sea gulls, wading birds), and mammals (e.g., dolphins, manatee). The flow of energy from primary producers to top predators is exemplified for marsh species in the trophic pyramid in Figure 4.6-2.

If oil injures lower levels of the food web (e.g. amphipods), it can impact all of these organisms.



Source: *Kate Sweeney for NOAA.*

Figure 4.6-1. Food web diagram for a typical estuarine ecosystem showing some feeding links among selected major trophic groupings. Lines and arrows indicate flow of food from source to consumer.

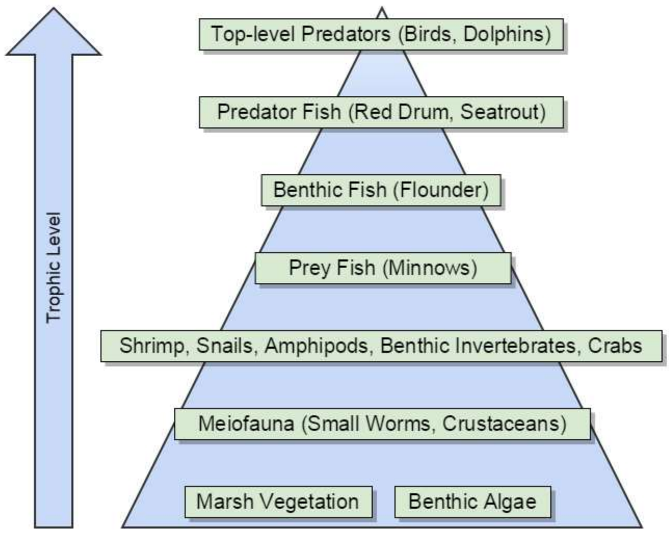


Figure 4.6-2. Simplified trophic pyramid for salt marsh species in the northern Gulf of Mexico. Primary producers such as marsh vegetation and benthic algae form the base of the nearshore food web, providing nutrients to other organisms as well as habitat. Injuries to marsh vegetation initiate a cascade of impacts to organisms at higher trophic levels.

4.6.1.2 Habitats of the Northern Gulf of Mexico Nearshore Ecosystem

The northern Gulf of Mexico nearshore ecosystem comprises numerous interconnected habitats (Figure 4.6-3). These nearshore habitats often occur adjacent to one another forming a complex mosaic of structural refuge and foraging habitat for fish, invertebrates, terrestrial animals, and migrating waterfowl (Grabowski et al. 2005; Powers & Scyphers 2015).

4.6.1.2.1 Estuarine Coastal Wetlands Complex

The estuarine coastal wetlands complex is comprised of coastal wetlands, adjacent nearshore waters, mudflats, and associated fauna, including nearshore oysters. Coastal wetlands are one of the most common habitats of the coastal Gulf of Mexico, particularly in Louisiana (Minello et al. 2003). Gulf of Mexico wetlands are an integral part of the estuarine food web. They also provide habitat for migratory and resident birds, mammals, insects, arachnids, protozoa, fish, and benthic fauna (e.g., crustaceans, mollusks, and nematodes). Benthic fauna of Gulf of Mexico wetlands and mudflats provide food for birds, fish, and other organisms, assist in the breakdown of detritus, increase microbial activity and productivity, oxygenate sediments, and help maintain healthy levels of nutrients in sediments (Carman et al. 1997; Curry et al. 2001). Nearshore oysters (i.e., those located within 50 meters of shore), which are included in the coastal wetland habitat complex, form clusters on and adjacent to the marsh edge. They provide various ecosystem functions, such as habitat to marsh fauna and shoreline stabilization.

Coastal wetland habitat serves as a key base of the productive Gulf of Mexico food web. This habitat supports animals using the marsh surface (e.g., shrimp, snails, fish, crabs, and insects) and animals residing adjacent to the marsh (e.g., nearshore oysters) (Peterson & Howarth 1987). The composition of the vegetative community varies according to region, salinity, tidal inundation, and other characteristics related to shoreline type.

Salt marshes in the northern Gulf of Mexico are characterized by smooth cordgrass (*Spartina alterniflora*), which often occurs in pure stands or with black rush (*Juncus roemerianus*), saltgrass (*Distichlis spicata*), and other, less common species. Salt marshes may be found on the mainland or on the sheltered side of barrier islands. Back-barrier salt marshes are high-energy environments that often contain coarse sediment that has washed in from the seaward (beach) side. These marshes are also lower in soil organic matter than mainland salt marshes (Hester & Willis 2015a).

Another type of coastal wetland habitat in the northern Gulf of Mexico is the mangrove-salt marsh complex, which was evaluated in Louisiana. In this habitat, mangroves exist at the northern limit of their range in "stunted" form. Mangrove habitats are primarily comprised of a mixture of black mangrove (*Avicennia germinans*) and herbaceous halophytes, such as smooth cordgrass (*Spartina alterniflora*) (Willis & Hester 2015a). Mangroves are woody, halophytic trees or shrubs that inhabit low-energy coastal areas throughout the tropics and subtropics (Snedaker et al. 1996). Mangrove roots trap sediment, stabilize shorelines, and build islands. They serve as nesting habitat for many coastal birds (e.g., brown pelicans) and as nursery habitat for crustaceans and fish (Day et al. 2013; Houck & Neill 2009).

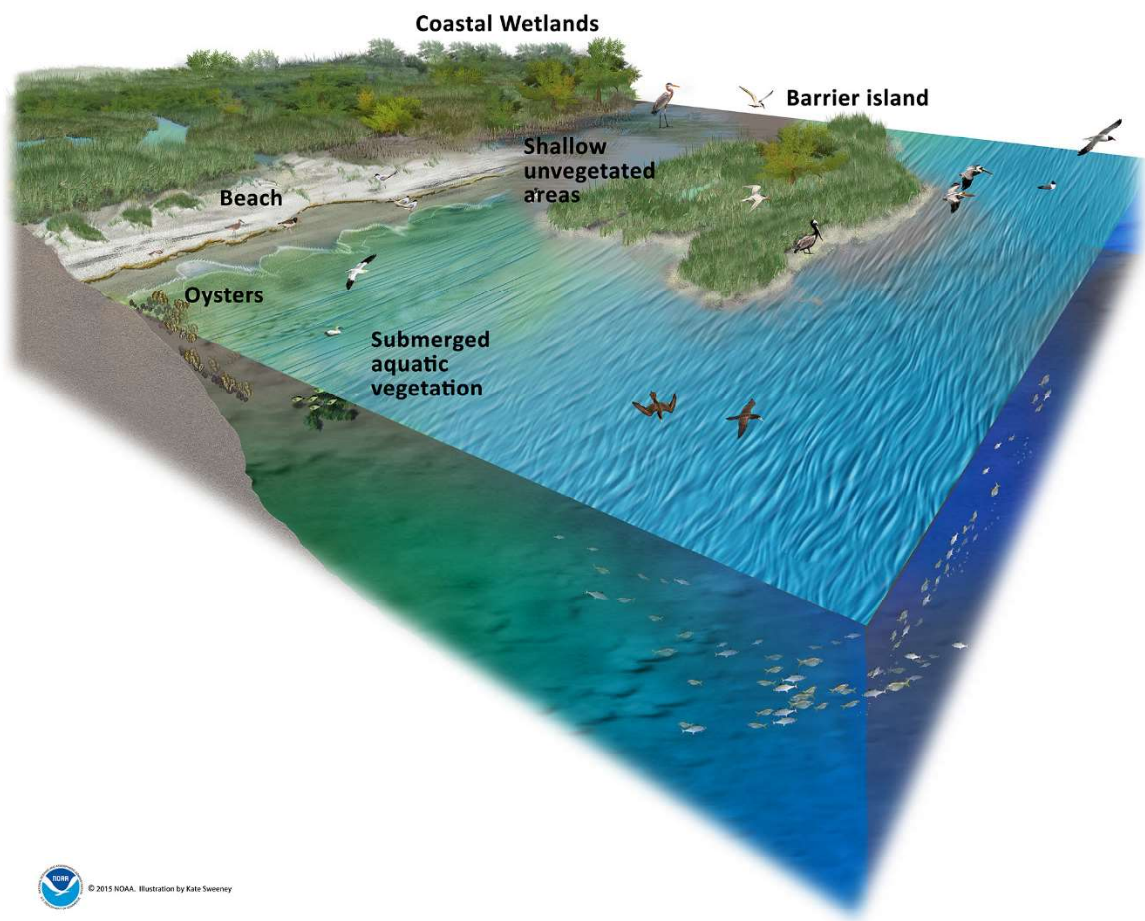
The Delta *Phragmites* marsh is found in the unique hydrology of the Mississippi River Deltaic Plain, which supports wide swaths of pure stands of the common reed (*Phragmites australis*). Freshwater input from the Mississippi River creates a brackish environment favored by the species. These marshes are extensively flooded due to high flow of the Mississippi River and substantial exposure to wind and wave energy. As a result, Delta *Phragmites australis* marshes rarely, if ever, drain (Hester & Willis 2015c).

4.6.1.2.2 Oyster Reefs

Oysters in the northern Gulf of Mexico form nearshore and subtidal reefs, comprised of the eastern oyster (*Crassostrea virginica*), a filter-feeding shellfish. Reefs are natural accumulations of oyster shell built over time by the growth of multiple generations. Subtidal oysters (i.e., those greater than 50 meters from shore) are most abundant in semi-enclosed bays, preferring water depths less than 12 meters and salinities between 15 and 30 parts per thousand; and these oysters generally do not tolerate sustained freshwater inputs (VanderKooy 2012). Oyster reefs provide a wide range of ecological functions that support other Gulf of Mexico coastal habitats, including salt marshes, SAV, and surrounding unvegetated areas (Coen et al. 2007; Meyer et al. 1997; Scyphers et al. 2011). These subtidal oyster reefs are among the most productive in the world, and the northern Gulf of Mexico subtidal reefs support a robust oyster fishery (LDWF 2011). In addition, oyster reefs, like salt marshes and SAV beds, serve as an important habitat for many species of crabs, fishes, and birds. As one example, oyster reefs are an important habitat for the American oystercatcher—a shorebird closely tied to coastal habitats that include intertidal oyster beds. Because of their reef-building capabilities, oysters are commonly referred to as natural ecosystem engineers. Oysters also improve water quality and shoreline stabilization (Powers et al. 2015a).

Nearshore oysters form fringing reefs or smaller hummocks on salt marsh shorelines, on intertidal mudflats, and between salt marshes and seagrass beds. In most Gulf states, these fringing reefs are not harvested and thus, serve as de facto sanctuary areas for oysters (Powers et al. 2015b). The oysters

contribute larvae that eventually settle in subtidal areas and are especially important in stabilizing marsh shorelines by providing hard structure and trapping sediment (Powers et al. 2015b).



Source: *Kate Sweeney for NOAA.*

Figure 4.6-3. The northern Gulf of Mexico nearshore ecosystem comprises numerous inter-connected habitats. These nearshore habitats often occur adjacent to one another forming a complex mosaic of structural refuge and foraging habitat for fish, invertebrates, terrestrial animals, and migrating waterfowl.

4.6.1.2.3 Beaches and Dunes

Sand beaches and dunes are found along mainland shorelines throughout the northern Gulf of Mexico (Figure 4.6-4). They are also found along the outer shorelines of barrier islands, on barrier spits, and on bars along passes (e.g., Southwest Pass and South Pass in Louisiana). These beaches and the coastal strand habitat are integral to the northern Gulf of Mexico ecosystem, playing many important ecological, recreational, and economic roles.

Northern Gulf of Mexico sand beaches and dunes are home to numerous organisms, including small crabs, clams, shrimp, and snails. These organisms live and feed on and within the sand and beach wrack (i.e., decomposing vegetation washed up on the shore by the surf). These small organisms in turn serve

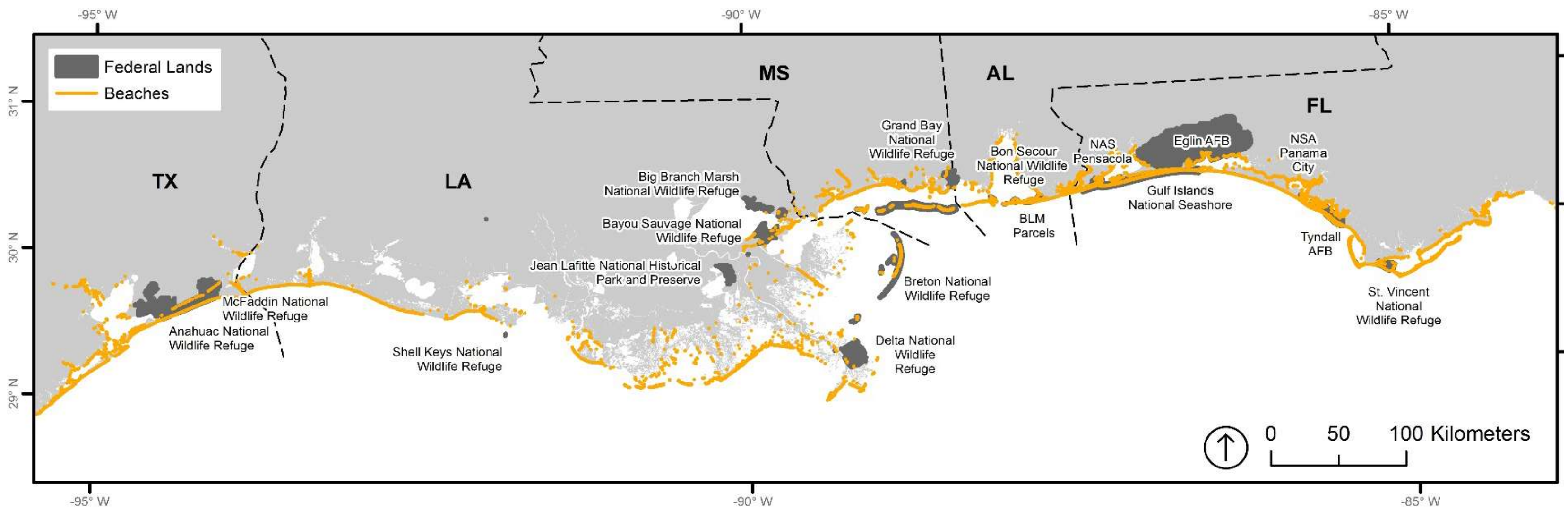
as an important food base for birds, mammals, and other animals that forage on the beaches. Sand beaches and dunes of the northern Gulf of Mexico are also nesting habitat for many federally-listed threatened or endangered turtles, mammals, and birds. Notably, the endangered loggerhead, Kemp's ridley, green and leatherback turtles all nest on sand beaches in the northern Gulf of Mexico (Dow et al. 2007). Several federally-listed endangered beach mice—the Perdido Key beach mouse, the Choctawhatchee beach mouse, the St. Andrew beach mouse, and the Alabama beach mouse—live their entire lives in coastal dunes; these mice species nest in the dunes and forage on the dune vegetation and in beach wrack (FWS 2006).

The beaches and dunes of the northern Gulf of Mexico are also important nesting and foraging habitat for many resident and migratory bird species. For example, Louisiana has identified many state species of greatest conservation need that nest on the state's barrier island beaches, including the American oystercatcher, the Wilson's plover, the brown pelican, and the least tern (LDWF 2011). Further, coastal beaches are home to approximately 70 percent of the wintering population of the federally-listed threatened piping plover (Elliott-Smith et al. 2009).

Sand beaches and dunes also provide a physical buffer, protecting habitat and human communities from storms and hurricanes. Beaches and dunes along the seaward facing side of northern Gulf of Mexico barrier islands protect the bays, estuaries, and wetlands behind them from the destructive forces of storms and hurricanes (Sutten-Grier et al. 2015). In addition to the ecological benefits provided, beaches and dunes provide many different recreational opportunities, including swimming, fishing, and sunbathing. This section focuses on natural resource injuries to sand beaches and dunes; see Section 4.10 for information on recreational losses.

4.6.1.2.4 Shallow Unvegetated Areas

Shallow unvegetated areas often comprise large portions of coastal and estuarine systems. These areas include mudflats or tidal flats, which are intertidal areas exposed at low tide. These structurally simple areas have been recognized as important habitats for economically significant crustaceans, such as blue crabs (*Callinectes sapidus*) (Lipcius et al. 2005). Tidal flats are an important foraging habitat for the piping plover, a globally threatened or endangered (depending on the population) migratory bird that winters in the northern Gulf of Mexico (Haig 1987). An important resident of shallow unvegetated areas is the Gulf sturgeon (*Acipenser oxyrinchus desotoi*), a threatened species under the Endangered Species Act of 1973, as amended (FWS & NOAA 1991). The Gulf sturgeon is a bottom-feeding, anadromous fish that migrates from salt water into large coastal rivers to spawn (FWS & GSMFC 1995; FWS & NOAA 1991, 2003b).



Source: *Zachary Nixon, Research Planning, Inc.*

Figure 4.6-4. Roughly 4,530 km of shoreline along the U.S. Gulf of Mexico (inclusive of the Florida Keys) can be described as sand or sand-and-shell beach, as derived from NOAA ESI data. Approximately 965 km of these beaches were impacted by the DWH oil spill (NOAA 1995a, 1995b, 2003, 2007, 2010).

4.6.1.2.5 Submerged Aquatic Vegetation

Submerged aquatic vegetation (SAV) beds are an important component of the nearshore ecosystem. SAV beds are submerged, rooted, vascular plants. These flowering plants grow in the open northern Gulf of Mexico and in saline, brackish, and fresh estuaries (SAV species found in saline waters are called seagrasses). By some estimates, the northern Gulf of Mexico has more than 50 percent of the total U.S. distribution of seagrasses and at least 5 percent of the known global occurrences (Green & Short 2003).

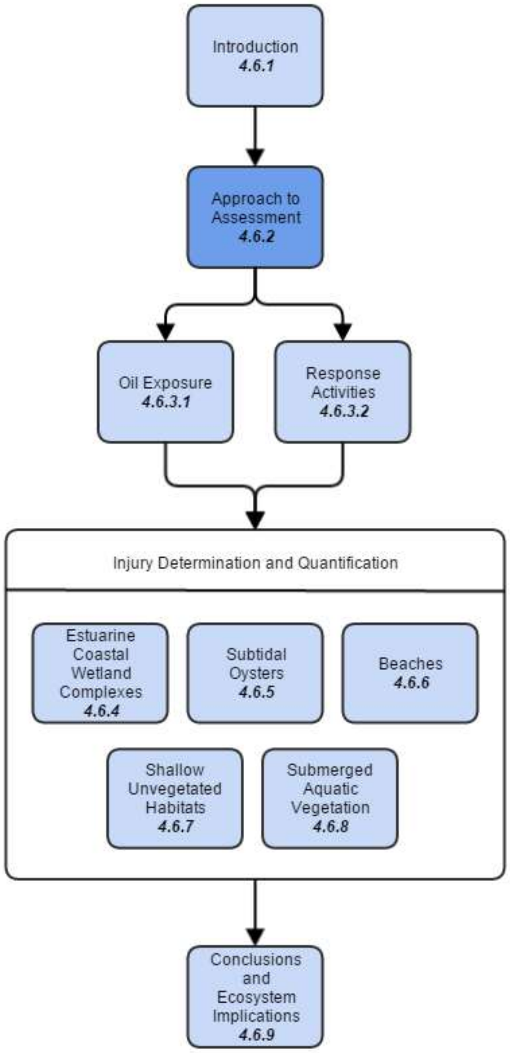
Freshwater SAV is a particularly important resource at the Barataria Preserve, a unit of the Jean Lafitte National Historical Park and Preserve in Louisiana (Poirrier et al. 2010). For several reasons, the seagrass beds inside the Chandeleur Islands are unique: they are the only existing marine seagrass beds in Louisiana; they are the largest, most continuous seagrass bed in the northern Gulf of Mexico; and they are part of the Breton National Wildlife Refuge, the second-oldest refuge in the National Wildlife Refuge System. These barrier Islands are prolific environs where hundreds of species of finfish, crustaceans, birds, and other wildlife flourish (Poirrier & Handley 2007).

SAV beds provide many ecological functions. They are the basis for a large amount of secondary productivity, a diverse food web, important biogeochemical processes, and one of the primary indicators of good water quality (Cosentino-Manning et al. 2015). They are key habitats for diverse fish and invertebrates, providing abundant food for consumers and complex physical structures where animals can find refuge from their predators. The physical structure of seagrass beds creates high-friction sea-bottom that damps tidal currents and surface waves and helps suspend and stabilize sediments. These plant beds are also important centers for biogeochemical processes that involve the cycling and transformation of carbon, nitrogen, phosphorus, and other key elements (EPA 2000).

4.6.2 Approach to the Assessment

Key Points

- The Trustees developed a conceptual model, outlining the oil pathway, oil exposure to resources, and mechanisms of injury to those resources.
- The Trustees selected natural resources in the ecosystem to serve as key indicators to evaluate effects due to oiling.
- The Trustees conducted studies to: 1) document whether a resource was *exposed* to oil or response actions (exposure studies) and 2) document whether *injury* occurred (injury studies).
- Mechanisms of injury to plants and animals from oiling include both physical smothering and toxicity from ingestion and dermal exposure.
- Mechanisms of injury to plants and animals from response efforts include intolerance of low salinity water, reduced food quality/quantity, and physical smothering/disturbance.



4.6.2.1 Overview of Assessment Approach

To assess the effects of the *Deepwater Horizon* oil spill on the nearshore ecosystem, the Trustees conducted numerous studies of key habitat types and resources. The Trustees' assessment approach was driven by a conceptual model of the pathways and mechanisms by which oil and response actions could have affected nearshore resources.

Because it was not logistically possible to study the entire nearshore ecosystem, the Trustees selected components of the ecosystem to serve as key indicators of a complex system. Many selected components are considered keystone, foundational, or indicator species. Selection was based on some combination of the following factors:

- Importance of functional role in ecosystem.
- Representation of various trophic levels, exposure pathways, life stages, and life histories.
- Prevalence.
- Societal value.
- Known sensitivity to oil or *Deepwater Horizon* response actions.

The multifaceted approach was intended to evaluate various injuries, including lethality, impaired growth, impaired reproduction, and other measured or observed adverse effects. The components considered in this system (see previous list) provides a framework for understanding impacts across the ecosystem. However, these components do not fully reflect all injury to the ecosystem or account for compounding effects of individual injury components.

4.6.2.2 Description of the Approach to Assessment

The northern Gulf of Mexico nearshore ecosystem is a complex, interrelated system. As described above, key indicators were assessed to represent the health of the broader ecosystem. The Trustees' assessment was organized by the following predominant nearshore habitat types, with one or more indicators selected within each habitat type:

- Coastal wetlands.
- Subtidal oyster reefs.
- Beaches and dunes.
- Shallow unvegetated areas.
- SAV beds.

Injuries to nearshore surface water from oil exposure, though relevant to the nearshore ecosystem, are addressed in Section 4.4, Water Column.

In addition to potential injuries due to oiling, response actions taken as a result of the *Deepwater Horizon* spill caused injury to the nearshore ecosystem: summer river water releases implemented to decrease the likelihood of oil reaching the nearshore area adversely affected oysters, shrimp, and SAV (Powers et al. 2015a); response vessels left propeller scars in SAV beds; stranding of boom in marsh; and beach cleaning to remove oil from the sand disturbed beach infauna (i.e., animals living in sediment). These potential injuries were also assessed.

Studies achieved one of two broad objectives: 1) documenting whether the resource was *exposed* to oil or response actions (exposure studies) and 2) documenting whether *injury* occurred (injury studies).

4.6.2.2.1 Exposure Studies

Exposure studies generally focused on pathways resulting from oil interaction with the shoreline. These studies documented oil components on or in coastal wetland soils and beaches, nearshore sediments, the surf mixing zone, and tissues of nearshore animals. These studies were intended to represent the various pathways by which this cohesive and connected ecosystem was likely exposed.

Exposure of the nearshore environment to oil was documented through field surveys. These surveys were conducted under the *Deepwater Horizon* response and the NRDA. Shoreline oiling was evaluated along the northern Gulf of Mexico coast from Texas to Florida by survey teams on foot and by boat. In this section, shoreline is defined as the land/water interface and was generally comprised of coastal wetland and beach habitats. Many shoreline stretches were surveyed numerous times in the months following the spill. Visual observations of oiling were recorded, and oil samples were collected to confirm the presence of MC252 oil. These shoreline surveys not only indicated exposure of coastal wetlands and beaches to oil, but also indicated exposure to subtidal oyster reefs, shallow unvegetated areas, and SAV beds over which the oil traveled before reaching shore. In addition to the shoreline

surveys, exposure studies specific to SAV beds were conducted, whereby sorbent materials were placed in SAV beds to indicate the presence of oil. Observations of nearshore surface oiling are discussed in Section 4.2 (Natural Resource Exposure) and Section 4.4 (Water Column).

Visual observations of oiling were paired with total polycyclic aromatic hydrocarbons (TPAH50) chemistry of sediment, soil, and tissue samples. Forensic analysis (chemical fingerprinting) was also used to identify the likelihood of MC252 oil.

4.6.2.2.2 Injury Studies

The approach to evaluating injury to the nearshore environment was multi-dimensional. The Trustees conducted field studies and laboratory toxicity testing using representative test species and MC252 oil. The assessment also considered data collected outside the NRDA where relevant.

Field studies were conducted across the spectrum of oiling conditions, from areas where heavy oiling persisted over time to areas where oiling did not occur. Toxicity studies also tested a range of oiling conditions. Many studies spanned multiple years to capture the effects of oiling over time and potential recovery to pre-spill conditions.

The assessment also included use of numeric models and assumption-based calculations to estimate injury or provide interpretive information.

4.6.2.2.3 Conceptual Model: Pathway, Exposure, and Injury

Figure 4.6-5 outlines the oil pathway, oil exposure to representative resources, and mechanisms of injury to the representative species and habitats. Mechanisms of injury from *oiling* included physical smothering, toxicity by ingestion, and toxicity by dermal exposure. Mechanisms of injury from *response actions* included intolerance of low salinity water, reduced food quality/quantity, physical smothering, and physical disturbance.

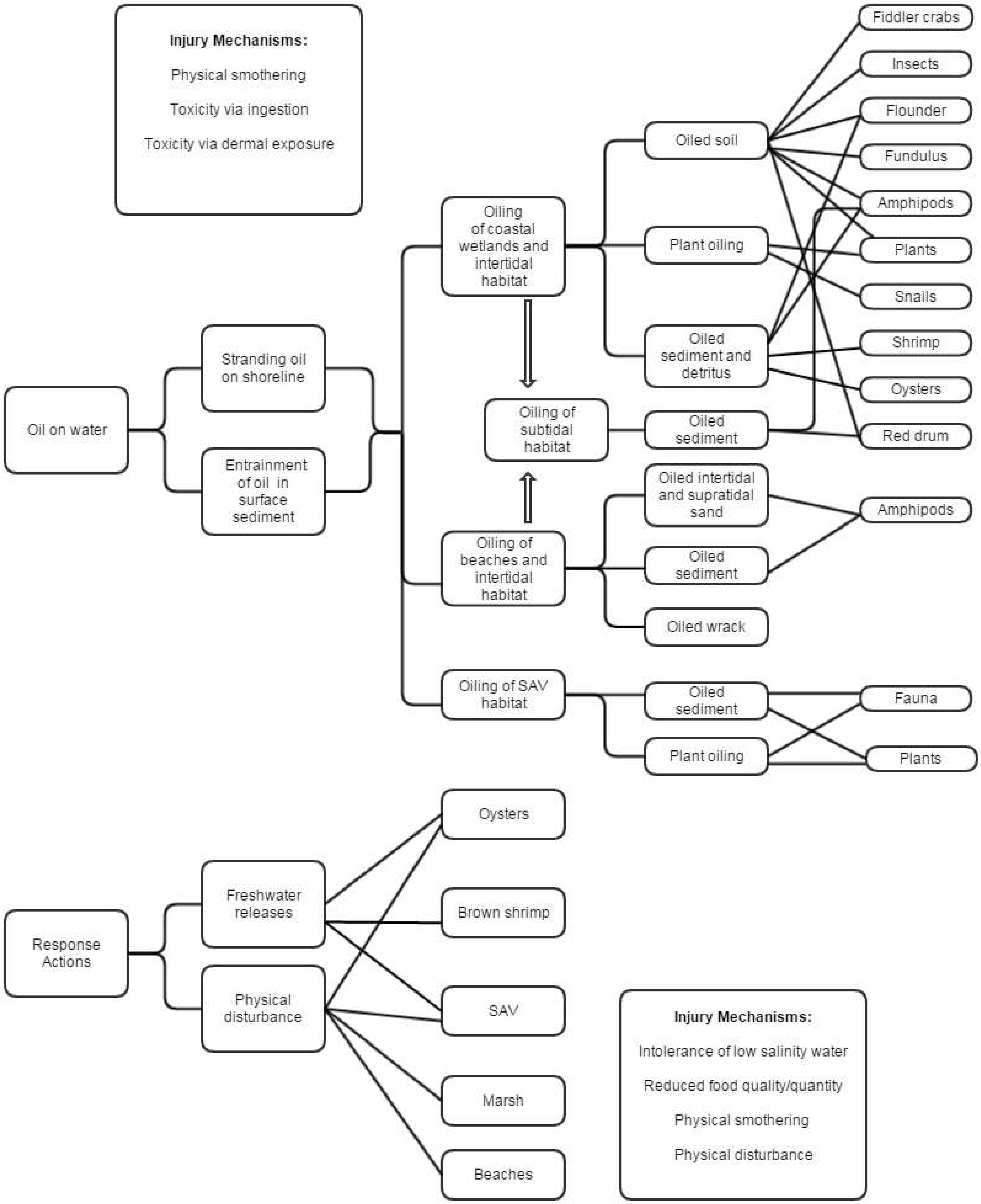


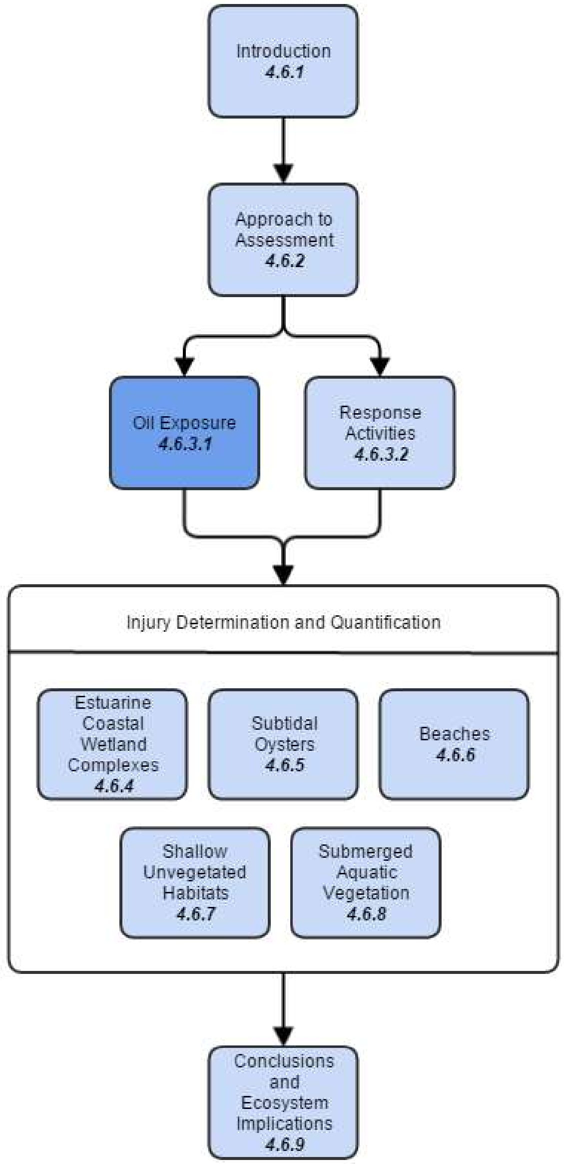
Figure 4.6-5. Pathways of exposure of representative species to oil and response actions and mechanisms of injury. The diagram illustrates the complexity of the interactions among the oil and response actions and the nearshore resources evaluated. Most resources were exposed via multiple pathways.

4.6.3 Exposure

4.6.3.1 Exposure to Oil

Key Points

- Oil was observed on more than 1,300 miles (2,100 kilometers) of shoreline from Texas to Florida, with samples collected from many areas documenting the presence of MC252 oil.
- Coastal wetland soils, nearshore ocean sediments, and tissues of SAV and nearshore animals were evaluated for TPAH50 concentrations as part of the nearshore assessment (see Section 4.2, Natural Resource Exposure).
- For Louisiana mainland salt marsh soils, fall 2010 TPAH50 concentrations along oiled shorelines were orders of magnitude higher than ambient concentrations or those measured at "no oil observed" sites. In other Louisiana coastal wetland habitats and in Mississippi and Alabama, TPAH50 concentrations also tended to correspond to shoreline oiling categories, and concentrations decreased over time.
- More than one year after the spill, TPAH50 concentrations in sediments collected 0-50 meters offshore of Louisiana mainland salt marshes were two to three times higher along heavily oiled shorelines compared to ambient concentrations.
- TPAH50 concentrations in sediments adjacent to unvegetated shorelines were not related to degree of shoreline oiling.
- TPAH50 concentrations in nearshore animal tissue were highly variable and were not correlated to shoreline oiling; however, sample size was very limited.

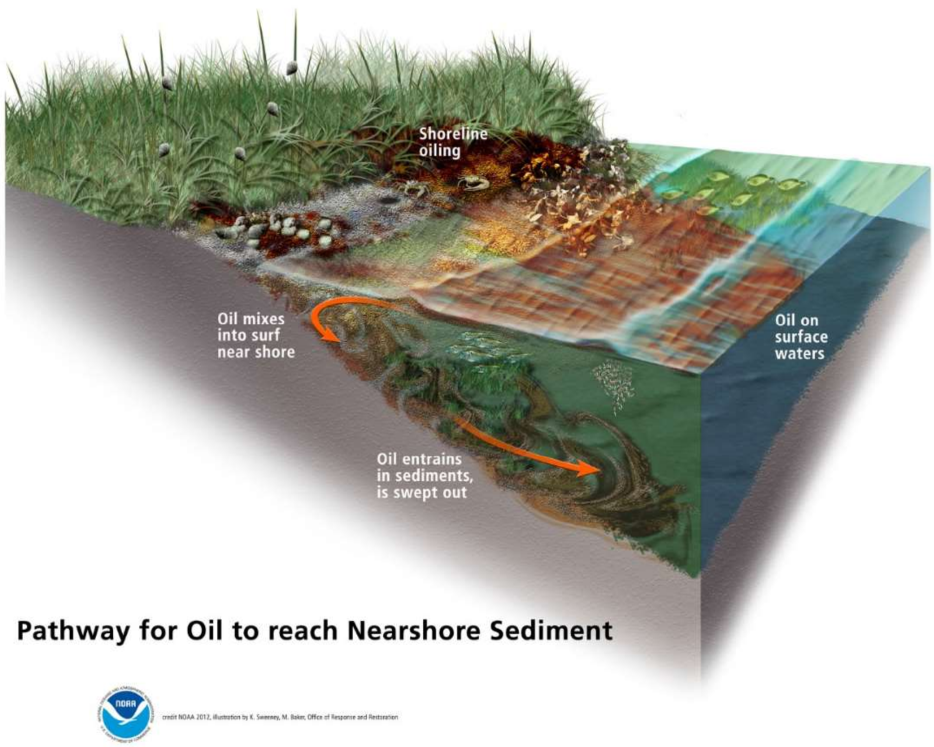


4.6.3.1.1 Pathways

As described in Section 4.2 (Natural Resource Exposure), some portion of the oil that reached sea surface was carried toward shore by wind and currents (Figure 4.6-6). Some of this oil washed up on shore and became "stranded" in several forms, including:

- Discrete tar balls (less than 10 centimeters diameter).
- Patties (10–50 centimeters).
- Oil mats (greater than 50 centimeters).

These forms sometimes occurred as viscous emulsions of oil but more often were mixtures of sand bound by lesser amounts of oil (see Section 4.2, Natural Resource Exposure, for more detail). The stranded oil produced a highly visible impact on hundreds of miles of the region's beaches and coastal wetland marshes during the summer of 2010 (Figure 4.6-7; (Mendelssohn et al. 2012; Michel et al. 2013; OSAT-2 2011). Oil stranded in coastal wetlands typically pooled on the surface rather than penetrating into the marsh soils (Figure 4.6-8; (Michel et al. 2013). In more dynamic beach environments, oil often mixed with the sand and became buried. Also observed along shorelines were oily coatings on rocks, shell hash, wildlife, and stems of coastal wetland vegetation. Nearshore exposure pathways are summarized in Zhang et al. (2015a).



Source: *Kate Sweeney for NOAA.*

Figure 4.6-6. Illustration of oil pathways in a nearshore marsh environment. Oil floating on surface water was carried toward shore. The oil then either stranded onshore or mixed with nearshore sediments. A portion of the oil in nearshore sediments was swept offshore

Source: *NOAA Deepwater Horizon SCAT Program.*

Figure 4.6-7. Heavy oiling conditions in the coastal wetlands of Bay Jimmy, Louisiana, in the months following the spill.

Source: *NOAA Deepwater Horizon SCAT Program.*

Figure 4.6-8. Pooled oil under a coastal wetland vegetation mat in Bay Jimmy, Louisiana, September 2010.

4.6.3.1.2 Observations of Shoreline Oiling

For the purposes of characterizing exposure to nearshore plants and animals, the Trustees used two approaches to describing oil on shorelines. The first approach (used for evaluating exposure to *nearshore animals*) characterizes the degree of oiling on any shoreline (wetlands or beach) based on linear surveys where oiling was observed. The second approach (used for evaluating exposure to *vegetation*) estimates lengths of wetland shoreline where different degrees of plant stem oiling occurred.

For the first approach, shoreline lengths were based on cumulative visual observations of oiling by the response and Trustees from the time of the spill over a period of approximately 4 years. The US Coast Guard (USCG) and other agencies conducted shoreline surveys to characterize and prioritize shorelines for cleanup. These surveys were performed under the Shoreline Cleanup Assessment Technique (SCAT) program, and are described in Section 2.3.8 (Michel et al. 2013). The SCAT survey

What Is SCAT?

The Shoreline Cleanup Assessment Technique (SCAT) program is a well-established and internationally recognized program to characterize shoreline oiling and inform cleanup decisions. The *Deepwater Horizon* SCAT program, which was overseen by the Unified Command, was initiated before oil reached shore. From May 2010, through April 2014, SCAT Teams (composed of representatives from the USCG, NOAA, State, BP, and others as appropriate) surveyed shorelines potentially exposed to oil by foot and by boat. Data collected included visual observations and photographs of: the width, length, thickness, and distribution of oil on the shoreline surface and in the subsurface; the shoreline type; and documentation of oiled wildlife, stranded boom, and other response equipment on the shoreline.

dataset was supplemented with other available observational shoreline oiling data, including those collected during operational cleanup efforts and under the NRDA, most notably the rapid assessment surveys. The rapid assessment survey data describe shoreline oiling in some Louisiana marsh areas between August 14, 2010, and October 16, 2010. These data represent a supplemental source of surface shoreline oiling data for these locations.

Based on these data, oil was observed on at least 1,300 miles (2,100 kilometers) of the 5,931 miles (9,545 kilometers) of shoreline that was surveyed (Zachary Nixon et al. 2015b). These shoreline oiling observations were used to develop oil exposure categories and to estimate oiled shoreline lengths. These categories and associated oiled miles were then used to estimate the degree and extent of exposure in injury assessments to all wetland and beach fauna. Another approach to assessing exposure for coastal wetland plants is described in "Exposure of Coastal Wetland Plants" below.

For both beaches and coastal wetland habitats, oil exposure categories were developed that integrate the intensity and persistence of shoreline oiling (Table 4.6-1; (Zachary Nixon et al. 2015b). For *coastal wetlands*, five shoreline oil exposure categories were used:

- Heavier persistent oiling, where heavy or moderate oiling was repeatedly observed over a period of 12 weeks or longer.
- Heavier oiling, where moderate or heavy oiling persisted for less than 12 weeks.
- Lighter oiling, where only trace to light oiling was observed.
- "No oil observed" during the surveys used for this analysis; however, other data indicates some of these areas ultimately were oiled.
- Shoreline not surveyed during the surveys used for this analysis.

Beaches were classified using a similar framework, but two additional categories were used to account for significant subsurface oiling and persistence over time (see Table 4.6-1). Under this framework, "other" habitats are hardened shorelines such as riprap and rocky shores. The same oil categories for wetlands were applied to the "other" habitat category.

The shoreline was mapped using these oil exposure categories (Figure 4.6-9); and from these maps, the lengths of shoreline were calculated for each exposure class, habitat type (i.e., beach, wetland, or other), and state (Table 4.6-2 and Table 4.6-3) (Zachary Nixon et al. 2015b).

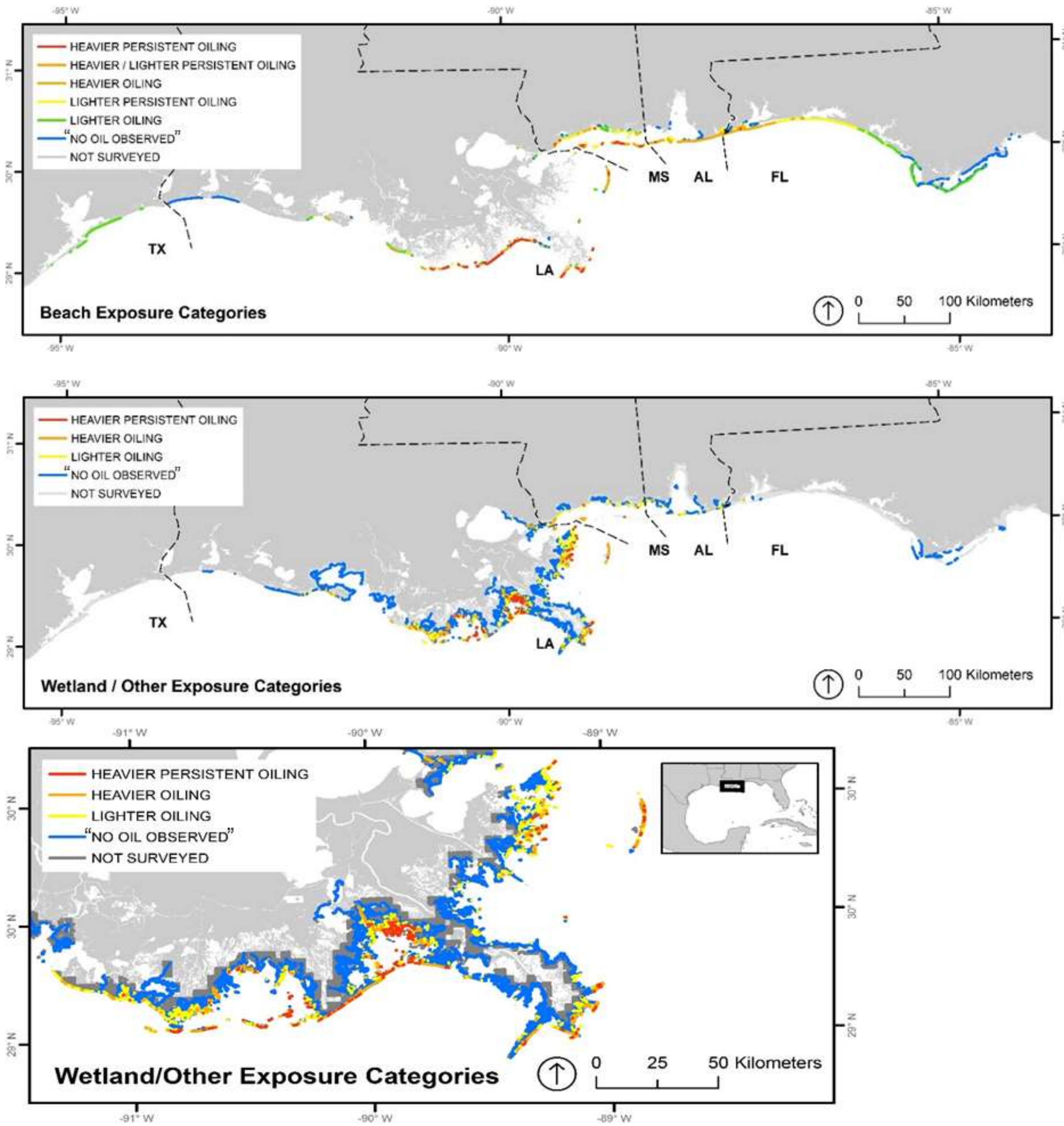
Why Was Oil Found at Locations Designated as "No Oil Observed"?

"No oil observed" is a shoreline category intended to describe areas where oiling was not observed during linear shoreline surveys. The Shoreline Cleanup Assessment Technique (SCAT) survey and NRDA rapid assessment survey were the primary datasets used to inform the oiling categories and estimate oiled shoreline miles for evaluating exposure to wetland and beach animals. If neither survey detected oil in a given area, that area was described as "no oil observed." However, in some instances, oil came ashore after a segment was surveyed. Other field sampling events later found oiling in some of these areas designated as "no oil observed," and some areas likely experienced oil that was never detected.

Table 4.6-1. Oil exposure category definitions for beaches and coastal wetland habitats.

Exposure Category	Definition—Beaches	Definition—Wetlands/Other
NOT SURVEYED*	Not Surveyed	Not Surveyed
"NO OIL OBSERVED"*	No Oil Observed during SCAT or NRDA rapid assessment surveys	"No Oil Observed" during SCAT or NRDA rapid assessment surveys
LIGHTER OILING	Maximum of "Light" or lesser surface or subsurface oiling and persistence of oiling for less than 26 weeks	Maximum of "Light" or lesser surface oiling
HEAVIER OILING	Maximum of "Moderate" or greater surface or subsurface oiling and persistence of oiling for less than 26 weeks	Maximum of "Moderate" or greater surface oiling and persistence of such oiling for less than 12 weeks
LIGHTER PERSISTENT OILING	Maximum of "Light" or less surface or subsurface oiling and persistence of oiling for 26 weeks or longer	Not used
HEAVIER / LIGHTER PERSISTENT OILING	Surface or subsurface oiling of "Moderate" or greater and persistence of "Light" or less surface or subsurface oiling for 26 weeks or longer	Not used
HEAVIER PERSISTENT OILING	Maximum of "Moderate" or greater surface or subsurface oiling and persistence of such oiling for 26 weeks or longer	Maximum of "Moderate" or greater surface oiling and persistence of such oiling for 12 weeks or longer

**NOT SURVEYED category applies to locations not surveyed by the field surveys used in this analysis. "NO OIL OBSERVED" category applies to locations where no oil was observed during the field surveys used in this analysis, but does not mean that no oil ever reached the segment.*



***Source:** Zachary Nixon et al. (2015b).*

Figure 4.6-9. Extent of shoreline oiling by oil exposure categories for beaches (top), coastal wetland and other shoreline habitats across the Gulf (middle), and coastal wetland and other shoreline habitats in Louisiana (bottom). Oil was observed from Texas to Florida.

Table 4.6-2. Lengths of shoreline oiling in miles (and kilometers) by oil exposure categories and state for beaches, coastal wetlands, and other habitats (rounded to the nearest whole number) (Zachary Nixon et al. 2015b).

Exposure Category		"NO OIL OBSERVED"*	LIGHTER OILING	LIGHTER PERSISTENT OILING	HEAVIER OILING	HEAVIER / LIGHTER PERSISTENT OILING	HEAVIER PERSISTENT OILING	TOTAL OILED
FLORIDA	Beaches	236 (380)	63 (101)	76 (123)	0 (0)	37 (60)	1 (1)	176 (284)
	Wetlands	146 (235)	0 (0)	NA	0 (0)	NA	0 (0)	0 (0)
	Other	16 (26)	1 (2)	NA	0 (0)	NA	0 (0)	1 (2)
ALABAMA	Beaches	18 (29)	4 (6)	37 (60)	1 (1)	43 (69)	1 (1)	85 (136)
	Wetlands	62 (100)	4 (7)	NA	0 (0)	NA	0 (0)	4 (7)
	Other	47 (76)	6 (9)	NA	1 (1)	NA	0 (0)	6 (10)
MISSISSIPPI	Beaches	21 (33)	14 (22)	72 (116)	1 (1)	24 (39)	11 (18)	121 (195)
	Wetlands	101 (163)	25 (41)	NA	2 (3)	NA	0 (0)	27 (44)
	Other	17 (27)	9 (15)	NA	0 (0)	NA	0 (0)	9 (15)
LOUISIANA	Beaches	73 (118)	39 (63)	24 (39)	9 (15)	56 (90)	53 (86)	182 (293)
	Wetlands	3,839 (6,178)	439 (707)	NA	171 (276)	NA	45 (72)	656 (1,055)
	Other	42 (68)	6 (10)	NA	2 (4)	NA	1 (2)	10 (16)
TEXAS	Beaches	0 (0)	35 (57)	0 (0)	0 (0)	0 (0)	0 (0)	35 (57)
	Wetlands	0 (0)	0 (0)	NA	0 (0)	NA	0 (0)	0 (0)
	Other	0 (0)	0 (0)	NA	0 (0)	NA	0 (0)	0 (0)
TOTALS	Beaches	348 (560)	154 (248)	209 (337)	10 (16)	160 (258)	65 (105)	600 (965)
	Wetlands	4,148 (6,675)	469 (754)	NA	173 (278)	NA	45 (73)	687 (1,105)
	Other	122 (197)	22 (36)	NA	3 (5)	NA	1 (2)	27 (43)

"NO OIL OBSERVED" category applies to locations where no oil was observed during the field surveys used in this analysis.

4.6.3 Exposure

Table 4.6-3 provides additional detail by presenting lengths of shoreline oiling for different coastal wetland habitats (Zachary Nixon et al. 2015b). For the purpose of deriving shoreline lengths, the Louisiana vegetation types described in Sasser et al. (2014) were applied to Louisiana wetlands. Mainland and back-barrier salt marshes included saline and brackish vegetation types. The category Delta *Phragmites* marsh and other fresh/intermediate marsh was comprised almost entirely of marshes on the Delta; however, the category also included small contributions from fresh and intermediate marshes off the Delta.

Louisiana mainland salt marshes represent the majority of shoreline oiling, with 509 miles (820 kilometers) oiled. The next largest category of oiling was Delta *Phragmites* marshes with 89 miles (143 kilometers), followed by Louisiana back-barrier salt marshes and Mississippi mainland salt marshes both with 18 miles (29 kilometers) (Zachary Nixon et al. 2015b).

Table 4.6-3. Lengths of shoreline oiling for coastal wetland habitat types by state and oiling category (Zachary Nixon et al. 2015b).

ALABAMA	Mainland Salt/Brackish Marsh		Back barrier Salt/Brackish Marsh		Delta/Inland Fresh/Intermediate Marsh		Mangrove/Marsh Complex	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	4	7	0	0	0	0	0	0
HEAVIER OILING	0	0	0	0	0	0	0	0
HEAVIER PERSISTENT OILING	0	0	0	0	0	0	0	0
	4	**7**	**0**	**0**	**0**	**0**	**0**	**0**
MISSISSIPPI	**Mainland Salt/Brackish Marsh**		**Back barrier Salt/Brackish Marsh**		**Delta/Inland Fresh/Intermediate Marsh**		**Mangrove/Marsh Complex**	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	18	29	8	12	0	0	0	0
HEAVIER OILING	0	0	2	3	0	0	0	0
HEAVIER PERSISTENT OILING	0	0	0	0	0	0	0	0
	18	**29**	**9**	**15**	**0**	**0**	**0**	**0**
LOUISIANA	**Mainland Salt/Brackish Marsh**		**Back barrier Salt/Brackish Marsh**		**Delta/Inland Fresh/Intermediate Marsh**		**Mangrove/Marsh Complex**	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	355	571	7	11	50	80	28	45
HEAVIER OILING	116	187	11	18	36	58	8	13
HEAVIER PERSISTENT OILING	39	62	0	0	3	5	3	5
	509	**820**	**18**	**29**	**89**	**143**	**39**	**63**

Although based on a compilation of results from several large surveys, these oiled shoreline lengths are not inclusive of all shoreline oil observations. The extent of actual oiling is thus likely greater than reported here. Further, no survey observed all northern Gulf of Mexico shorelines, and surveying a given segment of shoreline did not guarantee that all oil on that segment was observed. Oil can be difficult to find in marshes, and oil sometimes washed ashore after segments were surveyed.

Although the 2008 shoreline was the standard used by SCAT to support cleanup operations, it represents the land-water interface at a relatively low tide 2 years prior to the spill. More importantly, because of the spatial resolution of the 2008 shoreline layer, it does not capture many details of the vegetated land-water interface where most marsh oiling occurred. Consequently, marsh shoreline lengths that are based on the 2008 data layer underestimate the actual length of oiled vegetated marsh edge. To investigate the implications of this, the Trustees allocated the information in the shoreline exposure database onto a digital representation of the shoreline from 2010, focusing on the Louisiana marsh habitats where the most oil exposure occurred (Wobus et al. 2015). This analysis indicated that the length of the oiled marsh edge in Louisiana exceeds calculations based on the 2008 shoreline by up to 40 percent in some areas—in other words, the actual shoreline oiling was greater than the estimates reported in this section. It should also be noted that oiled shoreline lengths represent the cumulative shoreline observed to be oiled at any time over 4 years of observations. Some shorelines were oiled only once, and others were repeatedly oiled.

Exposure of Coastal Wetland Plants

While Trustees relied on the shoreline oiling categories and oiled shoreline lengths as the basis for exposure and quantification for many nearshore receptors, another approach was used for coastal wetland plants. For these plants, Trustees conducted a pre-assessment survey to collect shoreline and plant oiling information in 2010 (see Section 4.2, Natural Resource Exposure) (Zachary Nixon et al. 2015a). This extensive dataset was used to evaluate exposure to plants and serve as the basis for sampling coastal wetland vegetation. Locations were divided by habitat type (e.g., marsh versus mangrove) and were further divided into five plant stem oiling categories: reference (0 percent stem oiling and “no oil observed”), 0 to 10 percent, 10 to 50 percent, 50 to 90 percent, and 90 to 100 percent (Zachary Nixon et al. 2015a). As mentioned above, some shoreline locations characterized as “no oil observed” exhibited plant oiling; and when this occurred, those locations were not used as reference locations for other NRDA studies.

These pre-assessment data provided Trustees the ability to estimate miles of coastal wetland impacted based on plant oiling. Two approaches based on different statistical methods were used to calculate ranges of oiled shoreline lengths for the five plant stem oiling categories (Goovaerts 2015; Zachary Nixon et al. 2015a). The range of shoreline length for each plant stem category is shown in Table 4.6-4. In addition to the categories shown in the table, these results also yield an estimated 99 kilometers (61 miles) of unobserved shoreline oiling where plant oiling occurred but where oiling was not documented in the shoreline oiling database (Zachary Nixon et al. 2015a). This represents 1.6 percent of the 6,178 kilometers (3,839 miles) wetland shoreline in Louisiana where no oil was observed during SCAT or rapid assessment surveys (Table 4.6-2).

Table 4.6-4. This table summarizes the estimated miles of coastal wetland vegetation impacted by oiling. Data in the two columns provide a range of miles impacted for each plant stem oiling category (Goovaerts 2015; Zachary Nixon et al. 2015a).

Plant Stem Exposure	Weighted Lengths Based on Shoreline Oiling Classifications Miles (kilometers)	Geographically-Weighted Lengths Based on Linear Regression Miles (kilometers)
90-100% Plant Oiling	47 (76)	67 (109)
50-90% Plant Oiling	78 (125)	140 (225)
10-50% Plant Oiling	152 (245)	390 (628)
0-10% Plant Oiling	73 (117)	124 (199)
Total	**350 (564)**	**721 (1,161)**

4.6.3.1.3 Nearshore Oiling Chemistry

Coastal wetland soils, nearshore sediments, tissues of SAV, and tissues of nearshore animals were evaluated for TPAH50 concentrations as part of the nearshore assessment (see Section 4.2, Natural Resource Exposure for TPAH50 description). TPAH50 concentrations were measured in the nearshore environment as a representation of the toxic effects of oil (see Section 4.3, Toxicity). Samples were also analyzed to identify MC252 oil (Section 4.2, Natural Resource Exposure). Oil concentrations in water were primarily evaluated in the water column assessment (Section 4.2, Natural Resource Exposure; and Section 4.4, Water Column). This section reviews chemistry results across states and different habitat types.

Dispersants were widely applied to offshore environments in the weeks following the spill (Kujawinski et al. 2011). Chemical markers of the dispersants were found in tar balls and sand patties collected from beaches (Section 4.2, Natural Resource Exposure).

Ambient Soil and Sediment TPAH50 Concentrations

Ambient soil and sediment TPAH50 concentrations were calculated to provide a comparison to TPAH50 concentrations at oiled locations. For the purposes of our assessment, soil is considered to be the substrate that supports marsh plants. Extending shoreward of the marsh plants, the substrate is considered to be sediment. Historic data collected before the *Deepwater Horizon* spill were not sufficient to compute comparable ambient TPAH50 concentrations (Zhang et al. 2015a). Thus, a forensic-based approach[1] was applied to post-spill soil and sediment data collected in coastal wetlands and nearshore environments in Louisiana, Mississippi, and Alabama (as described in Section 4.6.3, Exposure; Emsbo-Mattingly 2015; Emsbo-Mattingly & Martin 2015). The approach was used to calculate a range of TPAH50 concentrations representing ambient conditions in the northern Gulf of Mexico (Zhang et al. 2015a). Mean ambient TPAH50 concentrations were found to vary substantially from region to region, with higher values along the Mississippi River Delta shoreline and lower values along undeveloped barrier islands (Table 4.6-5 and Table 4.6-6).

[1] For the forensic-based approach, a coastal wetland soil sample was determined to be a "representative ambient" sample if its forensic match was "Indeterminate" and if it was at least 100 meters from any manifestations of MC252 oil.

Table 4.6-5. Ambient TPAH50 concentrations in coastal wetland soils of the northern Gulf of Mexico. Concentrations were highest in the Mississippi River Delta and lowest on Louisiana's barrier islands (Zhang et al. 2015b).

State/Region	Habitat	Count	Average (ppb)	Standard Deviation (ppb)	Min (ppb)	Max (ppb)
Louisiana	Mainland Herbaceous Salt Marsh	24	278	169	51	737
	Back Barrier Herbaceous Salt Marsh	6	26	20	2	46
	Coastal Mangrove Marsh	20	244	238	0	766
	Delta *Phragmites* Marsh	18	4,278	5,918	1,211	24,448
Mississippi/Alabama—Mississippi Sound	Mainland Herbaceous Salt Marsh	30	254	225	19	953
	Island Herbaceous Salt Marsh	12	130	108	7	358
Alabama—Perdido Bay	Mainland Herbaceous Salt Marsh	9	210	278	3	679

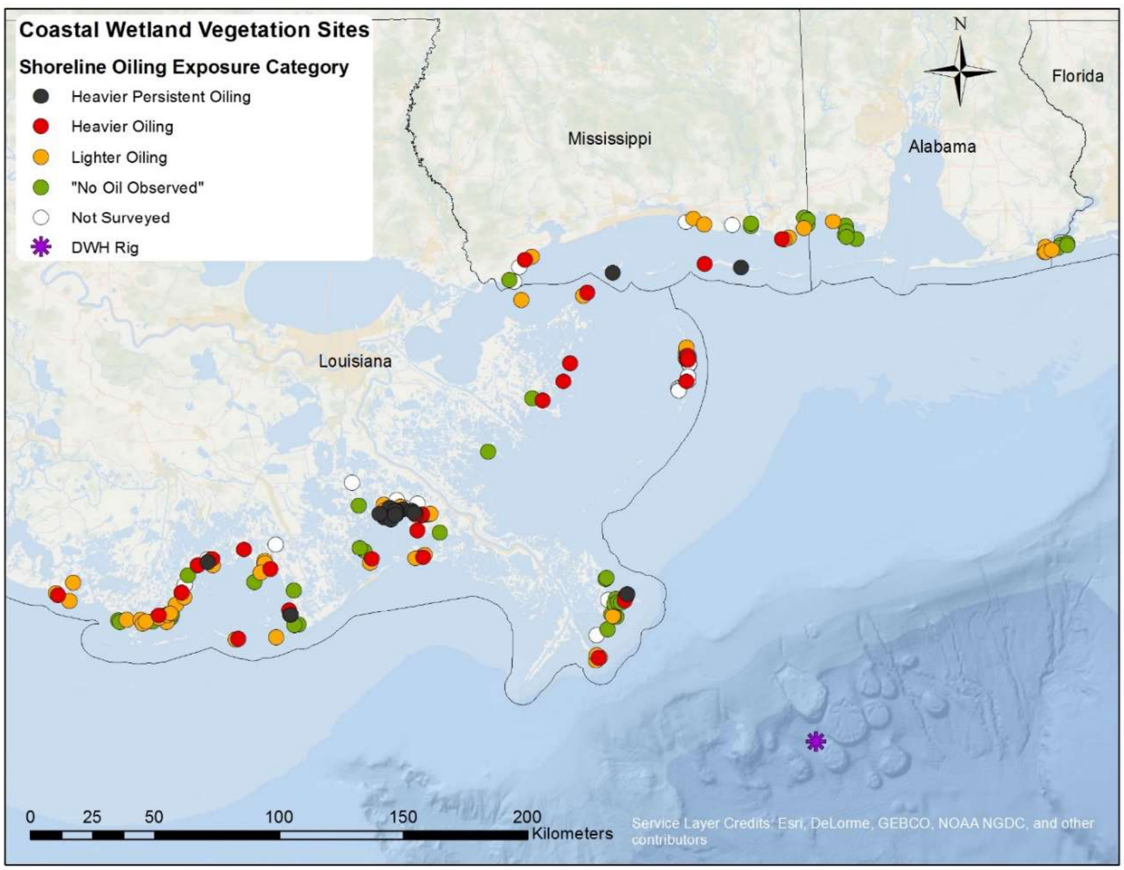
Table 4.6-6. Ambient TPAH50 concentrations in nearshore sediments adjacent to shorelines in the northern Gulf of Mexico. Sediment concentrations were highest adjacent to the Mississippi River Delta and lowest adjacent to the coastal wetlands of Louisiana's barrier islands (Zhang et al. 2015a).

State	Habitat	Distance to Shore (m)	Sample Size	tPAH Concentrations (ppb)			
				Average	Standard Deviation	Min	Max
Louisiana	Unvegetated	0-50	4	718	707	105	1,506
		50-500	43	513	664	0	2,067
	Mainland Herbaceous Salt Marshes	0-50	58	264	422	8	2,934
		50-500	106	167	125	9	828
	Back Barrier Herbaceous Salt Marshes	50-500	5	41	43	7.0	105
	Mangrove/Marsh Complex	0-50	3	74	1	73	75
		50-500	6	109	109	7	238
	Delta *Phragmites*	0-50	59	3,015	3,049	206	13,521
		50-500	57	1,818	1,920	424.9	13,130
Mississippi	Unvegetated	0-50	11	1,755	3,313	3	9,780
		50-500	26	67	189	0	772
Alabama	Unvegetated	0-50	8	124	218	0	640
		50-500	38	68	130	0	526
Florida	Unvegetated	0-50	45	100	201	0	896
		50-500	58	152	412	0	2,084

Post-Spill Coastal Wetland Soils

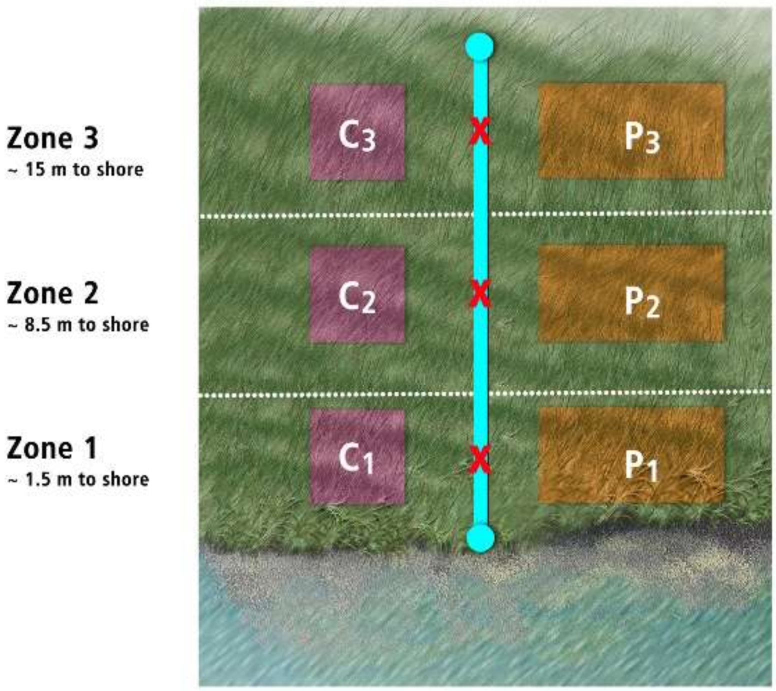
From 2010 to 2013, soil samples for TPAH50 analysis were collected in coastal wetlands across Louisiana, Mississippi, and Alabama (Figure 4.6-10).

Several wetland habitat types were sampled, including mainland marshes, mangrove-marsh complexes, back barrier marshes, Mississippi and Alabama island marshes, and Delta *Phragmites* marshes (Hester & Willis 2015b; Hester et al. 2015; Willis & Hester 2015a; Willis et al. 2015). At each site, a transect was extended perpendicular from the marsh shoreline edge into the marsh. Up to three zones were established for each transect, as shown in Figure 4.6-11 (J. Oehrig et al. 2015). Zone 1 is adjacent to the marsh edge, and zones 2 and 3 extend into the marsh. At each site, TPAH50 concentrations were analyzed by zone. Average TPAH50 soil concentrations in zone 1 for coastal wetland vegetation sites are presented in Table 4.6-7(Zhang et al. 2015b). TPAH50 soil concentrations increased with each oiling category from "no oil observed" to heavy persistent. For Louisiana mainland salt marshes, fall 2010 TPAH50 concentrations along oiled shorelines were orders of magnitude higher than ambient concentrations or those measured at "no oil observed" sites. In other Louisiana coastal wetland habitats and in Mississippi and Alabama, TPAH50 concentrations also tended to vary across shoreline oiling categories, and concentrations decreased over time (Zhang et al. 2015b).



Source: *Zhang et al. (2015b).*

Figure 4.6-10. Coastal wetland sampling locations classified by oil exposure categories. Soil TPAH50 analysis was conducted at these locations. See TPAH50 concentrations in Table 4.6-7.



__Source:__ J. Oehrig et al. (2015).

Figure 4.6-11. Example of an herbaceous salt marsh transect. Up to three zones were delineated for each transect, depending on the extent of oiling into the marsh. The center of zone 1 was 5 feet (1.5 meters) inland from the shoreline. The average centers of zones 2 and 3 were approximately 28 feet (8.5 meters) and 49 feet (15 meters) from the shoreline, respectively. Cover (C) and productivity (P) plots were established in each zone to sample plants and soil, and collect observational data.

Table 4.6-7. Soil TPAH50 concentrations in zone 1 of coastal wetland vegetation sites. Average concentrations and standard error (SE) are included in the table. Concentrations along Louisiana salt marsh oiled shorelines were orders of magnitude higher than concentrations measured at "no oil observed" sites (Zhang et al. 2015b).

			Soil TPAH50 Concentrations (ppb)									
			Fall 2010		Spring 2011		Fall 2011		Fall 2012		Fall 2013	
State	Habitat	Shoreline Exposure	Average	SE	Average	SE	Average	SE	Average	SE	Average	SE
Louisiana	Mainland Herbaceous Salt Marsh	HEAVIER PERSISTENT OILING	127,558	46,433	130,351	35,078	95,842	32,125	126,834	68,075	31,920	23,856
		HEAVIER OILING	12,398	6,684	4,875	3,050	2,552	1,255	520	74	260	24
		LIGHTER OILING	4,477	2,538	2,707	1,095	681	354	1,785	644	674	101
		"NO OIL OBSERVED"	394	94	556	162	527	172	711	295	918	388
	Back Barrier Herbaceous Salt Marsh	HEAVIER OILING	8,695	6,132	36,285	22,614	884	333	486	110	NA	NA
		LIGHTER OILING	43	24	26	6	37	22	NA	NA	NA	NA
		"NO OIL OBSERVED"	33	14	41	1	59	NA	15	NA	NA	NA
	Coastal Mangrove Marsh	HEAVIER PERSISTENT OILING	1,065	333	1,623	497	353	67	NA	NA	NA	NA
		HEAVIER OILING	966	455	658	313	555	169	682	490	841	652
		LIGHTER OILING	311	151	337	40	304	56	594	158	70	NA
		"NO OIL OBSERVED"	711	451	343	119	270	58	200	9	371	NA
	Delta *Phragmites* Marsh	HEAVIER PERSISTENT OILING	281	NA	896	NA	529	NA	48	NA	NA	NA
		HEAVIER OILING	1,128	520	1,233	604	377	169	913	NA	NA	NA
		LIGHTER OILING	1,350	507	1,229	813	1,223	346	1,415	437	NA	NA
		"NO OIL OBSERVED"	1,763	402	3,690	1,957	2,004	374	4,991	2,529	NA	NA
Mississippi/ Alabama - Mississippi Sound	Mainland Herbaceous Salt Marsh	HEAVIER OILING	NA	NA	362	NA	89	NA	851	NA	NA	NA
		LIGHTER OILING	NA	NA	283	NA	408	NA	161	NA	212	NA
		"NO OIL OBSERVED"	NA	NA	202	70	277	88	258	111	57	15
	Island Herbaceous Salt Marsh	HEAVIER PERSISTENT OILING	NA	NA	446	NA	415	NA	730	NA	782	NA
		HEAVIER OILING	NA	NA	11	NA	4	NA	8	NA	1	NA
		"NO OIL OBSERVED"	NA	NA	71	20	130	42	NA	NA	NA	NA

[1] Weighted average soil TPAH50 concentrations are weighted to account for stratified random sampling and preferential analysis of samples indicating likely presence of oil.

Post-Spill Sediment

Surface sediment samples were collected in the nearshore environment in 2010 and 2011 adjacent to both vegetated and unvegetated shorelines.

Coastal Wetlands

For sediment samples collected offshore of mainland salt marshes in Louisiana, sediment TPAH50 concentrations were generally higher along oiled shorelines when compared to shorelines where "no oil was observed" or ambient sediments (Table 4.6-8) (Zhang et al. 2015a). This pattern was especially strong for sediments closer to shore (0-50 meters). In 2011, concentrations in the heavier persistent oiling and heavier oiling categories were two to three times higher than ambient concentrations. The other states sampled did not display a pattern of greater TPAH50 sediment concentrations along oiled shorelines (Zhang et al. 2015a).

Table 4.6-8. 2011 weighted[1] sediment TPAH50 concentrations offshore of mainland salt marshes in Louisiana. For samples collected offshore of mainland salt marshes in Louisiana, sediment TPAH50 concentrations were generally higher along oiled shorelines compared to shorelines where "no oil was observed" or ambient sediments (Zhang et al. 2015a).

Shoreline Exposure	Distance to Shore (m)	Count	Sediment TPAH50 concentrations (ppb)			
			Average	Standard Error	Min	Max
HEAVIER PERSISTENT OILING	0-50	97	1,143	576	20	81,862
	50-500	30	261	21	29	574
HEAVIER OILING	0-50	71	907	179	6	26,900
	50-500	13	109	21	31	828
LIGHTER OILING	0-50	68	268	7	8	3,718
	50-500	18	179	33	28	698
"NO OIL OBSERVED"	0-50	54	401	90	26	10,576
	50-500	14	317	9	78	453

[1] Concentrations are weighted to account for preferential analysis of samples indicating likely presence of oil.

Unvegetated Shorelines

In 2010 and 2011, sediment samples were also collected adjacent to unvegetated shorelines, primarily beaches, in Florida, Alabama, Mississippi, and Louisiana. No relationship was detected between TPAH50 concentrations and degree of shoreline oiling (Zhang et al. 2015a). Average TPAH50 sediment concentrations are shown in Table 4.6-9 and Table 4.6-10.

Table 4.6-9. Summary of post-spill 2010 nearshore (within 500 meters) TPAH50 sediment concentrations along unvegetated shorelines. No data are shown for the “no oil observed” category in Louisiana, because only one such sample was collected and the concentration was below the detection limit. No relationship was detected between concentrations and degree of oiling (Zhang et al. 2015a).

State	Shoreline Oiling Exposure	Distance to Shore (m)	Sample Size	Sediment TPAH50 Concentrations (ppb)			
				Average	Standard Deviation	Min	Max
Louisiana	HEAVIER OILING	0-50	9	3,284	9,725	0	29,217
		50-500	16	280	389	0	1,573
	LIGHTER OILING	0-50	17	4,802	7,789	0	27,565
		50-500	12	665	690	0	1,664
Mississippi/ Alabama/ Florida	HEAVIER OILING	0-50	18	41	82	0	296
		50-500	37	77	196	0	832
	LIGHTER OILING	0-50	42	115	437	0	2,250
		50-500	38	63	200	0	1,107
	"NO OIL OBSERVED"	0-50	8	23	52	1	151
		50-500	11	75	131	2	434

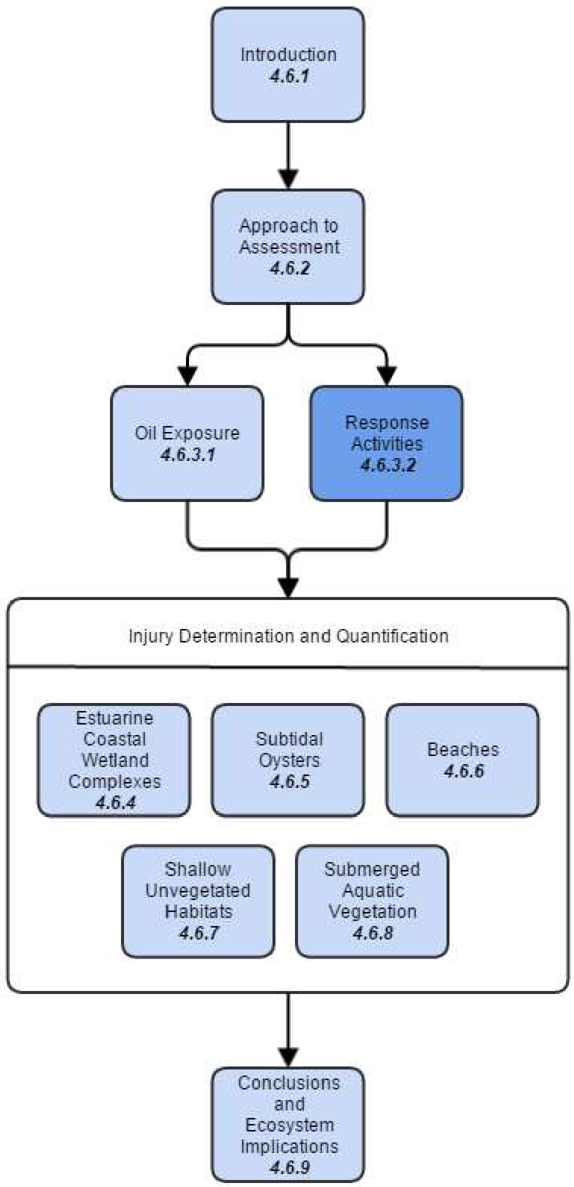
Table 4.6-10. Summary statistics of 2011 nearshore (within 500 meters) sediment TPAH50 concentrations along unvegetated shorelines. No relationship was observed between concentrations and degree of oiling (Zhang et al. 2015a).

State	Shoreline Oiling Exposure	Distance to Shore (m)	Sample Size	Sediment TPAH50 Concentrations (ppb)			
				Weighted Mean	Weighted Standard Error	Min	Max
Louisiana	HEAVIER PERSISTENT OILING	0-50	15	108	22	0.5	2,186
		50-500	2	1.1	0.1	1.1	1.2
	HEAVIER OILING	0-50	36	85	8	0.2	936
		50-500	29	90	13	1.5	823
	LIGHTER OILING	0-50	30	642	258	0.2	15,646
		50-500	23	1,940	520	0.4	14,068
	"NO OIL OBSERVED"	0-50	24	254	39	29	1,506
		50-500	5	395	223	84	1,656
Mississippi/ Alabama/ Florida	HEAVIER PERSISTENT OILING	0-50	1	0.9	-	0.9	0.9
		50-500	3	9	4	0.4	66
	HEAVIER OILING	0-50	79	59	33	0	11,830
		50-500	34	208	100	0	21,332
	LIGHTER OILING	0-50	63	8	2	0	156
		50-500	31	100	21	0	2,084
	"NO OIL OBSERVED"	0-50	56	105	6	0.1	1,738
		50-500	26	14	5	0.01	277

Sediment samples were also collected in seagrass beds surrounding the Chandeleur Islands before (May to July 2010) and after (August and September 2010) oil reached them. TPAH50 sediment concentrations were 8 to 12 times greater, on average, than baseline, pre-spill conditions (Cosentino-Manning et al. 2015). TPAH50 concentrations in nearshore animal tissue were highly variable and were not correlated to shoreline oiling; however, sample size was very limited (Jacob Oehrig et al. 2015).

Surface Water Oiling

The Trustees' assessment of water chemistry demonstrates that *Deepwater Horizon* oil was present as floating oil in nearshore/estuarine waters (Zhang et al. 2015b). The Trustees evaluated water chemistry data in the areas where oil was floating in nearshore/estuarine waters, including Terrebonne, Barataria, and Mobile Bays, and Chandeleur and Mississippi Sounds. Injury from these exposures is discussed in Section 4.4 (Water Column).



4.6.3.2 Response Activities

Following the *Deepwater Horizon* oil spill, a large response effort was initiated to minimize adverse effects to the region. These efforts included physical response actions and river water releases.

4.6.3.2.1 Physical Response Actions

Physical response actions relevant to the nearshore environment included extensive mechanical and manual removal of oil from beaches and other shorelines, including marshes, placement of boom to collect or deflect oil, and the construction of berms in Louisiana and Alabama to intercept oil. The placement of boom and construction of berms are described in Chapter 2 (Incident Description). Due to improper placement, equipment failure, and wave action, many of these booms became stranded during storms in early July 2010 on shorelines throughout Louisiana, Mississippi, Alabama, and Florida; most strandings occurred in sensitive salt marshes and mangrove habitats in Louisiana (Michel & Nixon 2015).

In marshes, onsite response activities included: placing booms adjacent to shorelines to prevent oil from reaching shorelines; flushing marsh surfaces with water; cutting and raking marsh vegetation; removing wrack and vegetation; raking heavy oil deposits from soil surfaces; and placing loose sorbent material

(Zengel et al. 2015a). Most onsite response activities involved landing boats on the marsh edge and deploying crews at the sites. Such activities could result in trampling, smothering, and burying of oyster habitat, as described in Section 4.6.4.5.9 (Nearshore Oysters).

The nature and extent of the beach cleanup effort was unprecedented in the U.S. history of oil spills (Michel et al. 2014). It extended across hundreds of miles of sand beach shoreline and required multiple years to complete (Michel et al. 2013). The extensiveness and invasiveness of the effort was largely reflective of the complex nature of the oiling of sand beaches. The distribution of oil was complex because oil stranding and re-oiling of sand beaches from submerged oil mats (see Section 4.2, Natural Resource Exposure) occurred in discontinuous waves over a period of months and in many different environmental conditions (e.g., wide tidal ranges, storms, and hurricanes) (Michel et al. 2015). Consequently, cleanup operations employed numerous different types of manual and mechanical treatments to remove the oil. These treatments ranged from manual techniques involving crews of workers digging out oil with hand-held tools to the use of large excavators and sand-sifting equipment during cleanup projects in late 2010. Figure 4.6-12 shows some of the machinery used during the response efforts.

Source: *NOAA Deepwater Horizon SCAT Program*

Figure 4.6-12. Top photo: "Big Dig" sifting operations on Pensacola Beach, Florida, on December 17, 2010. Bottom photo: "Power Screen Chieftain" sifting piles of oiled sand removed by mechanical scraping on Grand Terre II on October 30, 2010.

Specifically, the types of response activities that occurred on sand beaches included:

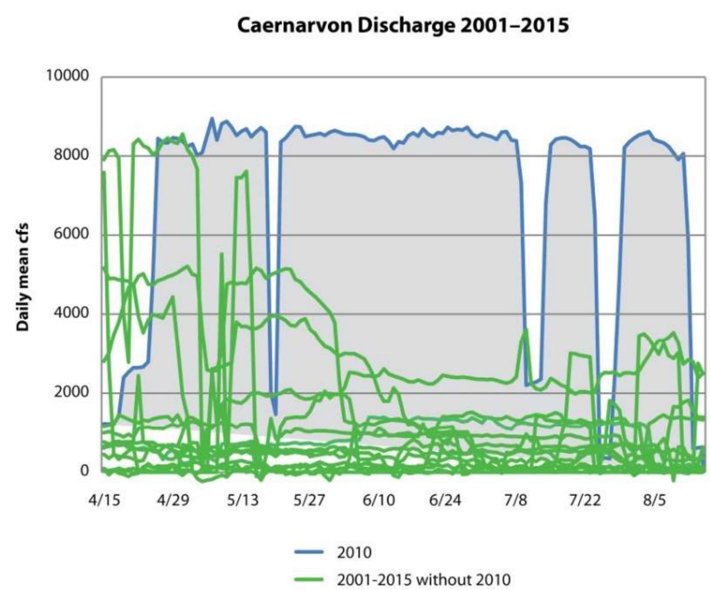
- **Manual treatment by response crews** using hand tools and vehicles to transport crews and wastes.
- **Augering** to search for buried oil.
- **Sifting** to separate oil from sand and remove it.
- **Tilling** to break up oil and expose it to air, with the expectation that this would accelerate biodegradation.
- **Excavating and dredging** to remove large volumes of oiled sediments for sifting or disposal.

Sand beaches were adversely affected by both (1) the direct exposure of oil to the habitat and natural resources utilizing the habitat and (2) collateral injuries associated with different response activities; and some of the response activities were very intense and conducted long after beaches had started to recover from oil exposure (Michel et al. 2015). The Trustees have consequently assessed both pathways and types of injury, as described in Section 4.6.6 (Beach Assessment).

4.6.3.2.2 River Water Releases

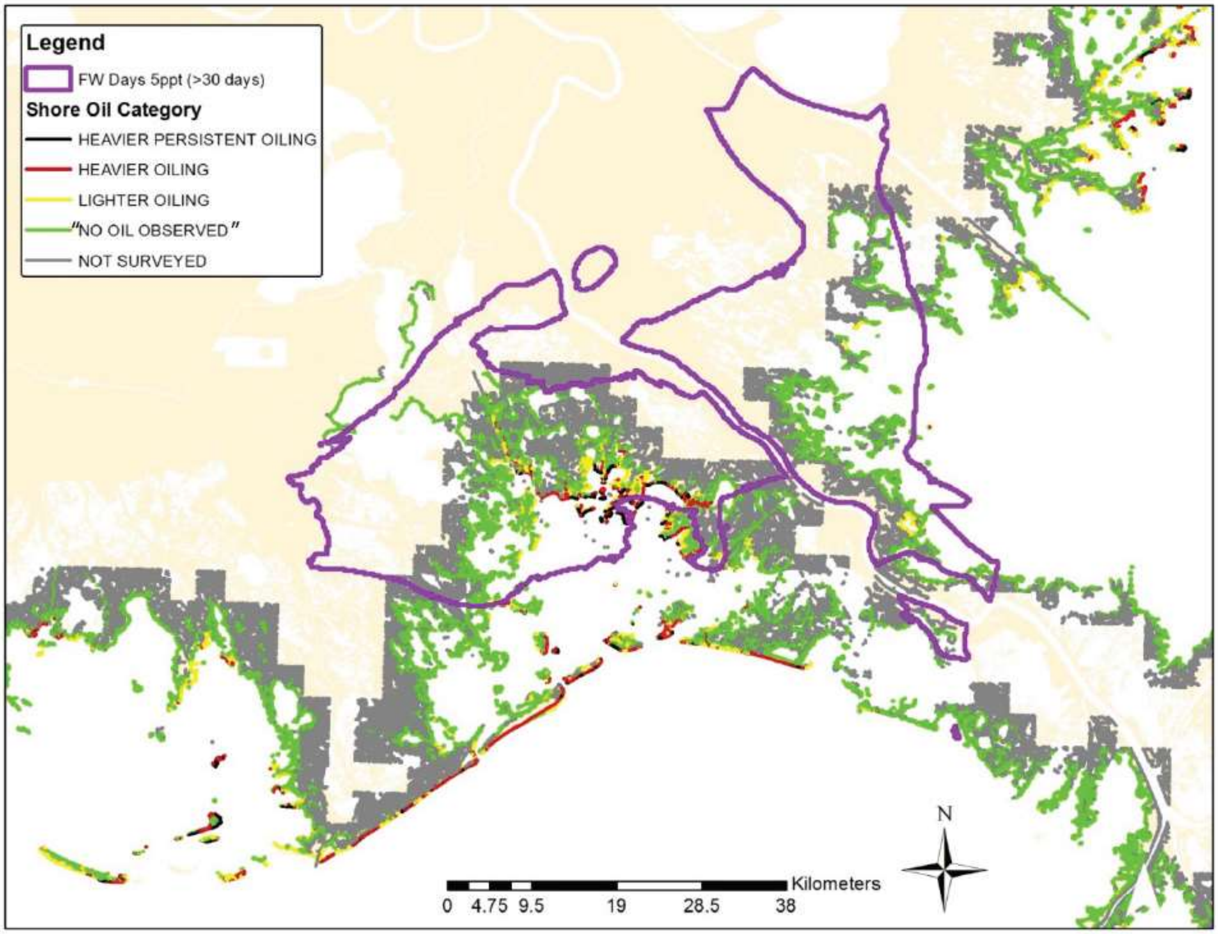
With oil approaching the shoreline, salinity control structures at nine separate locations in Louisiana (Davis Pond, Caernarvon, Bayou Lamoque, West Pointe a la Hache, Violet Siphon, White Ditch, Naomi Siphon, Ostrica Lock, and Bohemia) were opened as part of a series of response actions intended to reduce the movement of oil into sensitive marsh and shoreline areas.

The largest two of these structures allowed river water to flow into Barataria Bay and Black Bay/Breton Sound. The Caernarvon structure was opened on April 23, 2010, and remained open through the first two weeks of August at or near maximum capacity (approximately 8,000 cubic feet per second) (see Figure 4.6-13 for Caernarvon flow history) (Rouhani & Oehrig 2015b). Davis Pond remained open from May 8 through September 10, 2010, with flow ranging from 7,000 to 10,000 cubic feet per second. As demonstrated by the historical flow rates for Caernavon (depicted in green in Figure 4.6-13), these salinity control structures are typically opened during specific times of the year, for limited durations, and with controlled flow rates intended to make targeted changes to salinity levels in the state's coastal waters. In contrast, when used as a *Deepwater Horizon* oil spill response action (depicted in blue in Figure 4.6-13), these structures were opened at or near maximum capacity for extended periods of time to repel the approaching MC252 oil. By the time the MC252 well was shut down and the salinity control structures were closed in late 2010, the salinity levels in Louisiana coastal areas were significantly reduced (Rouhani & Oehrig 2015b). Figure 4.6-14 illustrates the geographic extent of areas impacted by river water releases conducted as part of response actions. These areas experienced an increase in the number of consecutive days where salinity levels were less than 5 parts per thousand when compared to historical baseline years (2006-2009) (Rouhani & Oehrig 2015b).



Source: *Powers et al. (2015a).*

Figure 4.6-13. Caernarvon discharge releases between 2001 and 2015. The discharge rate in 2010 to protect shorelines from oiling were significantly higher than discharge rates in the other years shown.



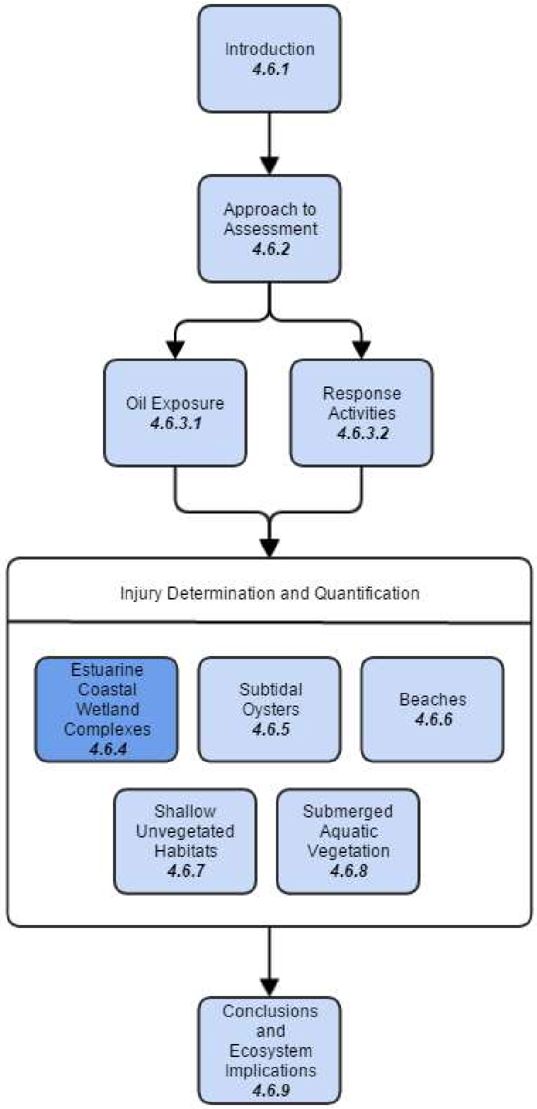
Source: *Rouhani and Oehrig (2015b).*

Figure 4.6-14. Spatial extent of impacts of the summer river water releases in response to the approaching *Deepwater Horizon* oil. Salinity levels in the areas outlined in purple were much lower for much longer than in a typical year.

4.6.4 Estuarine Coastal Wetlands Complex Injury Assessment

Key Points

- The Trustees conducted an extensive and multifaceted assessment of injury to coastal wetlands and associated fauna.
- Injury to the coastal wetlands was observed across wide swaths of the northern Gulf of Mexico and to representative faunal species living in or utilizing coastal wetland habitat.
- Injury occurred in all oiling exposure categories, with more severe and broader injuries documented along more heavily oiled shorelines.
- Cleanup activities such as raking and boom placement impacted marsh fauna and coastal wetland habitat.
- Restoration design should ensure that adequate depth and length are incorporated to develop marsh habitats that will be sustainable and provide the benefits of the land/water interface where many species thrive.



Coastal wetland shorelines along the northern Gulf of Mexico were highly vulnerable to the *Deepwater Horizon* oil spill due to their location and sensitivity to oil. This habitat and associated fauna's vulnerability to oiling effects are heightened by the ongoing influence of non-oil-related stressors (Michel & Nixon 2015). This section describes the results of the Trustees' assessment of injury to coastal wetlands and associated fauna (including nearshore oysters that fringe salt marshes) resulting from the *Deepwater Horizon* incident. The section presents the known effects of oil on coastal wetlands (Section 4.6.4.1), , an overview of the Trustees' approach to assessing the effects of the *Deepwater Horizon* incident on coastal wetlands (Section 4.6.4.2), and a summary of this assessment's findings (Section 4.6.4.3). The section also explains the injury characterized for each evaluated component of coastal wetlands, including plants (Section 4.6.4.4), fauna (Section 4.6.4.5), shoreline erosion (Section 4.6.4.6), and injury from response actions (Section 4.6.4.7).

4.6.4.1 Effects of Oil on Coastal Wetland Habitat

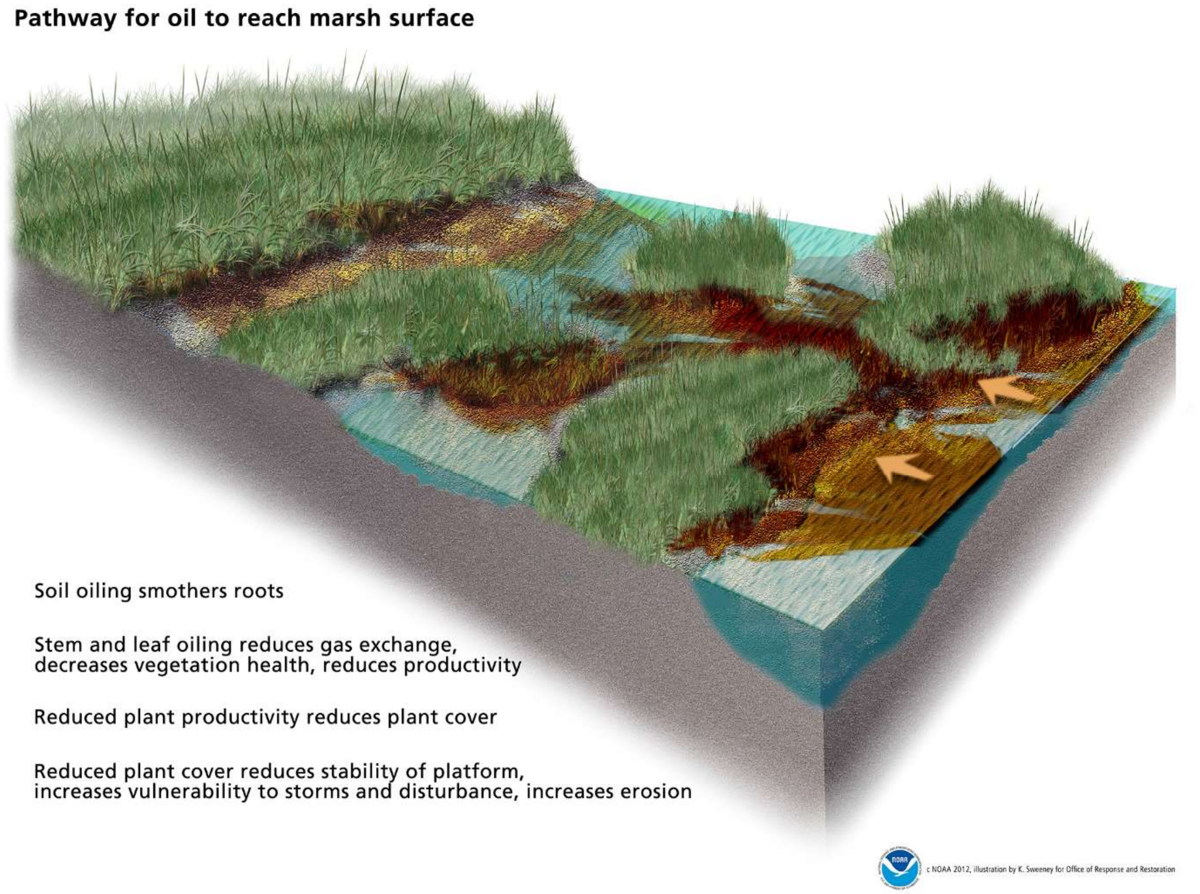
Oiling has been documented to adversely affect coastal wetland vegetation and associated fauna (Mishra et al. 2012; Powers & Scyphers 2015). Oil can wash up at the marsh edge, oiling soil and coating vegetation (Figure 4.6-15). It can also penetrate the marsh through tidal creeks and wash-over events, and become stranded in the marsh interior where it can coat plant stems and soil. Through these basic pathways, oil can directly foul plants and animals, causing physical harm through smothering or toxicity through dermal contact or ingestion. These mechanisms can cause mortality or sublethal effects, such as impairment of reproduction, reduced growth, or reduced ability to avoid predators. Actions taken in response to oil spills (e.g., placement of boom) can also negatively impact coastal wetlands (Martınez et al. 2012). Response actions and plant death can also accelerate erosion of this important habitat type.

A major concern for Gulf of Mexico wetlands is whether sustainability and resistance to future disturbances have been reduced due to the *Deepwater Horizon* spill. These wetlands, particularly those in the Mississippi River Delta, remain viable only if their rate of elevation change (typically from additional silt deposits) keeps pace with the rate of relative sea level rise (from land subsidence and global sea level rise) Day et al. (2011). Many factors can exacerbate wetland loss and erosion. These include physical or chemical disturbance that results in the trampling or death of plants along the marsh edge, changes in soil strength and stability, and smothering or crushing of nearshore oyster cover (DeLaune et al. 1984; Peterson et al. 2003b).

4.6.4.2 Approach to the Assessment

The Trustees' used a multifaceted approach to evaluate coastal wetland impacts as a result of the *Deepwater Horizon* incident. Vegetative community studies were conducted in the dominant wetland habitat types and regions that were most exposed to oiling. These habitat types and regions included: salt marshes along the coasts of Louisiana, Mississippi, and Alabama; Louisiana back barrier islands; and Mississippi and Alabama islands; mangrove-marsh complexes in Louisiana; and *Phragmites* marshes on the Mississippi River Delta. (Section 4.6.1, Introduction, describes these habitat types.) The studies were designed to evaluate effects of oiled marsh compared to reference locations. Surveys were conducted over a period of 4 years to evaluate the long-term effects from oiling and to characterize any recovery.

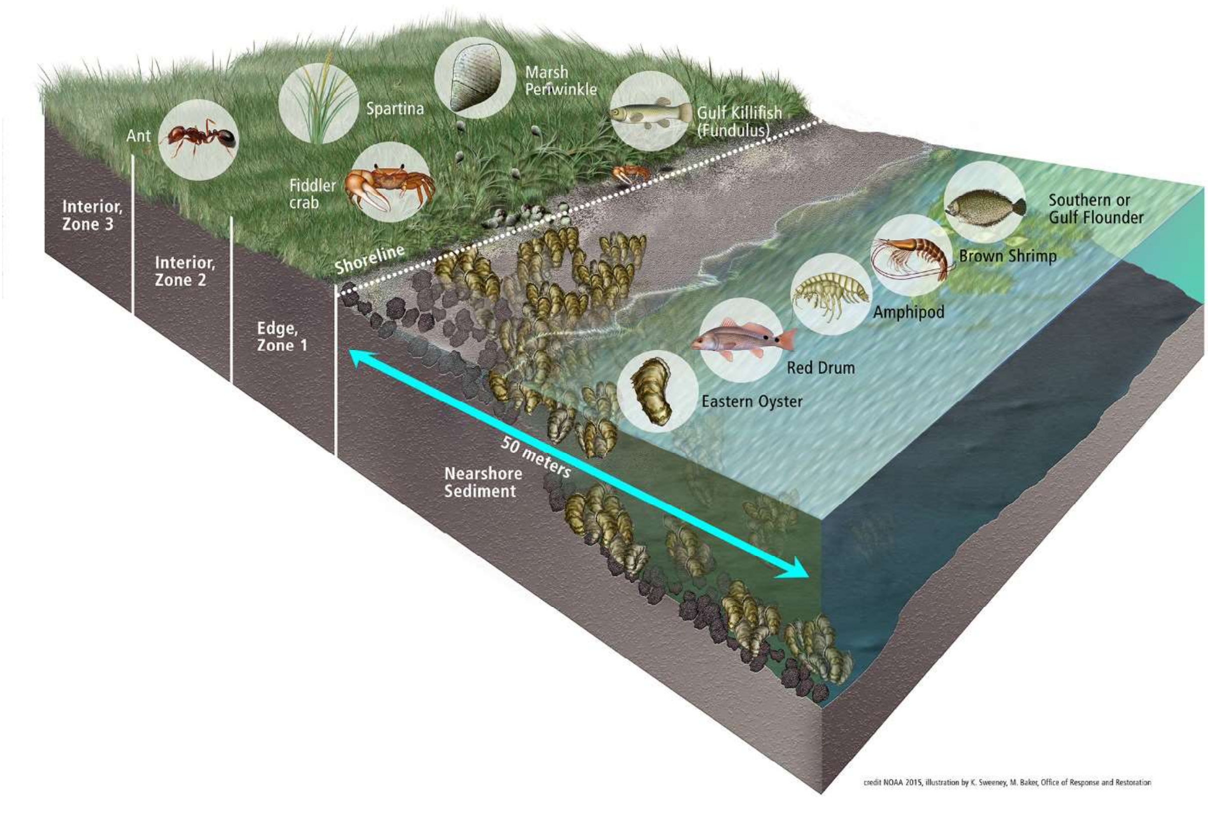
Trustees also studied the effects of *Deepwater Horizon* oiling on marsh fauna. Several taxa were selected to represent injury to the marsh faunal community more broadly. Thus, total faunal losses are expected to be much higher compared to the sum of losses of these representative species. Trustees selected these taxa based on their importance to the ecosystem, their representativeness of exposure pathways, their prevalence, their sensitivity to oil, or some combination of these factors. The studies integrated results from the Trustees' Toxicity Testing program (see Section 4.3, Toxicity) and were designed to capture impairment (sublethal effects), mortality (lethal effects), and effects that may occur on different timescales.



Source: *Kate Sweeney for NOAA.*

Figure 4.6-15. Diagram of coastal wetland oiling. Oil can wash up at the marsh edge, oiling soil and coating vegetation. It can also penetrate the marsh through tidal creeks and wash-over events, and become stranded in the marsh interior where it can smother plant roots. Stem and leaf oiling can reduce gas exchange, decreasing vegetation health and productivity. Reduced productivity can in turn lead to reduced plant cover and root structure, which can reduce the stability of the marsh platform, increasing the marsh's vulnerability to storms and disturbance and increasing erosion.

Laboratory and field studies evaluated marsh periwinkles (*Littoraria irrorata*), brown and white shrimp (*Farfantepenaeus aztecus* and *Litopenaeus setiferus*), southern flounder (*Paralichthys lethostigma*), red drum (*Sciaenops ocellatus*), killifish (*Fundulus* spp.), amphipods (*Leptocheirus plumulosus*), fiddler crabs (*Uca longisignalis*), insects, and nearshore oysters (*Crassostrea virginica*) that fringe the marsh edge (Figure 4.6-16). Evaluation of impacts relied on NRDA field studies, non-NRDA field studies, and laboratory toxicity studies. For fiddler crabs and marsh insects, other researchers' studies on the effects of *Deepwater Horizon* oil are also summarized.



***Source:** Kate Sweeney for NOAA.*

Figure 4.6-16. Schematic diagram of a generalized, tidally influenced salt marsh and nearshore environment illustrating the components studied to represent injury to the nearshore ecosystem. Coastal wetland vegetation impacts were studied at various distances from the wetland edge (illustrated here as zones 1, 2, and 3). All three zones are regularly inundated by tides.

The Trustees also studied the effects of the *Deepwater Horizon* incident on coastal wetland erosion. These studies analyzed aerial imagery collected at intervals following the spill and field measurements of shoreline change over time. Impacts from response activities, such as placement of boom in marshes, were also evaluated.

4.6.4.3 Brief Summary of Impacts

Injury to the coastal wetlands was observed across wide swaths of the northern Gulf of Mexico. Injury occurred in all oiling exposure categories, with more severe and varied injuries documented along more heavily oiled shorelines. Multiple model species were affected, including mainland salt marsh plants (reduced plant cover and aboveground biomass), periwinkles (reduced abundance), shrimp (reduced growth and biomass), amphipods (reduced survival and biomass), *Fundulus* spp. (reduced hatch success and biomass), juvenile southern flounder (reduced growth and biomass), red drum (reduced growth and biomass), fiddler crab (reduced burrow density), insects (reduced abundance), and nearshore oysters (reduced cover and biomass). Marsh edge habitat also suffered increased erosion.

Animals using the edge of the marsh for refuge and forage were exposed to oil through contact with oiled plants, soil, sediment, and detritus on the marsh surface as it floods with the tide, as well as

through ingestion or contact with oil entrained in submerged sediments near the edge. Toxicity testing conducted using marsh soil containing MC252 oil demonstrates that PAH concentrations found in oiled marsh areas are toxic to many marsh species (Morris et al. 2015). Cleanup and oil removal activities at the edge of marshes smothered, crushed, or removed animals and vegetation in oiled areas (Michel & Nixon 2015). The release of river water as part of spill response actions also reduced growth of juvenile brown shrimp (Powers & Scyphers 2015).

Injuries to evaluated components of coastal wetlands are described in more detail below.

4.6.4.4 Vegetation

Key Points

- Adverse effects to coastal wetland vegetation were observed across numerous habitat types and regions.
- In Louisiana salt marshes, reductions in live aboveground biomass and live plant cover ranged from 11 to 53 percent compared to reference over a total 350 to 721 miles (563-1,161 kilometers). Recovery time estimates range from within 2 years after the spill for lighter oiling categories to 8 years after the spill for heavier oiling categories.
- Mainland salt marsh vegetation in Mississippi and Alabama was adversely affected by the oil spill based on reductions in live aboveground biomass.
- Louisiana mangrove-marsh complexes sustained oil-related impacts based on multiple indicators of the reduced vegetative extent of mangroves with plant oiling.
- Louisiana delta *Phragmites australis* marshes showed detectable effects due to increases in dead cover and dead aboveground biomass along the marsh edge.
- Impacts of plant oiling were not detected in Louisiana back barrier islands. Small sample size limited the ability to detect change.

Coastal wetlands serve as a key base of the productive Gulf of Mexico aquatic and terrestrial food webs, supporting animals that use the marsh surface (e.g., shrimp, snails, fish, crabs, and insects) and animals that reside adjacent to the marsh (e.g., nearshore oysters) (Peterson & Howarth 1987). The composition of the vegetative community can vary according to region and hydrogeomorphic setting, including characteristics such as salinity and tidal inundation (Sasser et al. 2014). For the purposes of this assessment, Trustees focused on several broad categories of coastal wetland types: salt marsh dominated by smooth cordgrass found on the mainland and back barrier islands in Louisiana, Mississippi and Alabama; woody black mangrove-salt marsh complexes located on the Louisiana mainland and back barrier islands; and *Phragmites* marshes on the Mississippi River Delta in Louisiana (see Section 4.6.1, Introduction, for more detail).

4.6.4.4.1 Injury Determination

Coastal wetland vegetation can be impacted through many different physical and chemical mechanisms (Baker 1970; Mendelssohn et al. 2012; Pezeshki et al. 2000). Oil contamination can affect vegetation productivity in numerous ways, ranging from reduced photosynthesis, stem density, and biomass to complete mortality (Alexander & Webb Jr. 1987; Anderson & Hess 2012; Day et al. 2013; DeLaune et al. 1979; Dowty et al. 2001; Hester et al. 2015; Lin & Mendelssohn 1996; Lin et al. 1999; Lin et al. 2002; Mendelssohn et al. 1990; Pezeshki et al. 2000). The specific mechanisms and severity of injury to vegetation depend on many factors, including oil type, degree of weathering of the oil, spill volume, seasonality of exposure, soil type and exposure, and coverage of aboveground tissues (Baker 1970; Biber et al. 2014; Lin & Mendelssohn 2012; Mishra et al. 2012; Pezeshki et al. 2000). One of the most significant factors determining the type and severity of injury is whether the oil predominantly coats the aboveground tissues or the soil (Pezeshki et al. 2000).

Mangrove habitats are particularly susceptible to the stranding of oil and are considered the most sensitive of all coastal ecosystems to oil spills[2] (Hayes & Gundlach 1979). In fact, it has been speculated that mangroves may take decades to recover from spills (Odum & Johannes 1975). The effects of oil on mangroves are thought to be dependent on various factors controlling oil persistence, such as oil type, the elapsed time between a spill and the stranding of oil, winds, currents, and tides (Getter et al. 1981; Snedaker et al. 1996). Oil can also directly and adversely impact coastal wetland soils, which in turn negatively affects marsh flora and fauna. It can decrease soil accretion rates and soil strength, making wetlands more vulnerable to erosion and land loss (Culbertson et al. 2008; Day et al. 2011; Day et al. 2013; Habib-ur-Rehman et al. 2007; Rahman et al. 2010; Shah et al. 2003). Soil oiling can also limit gas exchange, which, in combination with microbial degradation of the oil, can decrease oxygen concentrations and increase sulfide concentrations in pore water (Judy 2013; Natter et al. 2012; Nyman 1999; Nyman & McGinnis III 1999; Pezeshki et al. 2000). High sulfide concentrations can be toxic to wetland plants and animals (Bradley & Dunn 1989; Koch & Mendelssohn 1989; Koch et al. 1990).

Overview of Vegetation Injury

Oiling from the *Deepwater Horizon* incident caused significant and widespread injury to coastal wetlands. In many heavily oiled areas, MC252 oil coated marsh grass, resulting in death and widespread reductions in plant productivity and health (Hester & Willis 2015d; Hester et al. 2015; Willis & Hester 2015a; Willis et al. 2015). Effects were most notably observed in the mainland salt marshes of Louisiana, where they persisted for all 4 years of study. However, effects were also evident in other habitat types and regions (Hester & Willis 2015d; Hester et al. 2015; Willis & Hester 2015a; Willis et al. 2015).

Trustees assessed the impacts on wetland vegetation using numerous measures of plant productivity, plant health, and measures of soil processes. These measures include, but are not limited to, live peak standing crop (as measured by aboveground biomass), percent vegetative cover, and visual observations

[2] Literature examining effects of oil on *Avicennia germinans* is somewhat sparse; therefore, this summary includes both documented oiling effects on *Avicennia germinans* and documented effects on other mangrove species, most notably the red mangrove *(Rhizophora mangle)*.

of plant stress (i.e., chlorosis). Numerous measures consistently demonstrated adverse effects associated with oiling.

The Trustees surveyed coastal wetlands of Louisiana, Mississippi, and Alabama for 4 years following the spill. In total, approximately 200 sites were repeatedly sampled across the range of coastal wetland communities in these states (Figure 4.6-10). Measures of plant productivity and health were collected at up to three distances from shore. Zone 1 was centered at 5 feet (1.5 meters) from the shoreline and represented the marsh edge. In oiled areas of the Louisiana mainland herbaceous habitat sites, the center points for zones 2 and 3 were located on average 26 feet (8 meters) and 46 feet (14 meters) from the shoreline (Figure 4.6-11).

Sampling locations represented the full spectrum of oiling conditions. Because adverse effects of oiling are believed to be largely driven by the vertical extent of plant stem oiling (Pezeshki et al. 2000), plant stem oiling was used to characterize the degree of oiling at each site. Sites were categorized into one of the following plant oiling classes:

- No oiling.
- Trace to ≤10 percent vertical oil coverage of the vegetation.
- 11 to ≤50 percent vertical oil coverage of the vegetation.
- 51 to ≤ 90 percent vertical oil coverage of the vegetation.
- 91 to ≤ 100 percent vertical oil coverage of the vegetation.

Plant oiling was measured based on visual observations in a pre-assessment survey conducted in the months following the spill. The plant oiling designations do not necessarily represent the maximum extent of oiling at any given site, because re-oiling occurred in some locations after the pre-assessment survey was conducted. Data from this survey were used to select sites for the coastal wetland vegetation injury study using a stratified random design.

Vegetation Injury by Habitat Type/Region

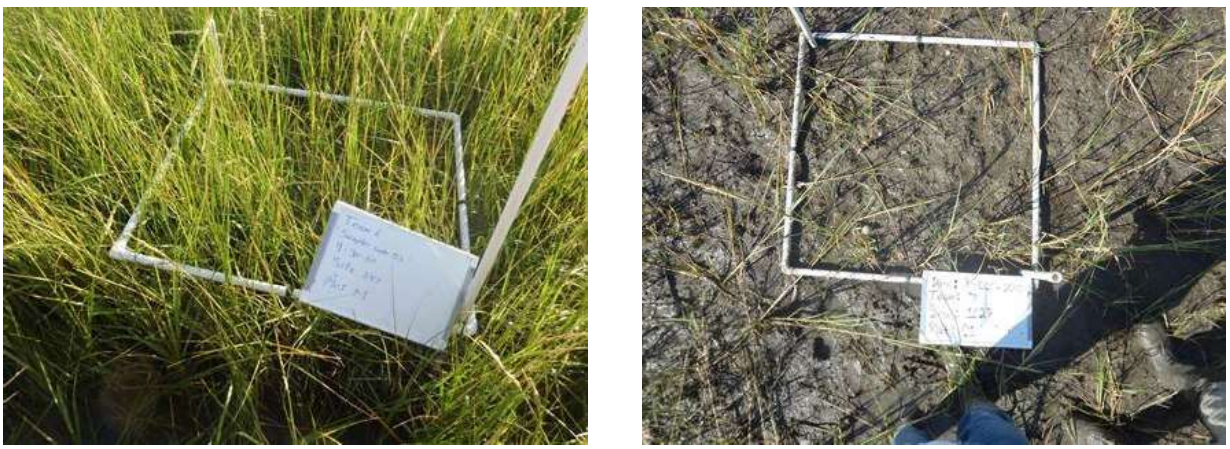
Numerous habitat types and regions suffered adverse effects to coastal wetland vegetation. Key measures of the health and productivity of the vegetative community included live aboveground biomass, vegetation cover, and the degree of chlorosis.

In Louisiana mainland herbaceous salt marshes, live aboveground biomass was reduced between 11 and 53 percent at oiled sites compared to reference sites (see Figure 4.6-17 for image of oiling effects) (Hester et al. 2015). Plant oiling reduced *S. alterniflora* live cover in zone 1 by 45 percent in fall 2010 (Hester et al. 2015). Sites exposed to trace or greater vertical oiling of plant stems suffered reduced live plant cover and live above ground biomass for the majority of the 4-year NRDA study. The highest plant oiling class suffered the most significant impacts, but all degrees of oiling showed adverse effects. Impacts were greatest along the marsh edge (zone 1), though more interior areas (zones 2 and 3) that comprise much greater acreages were also adversely affected (Figure 4.6-18). Marsh sites with trace or greater vertical oiling in the first 2 years of the study also exhibited declines in plant health, as represented by elevated chlorosis—the yellowing of leaves from a lack of chlorophyll.

As the study progressed over 4 years, fewer impacts to health and productivity were detected. Many sites, however, eroded or partially eroded over the course of the study, decreasing sample size and limiting the ability to detect differences (Hester et al. 2015). Thus, the fewer significant effects detected at the end of the 4-year study should not be interpreted as evidence of the vegetative community's full recovery (Hester et al. 2015).

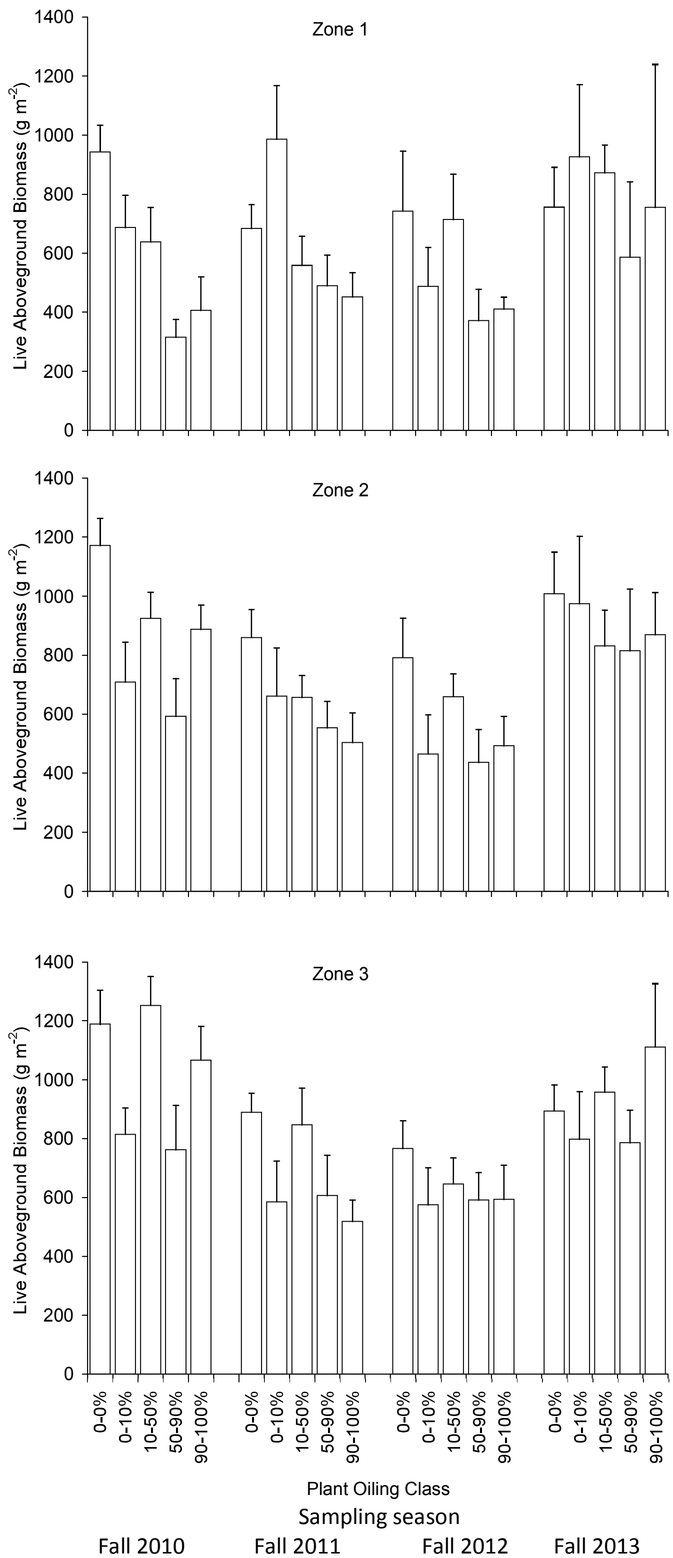
Studies of effects of *Deepwater Horizon* oiling on coastal wetlands were also conducted by researchers not involved in the NRDA. Results from these studies generally confirm the NRDA study findings, including reductions in aboveground biomass, plant cover, stem density rhizomes, and photosynthetic functioning in heavily oiled salt marshes (Biber et al. 2014; Lin & Mendelssohn 2012; RamanaRao et al. 2011; Silliman et al. 2012).

Similar trends of injury were observed in habitats and regions other than the Louisiana mainland salt marshes. However, the smaller sample size in these other locations limited the ability to detect statistically significant differences in productivity, health, and plant cover.



Source: *NOAA.*

Figure 4.6-17. Sampling sites in Louisiana mainland salt marshes in fall 2010. The photograph on the left shows abundant marsh grass at an unoiled site. The photograph on the right shows sparse vegetation at a heavily oiled site. Both sites were in the Timbalier Basin.



Source: *Hester et al. (2015).*

Figure 4.6-18. The effect of plant oiling class and zone within a sampling period on live aboveground biomass (mean +/- 1 standard error). Oiling reduced live aboveground biomass across oiling classes, zones, and years. Effects were generally greatest along the marsh edge (zone 1) and in the heavier oiling categories.

Mangroves

Mangrove-marsh complexes in Louisiana also sustained oil-related impacts (Willis & Hester 2015a). Because Louisiana mangrove habitats are comprised of a mixture of black mangroves (*Avicennia germinans*) and herbaceous halophytes, such as smooth cordgrass (*Spartina alterniflora*), reductions in the extent and health of any of these species results in the impairment of the functioning of these critical habitats (Hester et al. 2005). An important finding of the NRDA study is that significant reductions in the extent of both vegetative components of Louisiana mangrove habitat due to plant oiling were detected in the fall of 2011 (Willis & Hester 2015a). Further, the physiological health of mangroves, as indicated by visual estimation of chlorosis, was found to be diminished in the fall of 2011 due to plant oiling. This apparent delay in impacts reflects several factors. The initial (fall of 2010) field assessment for Louisiana mangrove habitat was conducted late in the growing season. Therefore, much of the herbaceous component of the mangrove habitat plots would be in a state of natural senescence. This circumstance made it difficult to distinguish healthy but dormant herbaceous vegetation from that injured by oiling (Willis & Hester 2015a).

Also, woody vegetation, such as mangroves, would not be expected to exhibit measurable reductions in growth only months after oil exposure (Snedaker et al. 1996). Oil spills are generally regarded as having long-term repercussions on mangrove habitats over an extended period of time, partially because of the reported persistence of petroleum hydrocarbons in the mangrove soils studied (Corredor et al. 1990; Snedaker et al. 1996; Willis & Hester 2015a).

Importantly, multiple indicators of injury to mangroves were significantly different between oiled and unoiled sites, indicating a consistent and real effect of oiling. Absolute and relative live mangrove cover diminished with increased plant oiling. Notably, it appears that mangrove growth rate (as measured by change in mangrove canopy extent) was reduced between 2010 and 2011 in areas exposed to greater than 10 percent plant oiling when compared to reference sites (Willis & Hester 2015a).

Although effects of oiling on mangrove vegetation were observed, the heterogeneity of mangrove habitats in the study area and the limited spatial extent of the study sites along the Chandeleur Islands limits the ability to expand these results over a larger area and to quantify mangrove vegetation losses.

Mississippi/Alabama Salt Marsh Vegetation

The oil spill also adversely affected mainland salt marsh vegetation in Mississippi and Alabama, although to a lesser degree than the Louisiana mainland salt marshes (Willis et al. 2015). Significant reductions in live aboveground biomass were detected in 2011 and 2012 in oiled Mississippi/Alabama salt marshes when compared to reference locations. Sampling of vegetation in marshes in Mississippi and Alabama did not begin until 2011. The variability of plant species composition in Mississippi marshes (some sites were dominated by black needlerush [*Juncus roemerianus*]) also may have influenced the ability to detect effects of oiling. Because of differences in the hydrologic regime and other habitat features, marshes in Perdido Bay, Alabama, were evaluated separately from other mainland marshes. The small datasets preclude the Trustees from expanding these results to a broader area and to quantify losses to vegetation in Mississippi and Alabama (Willis et al. 2015). However, independent analyses in the published literature are consistent with marsh injury in these states (Biber et al. 2014).

Back Barrier Island Salt Marsh Vegetation

These marshes are higher energy environments where oiling may be more transitory than in other, lower energy environments. NRDA studies did not reveal any oiling impact to marsh vegetation on Louisiana back barrier islands (Hester & Willis 2015b). However, only a few locations were sampled in this dynamic habitat type, due to the inability to obtain rights to sample in many areas. Oiled islands where birds were actively nesting, for example, were not sampled, even though they are areas of high ecological importance where assessment would otherwise be appropriate. Similarly, impacts of plant oiling were not detected on Mississippi and Alabama islands, where a small number of sites was also sampled (Willis & Hester 2015b). The absence of observed injuries to back barrier island salt marsh vegetation does not mean that oiled back barrier islands were not injured, since there is evidence of heavy oiling in many areas (for example, Cat Island; Figure 4.6-19).

Source: *P.J. Hahn, Pelican Coast Consulting, LLC.*

Figure 4.6-19. Cat Island sustained heavy oiling.

Delta *Phragmites* Vegetation

For the Delta *Phragmites australis* marshes, detectable effects were largely limited to increases in dead cover and dead aboveground biomass along the marsh edge (zone 1) in the fall of 2011 (Hester & Willis 2015d). Sampling in 2010 was initiated in late fall, when natural senescence had begun. This late-season sampling may have prevented detection of impacts in that year. Also limiting the ability to detect effects of oiling was the small sample size, due to land access restrictions and other logistical constraints. Consistent flooding in these areas, which is a function of river flow, may have prevented *Deepwater Horizon* oil from reaching soils and limited measurable effects to *Phragmites* vegetation. An explanation for low soil oiling, possibly due to flooding, is provided in an inundation analysis of sites (J. Oehrig et al. 2015). Because *Phragmites* plant stems are much taller than those of *Spartina*, a lower percentage of the total length of *Phragmites* stems were oiled when compared to *Spartina*, which may have provided some protection against oil toxicity. However, some areas of Delta *Phragmites* marsh received heavy oiling.

Confounding Factors

In observational studies, a critical aspect of the experimental design is to be able to determine the relationship between the variables of interest (e.g., plant oiling and plant health) and minimize the effects of possible confounding factors that have nothing to do with oil (e.g., soil quality and other naturally varying parameters) (Hester et al. 2015). Confounding factors were reduced by separating the potentially affected area into habitats and sub-regions to address known differences in soil type, grain size, soil stability, and dominant vegetation. Within each habitat and regional strata, no significant differences between plant oiling classes were detected for the key environmental setting variables measured in this study, including soil organic matter, soil bulk density, plot elevation, percentage of time flooded, average wave exposure index, and maximum wave exposure index (Nixon 2015). As a result, all plant oiling classes, including the reference category, are indistinguishable from each other regarding the likely confounding factors in this assessment: soil type, wave energy exposure, and hydrologic regime (as indicated by inundation duration) (as indicated by inundation duration; J. Oehrig et al. 2015). Further, multivariate analyses corroborate that reductions in metrics related to plant productivity (live plant cover and live aboveground biomass) were more closely related to metrics representative of oiling (plant oiling extent and soil TPAH50 concentration) than those representative of wave energy exposure (average and maximum exposure index) (Hester et al. 2015; Zhang et al. 2015a).

Injury Quantification

As described in Section 4.6.4.4.1 (Injury Determination), oiling caused many adverse effects to coastal wetlands, most prominently in the Louisiana mainland salt marshes. However, effects were also detected in other habitat types and regions. Table 4.6-11 demonstrates effects to mainland salt marshes in Louisiana in terms of the percent reduction in key vegetative metrics by plant oiling class compared to reference (0 percent plant oiling) sites (Zachary Nixon et al. 2015a). Table 4.6-11 also indicates the miles over which these impacts occurred. Dramatic effects in Louisiana mainland herbaceous marsh occurred over a total of 350 to 721 miles (563 to 1161 kilometers). Recovery time estimates range from 2 years after the spill for lighter oiling categories to 8 years after the spill for heavier oiling categories (Zachary Nixon et al. 2015a).

Figure 4.6-20 illustrates how injury to Louisiana mainland herbaceous marshes was quantified.

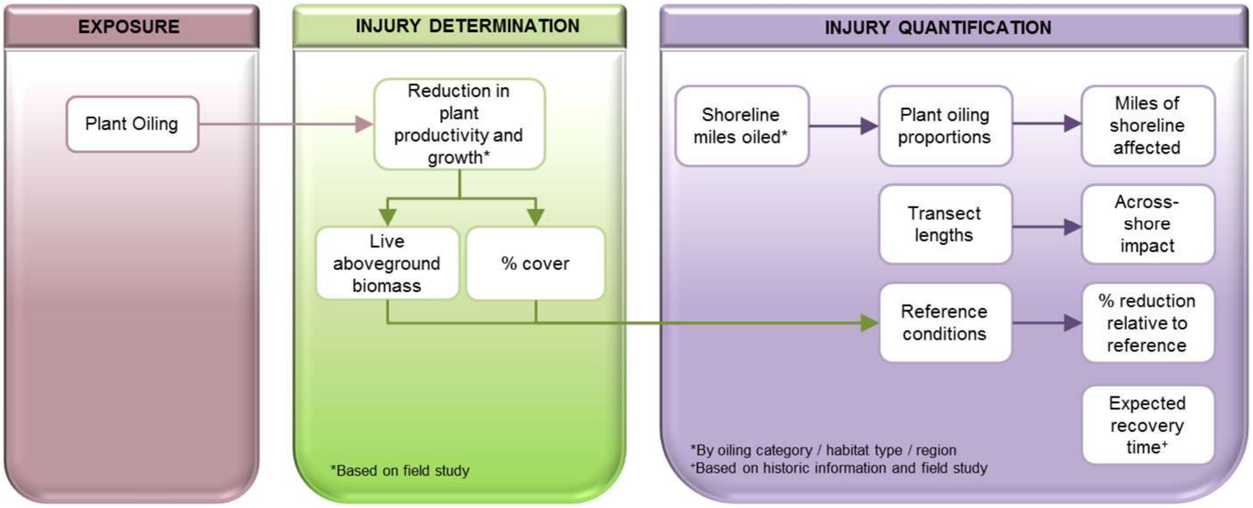


Figure 4.6-20. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to coastal wetland plants occurred. Plant aboveground biomass and percent cover at oiled sites were compared to reference sites to develop percent reductions relative to reference for each plant oiling class and habitat type. Oiled shoreline miles were converted from the shoreline oiling categories to plant oiling classes to determine the miles of shoreline affected. The length of field study transects indicated across-shore width of oil penetration. Expected recovery times were based on field data and literature.

Table 4.6-11. Percent reductions relative to reference for key coastal wetland vegetation metrics in Louisiana mainland salt marshes in 2010, the year of maximum impacts. Also included are the estimated time to recovery, the width of impact (perpendicular to shore), and the miles of shoreline affected (Goovaerts 2015; Zachary Nixon et al. 2015a).

Plant Oiling Category	Percent Reduction in Live Aboveground Biomass Relative to Reference	Percent Reduction in Live Cover Relative to Reference	Expected Recovery Time (Years)	Width of Impact Feet (m)	Weighted Lengths Based on Shoreline Oiling Classifications Miles (km)	Geographically-Weighted Lengths Based on Linear Regression Miles (km)
0-10% plant oiling	25.6	21.9	2	28.5 (8.7)	73 (117)	124 (199)
10-50% plant oiling	10.6	15.3	2	43.0 (13.1)	152 (245)	390 (628)
50-90% plant oiling	53.2	35.8	8	37.8 (11.5)	78 (125)	140 (225)
90-100% plant oiling	38.6	29.9	8	56.4 (17.1)	47 (76)	67(109)
TOTAL					350 (563)	721 (1,161)

4.6.4.5 Fauna

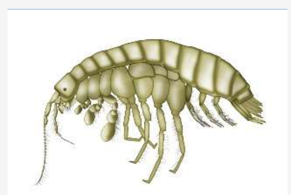
Marsh fauna are susceptible to the detrimental effects of oil, which can result in reduced growth, reproductive failure, and mortality and other adverse effects (e.g., McCall & Pennings 2012; Rozas et al. 2014). Biodiversity and population density of benthic organisms have been found to be significantly lower in oil-contaminated areas than in non-contaminated areas (Carman et al. 1997; DeLaune et al.

1984; Lindstedt 1978; Oberdörster et al. 1999). Oil pollution can alter recruitment and feeding patterns (Day et al. 2013). Benthic organisms (e.g., mussels, oysters, grass shrimp, and crabs) may accumulate higher concentrations of petroleum hydrocarbons in their tissues than nektonic species, presumably due to the fact that the detritus-based food web is readily contaminated (Lindstedt 1978). Periwinkle snails are susceptible to oiling impacts because they are closely associated with the marsh substrate and emergent vegetation, typically *S. alterniflora* (Hershner & Lake 1980; Hershner & Moore 1977; Krebs & Tanner 1981; Lee et al. 1981). After prior spills, fiddler crabs have sustained significant adverse effects years after exposure to oil, due to direct toxicity, smothering, and limited access to the marsh surface (Culbertson et al. 2008; Krebs & Burns 1978; Michel & Rutherford 2013). Ribbed mussels are also highly sensitive to oiling, and numerous cases of oil-induced mortality from toxicity or smothering have been reported (Culbertson et al. 2008; Michel & Rutherford 2013).

Trustees evaluated injury to marsh fauna by relating adverse effects observed in the field or in laboratory studies to shoreline oiling classifications and associated PAH concentrations in marsh soil and submerged sediment. An analysis of inundation events in 2010 provides evidence that fauna associated with marsh edge habitats would have been able to access the marsh surface (and be exposed to oiled sediment and vegetation) for a majority of the hydroperiod (J. Oehrig et al. 2015). Determinations of the length of shoreline affected are based on the 2008 SCAT shoreline. As previously noted, the length of the oiled marsh edge in Louisiana may exceed these estimates by up to 40 percent in some areas.

4.6.4.5.1 Amphipods

Key Points



- Amphipods are representative of organisms living in the soil and sediment that are a primary source of prey for many fish and invertebrates in the food web utilizing the marsh edge.
- Marsh soil conditions at heavier oiled and heavier persistently oiled Louisiana mainland herbaceous salt marsh would be expected to reduce survival of amphipods by 37 to 96 percent in 2010 when compared to shorelines where no oil was observed.
- The quantity of amphipods removed from the marsh ecosystem is estimated at 407 metric tons[3] over 155 miles (249 kilometers) of mainland herbaceous salt marsh.

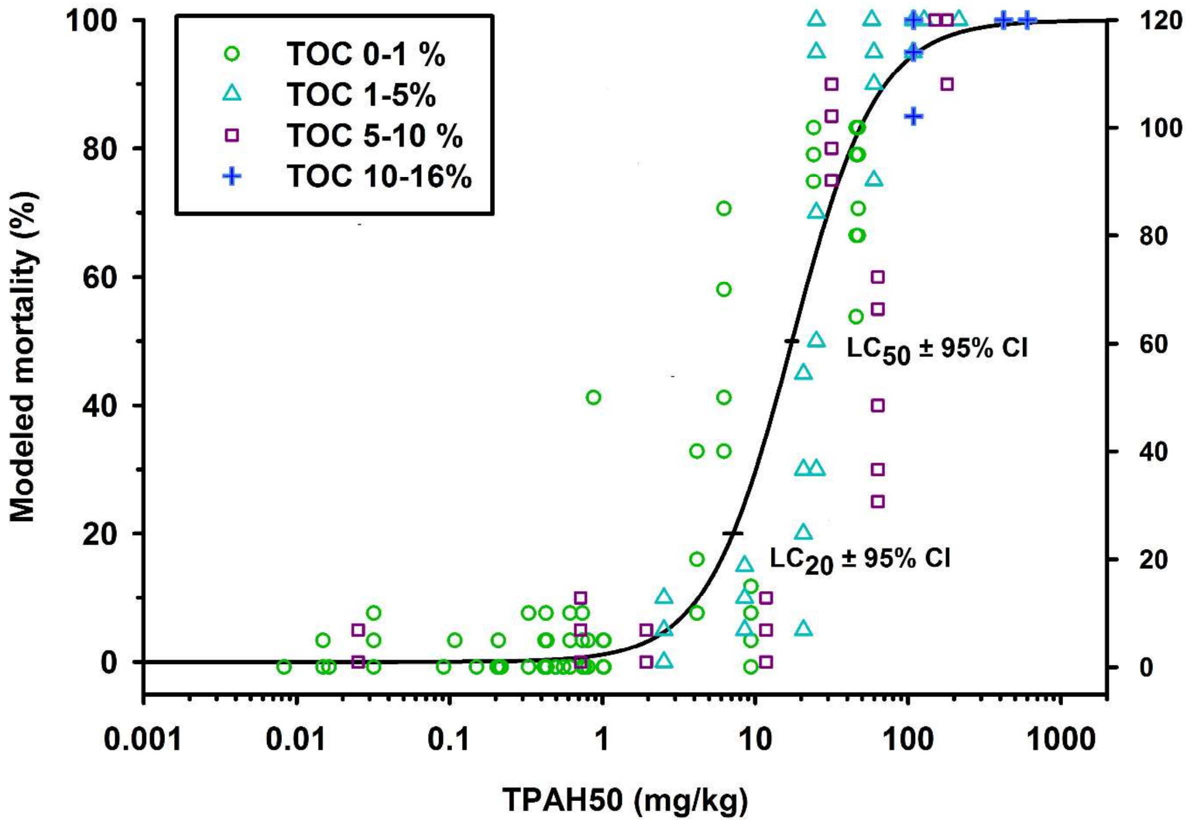
The estuarine benthic amphipod, *Leptocheirus plumulosus,* is a native of east coast estuaries. Although not a native species to the Gulf of Mexico, it is related both taxonomically and functionally to other amphipod crustaceans commonly found in Gulf of Mexico estuaries. *L. plumulosus* is frequently used to evaluate contamination because of its sensitivity and ease of culture and handling (Schlekat & Scott 1994). It burrows in bottom sediment and can filter food from water passing over the bottom or graze on algae and detritus on the sediment surface, which makes it susceptible to the effects of marsh oiling. Animals that burrow in sediments, including polychaetes, oligochaetes, and crustaceans, are a primary source of prey for many predators that aggregate near the marsh edge, including white and brown

[3] A metric ton (MT) equals 2,240 pounds (1,000 kilograms) or about the weight of four average sized bluefin tunas.

shrimp, *Fundulus*, and flounder (Fry et al. 2003; McTigue & Zimmerman 1998). The Trustees evaluated injury to *L. plumulosus* as a representative of sensitive animals that burrow in soils and sediment of mainland herbaceous and back barrier salt marshes, mangroves, and Delta *Phragmites* habitats.

Injury Determination

Marsh soil conditions at "heavier" and "heavier persistently" oiled mainland herbaceous salt marsh in Louisiana would be expected to reduce survival of amphipods by 37 to 96 percent in 2010 when compared to shorelines where no oil was observed (Powers & Scyphers 2015). Based on PAH concentrations measured in 2010, soils at the edge (zone 1) of heavier persistently oiled marshes would kill 96 percent of the amphipods using this area (Powers & Scyphers 2015). As part of the coastal wetland vegetation study mentioned above (Hester et al. 2015), PAH concentrations were measured in marsh surface soil samples taken at three distances from the vegetation edge between fall of 2010 and fall of 2013. The Trustees conducted laboratory studies to evaluate the effect of MC252 oil on amphipod survival. Amphipods were placed on sediments spiked with weathered MC252 oil for 10 days over a range of TPAH50 concentrations and total organic carbon content (0.4 to 15 percent) that represent those found at oiled marsh sites. Amphipods exposed to oiled sediment had a significantly higher rate of mortality than those exposed to clean sediments (Morris et al. 2015). However, TPAH50 soil concentrations from *Phragmites,* back barrier, and mangrove sampling stations did not exceed effects concentrations from the laboratory toxicity test.



Source: *Morris et al. (2015).*

Figure 4.6-21. This figure shows the relationship between TPAH50 concentrations in oiled marsh soil and death of amphipods over 10 days. As TPAH50 concentrations increase (horizontal axis), the percent of amphipods dying (vertical axis) increases. Twenty percent of amphipods die at a TPAH50 concentration of 7.16 parts per million, and 50 percent die at a concentration of 17.4 parts per million.

Injury Quantification

Toxicity of marsh soils to amphipods reduces the availability of this important prey species for fish, crabs, and birds (McTigue & Zimmerman 1998). Reduced survival in several toxicity tests was converted to lost amphipod production over time using literature values of densities of amphipods on the marsh surface (in two zones), number of reproductive events per year, growth information, and a calculation of the total area over which survival and production was reduced (Powers & Scyphers 2015). A survival “penalty” (i.e., the percent of animals that would die at the average soil concentration in each zone) was applied for areas where field concentrations exceeded minimum concentrations in the laboratory toxicity test associated with reduced survival (Figure 4.6-21). Injury was calculated for two areas: (1) an edge area to include a 6-meter wide swath of mainland herbaceous marsh surface where average measured soil TPAH50 concentrations exceeded concentrations (Morris et al. 2015; Powers & Scyphers 2015) shown to inhibit survival (2) and a 30-foot (9-meter) wide interior area between 20 and 49 feet (6 and 15 meters) from the marsh edge. These zone widths encompass the areas over which soil PAH

concentrations were measured. The adjacent submerged sediments within 169 feet (50 meters) of the marsh edge did not exceed concentrations toxic to amphipods (Powers & Scyphers 2015).

The quantity of amphipods removed from the marsh ecosystem is estimated at 407 metric tons (MT) over 155 miles (249 kilometers) of mainland herbaceous marsh shoreline in 2010 (Table 4.6-12). This effect occurred in the edge zone of over 39 miles (62 kilometers) of heavier persistently oiled shoreline from 2010 through 2013. In the interior zone, amphipod production was reduced over this shoreline length through 2011 (Powers & Scyphers 2015). For the 116 miles (187 kilometers) of mainland herbaceous shoreline that experienced heavier oiling conditions, amphipod production was reduced in 2010.

The approach to calculating these losses is illustrated in Figure 4.6-22. Sources of uncertainty in this calculation include: variations in concentration of PAHs in each zone; variation of interior zone widths; variation in response of the animals in the laboratory toxicity test; uncertainty in the length of shoreline miles oiled; and uncertainties in baseline densities of amphipods, growth ratios, and number of reproductive events per year. Injury in these areas will persist into the future as long as TPAH50 concentrations in heavier persistently oiled marsh soils remain elevated above concentrations associated with reduced survival (approximately 7.6 parts per million TPAH50).

Table 4.6-12. A total of 155 miles (249 kilometers) of oiled shoreline were toxic to amphipods. The outlined box represents shoreline lengths where injury occurred. Numbers within the highlighted box are summed to calculate total shoreline length affected.

LOUISIANA	Mainland Salt/Brackish Marsh		Back barrier Salt/Brackish Marsh		Delta/Inland Fresh/Intermediate Marsh		Mangrove/Marsh Complex	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	355	571	7	11	50	80	28	45
HEAVIER OILING	116	187	11	18	36	58	8	13
HEAVIER PERSISTENT OILING	39	62	0	0	3	5	3	5

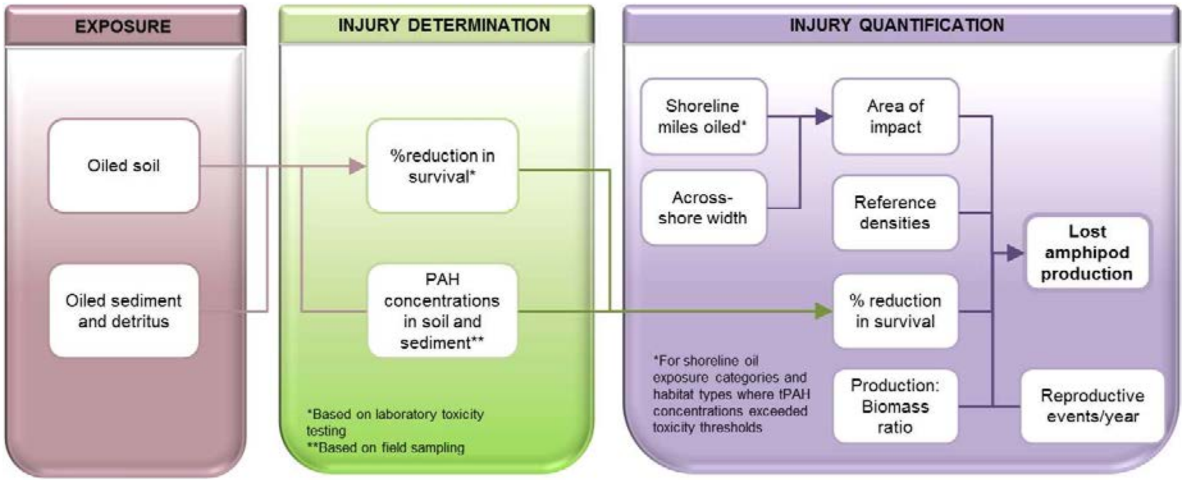


Figure 4.6-22. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to amphipods occurred. Concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-21) to determine the reduction in survival. The length and width of each zone is used to calculate how many amphipods would die. Literature values for growth and reproduction are used to quantify the weight of lost amphipods over the time that the concentrations exceed toxic thresholds.

4.6.4.5.2 Periwinkles

Key Points

- Marsh periwinkles are an important part of the marsh-estuarine food chain.
- Marsh periwinkles were affected by both oiling and cleanup actions after the spill. Densities of periwinkles were reduced by 80 to 90 percent at the oiled marsh shoreline edge and by 50 percent in the oiled marsh interior. Shoreline cleanup actions further reduced adult snail density and reduced snail size.
- An estimated 204 MT of periwinkles were lost over 39 miles (62 kilometers) of heavy persistently oiled shorelines in Louisiana mainland herbaceous marshes.
- Recovery of the number of snails may take 3 to 5 years once coastal wetland vegetation recovers, but a normal size range of snails is not expected until at least 2021.

Marsh periwinkles (*Littoraria irrorata)*, a salt marsh snail, are widely distributed, abundant, and conspicuous grazers on algae and fungi that grow on the stems and leaves of marsh plants and on soils (Montague et al. 1981; Subrahmanyam et al. 1976). Through their grazing activities, marsh periwinkles support the production of organic matter, nutrient cycling, and marsh-estuarine food chains (Kemp et al. 1990; Newel & Bárlocher 1993). They are important prey items for various animals found in salt marshes, including blue crab (*Callinectes sapidus*) (Hamilton 1976; Silliman & Bertness 2002). They are vulnerable to oiling impacts because they are closely associated with the soil and emergent salt marsh vegetation, typically *Spartina alterniflora* (Figure 4.6-23) (Hershner & Lake 1980; Hershner & Moore 1977; Krebs & Tanner 1981; Lee et al. 1981).

Source: NOAA Deepwater Horizon SCAT Program.

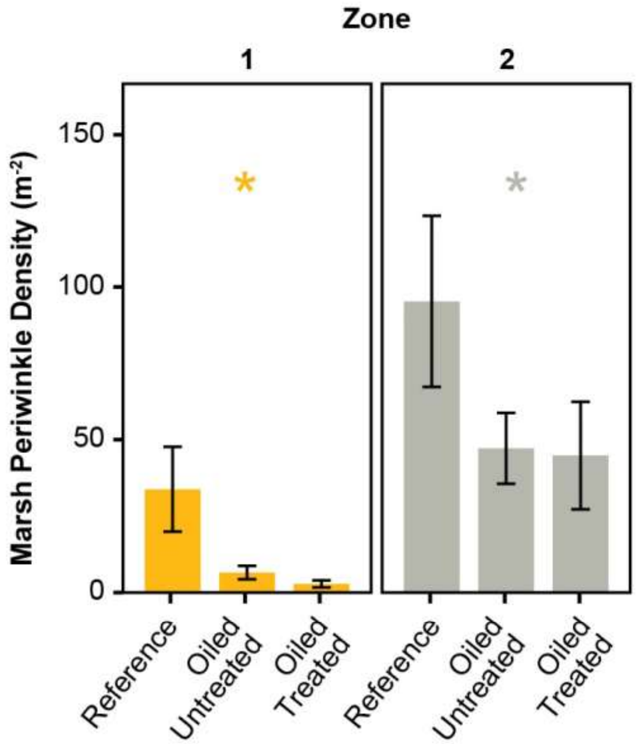
Figure 4.6-23. Heavily oiled salt marsh in Louisiana in the months following the spill. Marsh periwinkles are evident on vegetation stems.

Injury Determination

Oiling and response actions directly affected marsh periwinkles. Densities of periwinkles were reduced by 80 to 90 percent at the oiled marsh shoreline edge and by 50 percent in the oiled marsh interior (Zengel et al. 2015b). Shoreline cleanup actions further reduced adult snail density and reduced snail size. Periwinkles were reduced primarily as a result of oiling, but they were reduced further by cleanup disturbance during raking, washing, booming, and other activities. It is likely that heavy oiling directly killed periwinkles through physical smothering and fouling by thick emulsified oil, and perhaps also through toxic effects of chemicals in the oil (Zengel et al. 2015b). Additionally, plant death and resulting low vegetation cover limited the recovery of snails after the spill; and residual oiling on the marsh substrate and elevated TPAH50 levels in surficial marsh soils likely also negatively affected periwinkles (Zengel et al. 2015a).

A field study conducted in fall 2011 in mainland marshes in coastal Louisiana examined these impacts on periwinkles (Zengel et al. 2015b). Three types of study sites were selected: sites with heavier persistent oiling where cleanup actions were conducted, sites with heavier persistent oiling without cleanup actions, and reference sites where no oil was observed during marsh surveys. Marsh periwinkle snail density and shell lengths at sites with heavier persistent oiling were compared to snails in reference conditions. Periwinkle density and size were also evaluated between oiled sites, with and without shoreline cleanup treatments. Overall, 32 marsh edge sampling stations (zone 1) were located 2 meters from the shoreline, and 35 interior stations (zone 2) were located an average of 9 meters from the shoreline. An additional third zone of interior marsh was located inland of observed oil penetration, where stations were placed an average of 21 meters from the shoreline. No effects on periwinkles were

observed in the third zone located inland of observed oiling (Zengel et al. 2015b). Results for zone 1 and 2 are shown in Figure 4.6-24.



Source: *Zengel et al. (2015b).*

Figure 4.6-24. Numbers of periwinkle snails were much lower in oiled areas compared to unoiled (reference) areas. This was true at the edge of the marsh (zone 1) and in interior oiled areas (zone 2). Cleanup actions (treatment) further reduced the number of snails at the marsh edge.

The Trustees also conducted laboratory studies to evaluate the effect of MC252 oil on periwinkle survival and behavior. Snails placed in trays with flattened heavily oiled marsh stems were unable to move toward unoiled standing vegetation at the end of the trays. Snails in clean conditions were easily able to move short distances to reach standing vegetation within one to two hours. Eight hours of exposure to heavy oiling conditions caused increased snail death (Morris et al. 2015) (see Section 4.3, Toxicity).

In addition, indirect evidence suggests that the oil spill may have caused a recruitment failure in marsh periwinkles in 2010, due to widespread oiling in coastal waters that may have affected planktonic periwinkle larvae and led to a "missing generation" of snails that would have settled in marsh areas but did not (Pennings et al. 2015 [what is this report?]).

Other studies of salt marsh habitats oiled by the *Deepwater Horizon* oil spill also found reduced densities of marsh periwinkles within heavily oiled areas (Silliman et al. 2012; Zengel et al. 2015a; Zengel et al. 2014).

Injury Quantification

Had they not been killed by oiling and cleanup actions, periwinkles would have continued to survive and grow over time. Based on the total area of marsh with oiling conditions similar to those where periwinkles were killed, reductions in snail densities in oiled areas, and age/growth/survival relationship assumptions, the total loss of snails can be converted to “lost production” over time (Powers & Scyphers 2015). The approach to calculating these losses is illustrated in Figure 4.6-25. The density reductions observed in the field study from the edge and inland zones were applied over a 6-meter wide “edge” zone and a 9-meter wide “inland” zone (intended to bound the midpoints of each of the zones where injury was evaluated). The effect of oiling would equate to a total loss of 204 MT (whole wet weight) from the marsh system (Powers & Scyphers 2015).

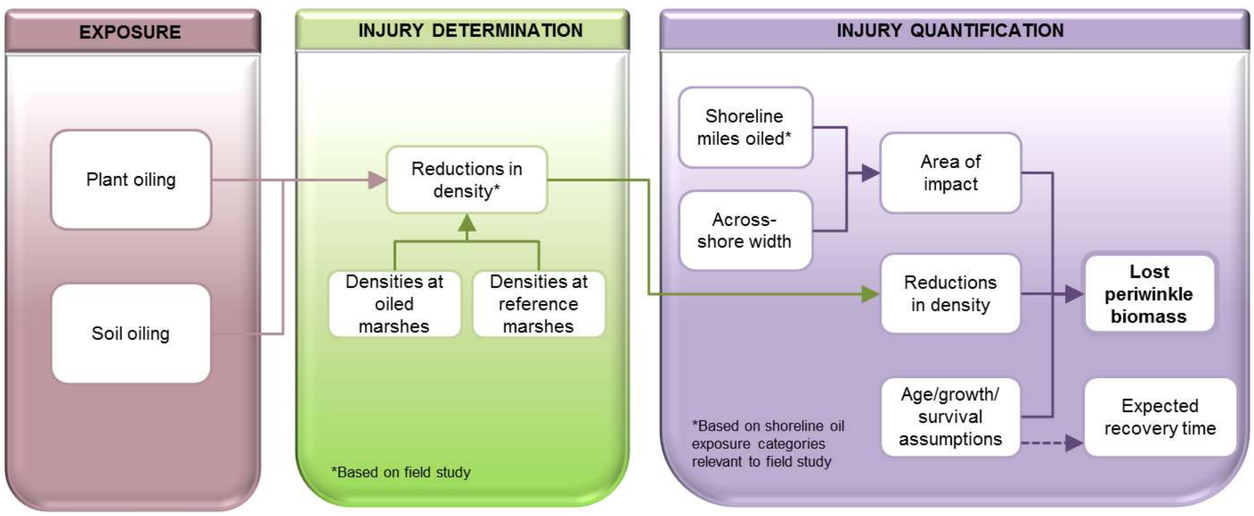
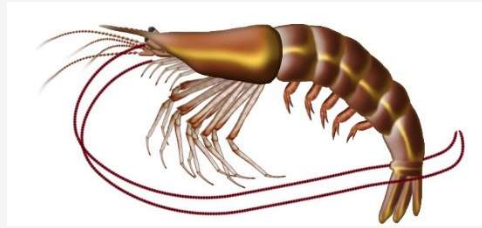


Figure 4.6-25. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to periwinkles occurred. Measured reductions in snail numbers in each zone (compared to reference sites) and the length and width of each zone are used to calculate how many snails died. Literature values for growth and survival are used to quantify the weight of lost snails over the time that effects were observed.

These effects would be expected over 39 miles (62 kilometers) of heavy persistently oiled shorelines in Louisiana mainland herbaceous marshes. Sources of uncertainty in this calculation include variations in interior zone widths, variation in the number of periwinkles found at each station, uncertainty in the length of shoreline miles oiled (shoreline miles were estimated for a subset of Louisiana heavily persistently oiled miles from Table 4.6-3), and uncertainties in baseline densities of snails and growth assumptions. Based on the size of adult periwinkles observed in this study and literature information on age and growth, assuming recovery of the same area of habitat as existed before the spill, population recovery is likely to take a minimum of 3 to 5 years once oiling and habitat conditions in affected areas are suitable to support normal recruitment, immigration, survival, and growth. Because periwinkle snails can live for 10 years or more, a normal size distribution of snails in affected areas would not be re-established until at least 2021 (Powers & Scyphers 2015).

4.6.4.5.3 Shrimp

Key Points



- White shrimp and brown shrimp are important components of the Gulf of Mexico ecosystem and food web and support a robust commercial fishery.
- Juvenile penaeid shrimp growth was dramatically affected by oiling.
- The total loss of shrimp production over 2010 and 2011 due to oiling is estimated at 2,089 MT. In comparison, the annual harvest of shrimp in Louisiana was 17,700 MT.
- The total marsh shoreline over which shrimp production was reduced due to oiling in 2010 and 2011 is 179 miles (288 kilometers).

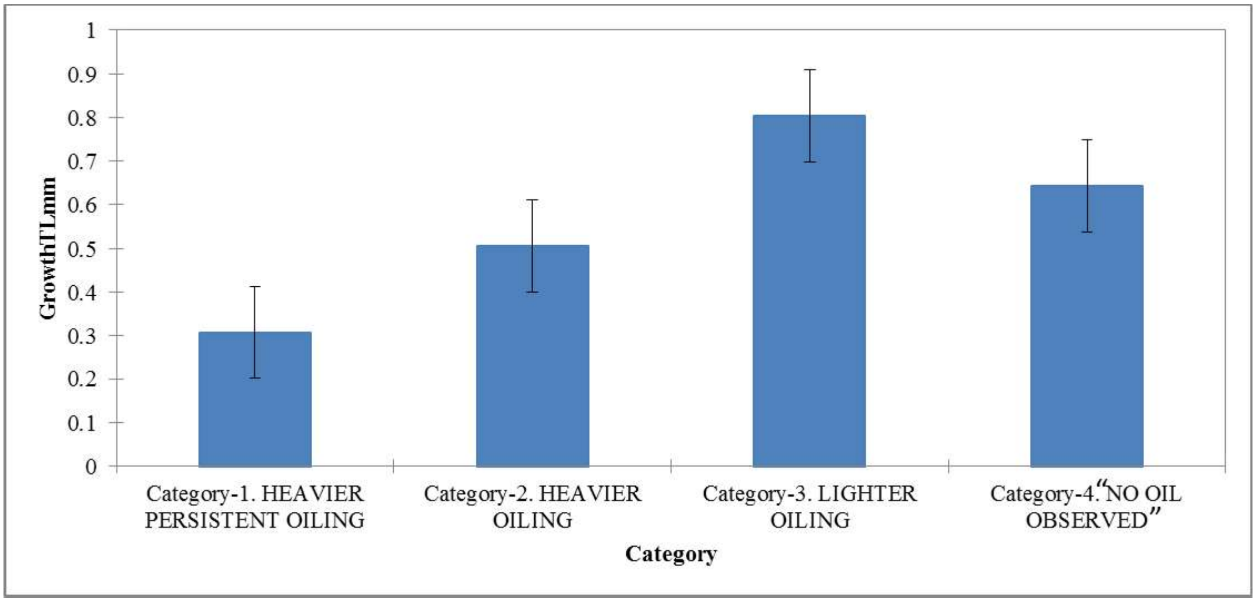
White shrimp (*Liptopenaeus setiferus)* and brown shrimp *(Farfantepenaeus aztecus*) are important components of the Gulf of Mexico ecosystem and food web and support a robust commercial fishery (Zimmerman et al. 2000). Adult shrimp spawn in open waters of the Gulf of Mexico. As they develop, they move into estuaries and settle to bottom sediments adjacent to marsh shorelines, where they grow rapidly (Fry et al. 2003; Haas et al. 2004; Minello & Rozas 2002). The shallow wetland habitats (particularly salt marsh and mangroves) of Barataria Bay and other northern Gulf of Mexico estuaries support high densities of juvenile brown shrimp and white shrimp (Roth 2009; Rozas & Minello 2010). Juvenile brown shrimp use the estuary in the winter and spring months, while juvenile white shrimp use the estuary in late summer and fall months. Both species are also found at least 3 meters onto the surface of flooded marshes, where they are opportunistic feeders on infauna (worms in the sediment), plants, and detritus (McTigue & Zimmerman 1998; Minello & Zimmerman 1991a).

Injury Determination

Juvenile penaeid shrimp growth was dramatically affected by oiling of marsh habitats in 2010 and 2011 (Powers & Scyphers 2015). Based on field growth assay conducted by Rozas et al. (2014), shrimp growth was reduced in areas of substantive oiling. Rozas et al. (2014) conducted experiments in May 2011 for brown shrimp and October 2011 for white shrimp. Shrimp were incubated in cages adjacent to the marsh edge at 25 sites for 1 to 2 weeks. All experiments were performed in Barataria Bay. Powers and Scyphers (2015) utilized data from Rozas et al. (2014) and updated the shoreline oiling classifications to be consistent with the NRDA classifications. Five shoreline oiling categories used in the Rozas et al. (2014) study design were reclassified using four NRDA shoreline oiling categories for purposes of injury assessment (Zachary Nixon et al. 2015b).

Rozas et al. (2014) demonstrates that along heavier persistently oiled and heavier oiled shorelines, growth of juvenile white shrimp was reduced by 31 to 46 percent, and juvenile brown shrimp growth was reduced by 27 to 56 percent compared to sites that did not experience oiling (Figure 4.6-26). Sediment PAH concentrations measured in the Rozas et al. (2014) study were less than 1 parts per million. However, correlations between shrimp growth reductions and heavy shoreline oiling (where

concentrations of PAHs in marsh soils are extremely elevated) indicate that the observed effects are the result of integrated exposure from multiple pathways, including contaminated sediment and runoff from oiled marsh habitat in 2011 (Powers & Scyphers 2015). These results are consistent with those of van der Ham and de Mutsert (2014), who suggest that "exposure to polycyclic aromatic hydrocarbons (PAHs) may have reduced the growth rate of shrimp, resulting in a delayed movement of shrimp to offshore habitats."



Source: *Powers and Scyphers (2015).*

Figure 4.6-26. Growth rates of juvenile brown shrimp associated with marshes of various degrees of oiling. Growth rates were reduced by 27 to 56 percent compared to sites that did not experience oiling.

In addition to the effects of marsh oiling on the growth of juvenile penaeid shrimp, summer river water releases and resulting reduced salinity as part of spill response likely reduced juvenile brown shrimp production by affecting benthic prey abundance or through the stress of adapting to lower salinity conditions (Adamack et al. 2012). Adamack et al. (2012) modeled the effects of a late April/May water release and concluded that shrimp production would be 40 to 60 percent less than under baseline conditions with no water release. Benthic prey quantity dropped from 60 mg/core to 5 mg/core in areas where salinity was less than 5 parts per thousand (Rozas & Minello 2011) in data used by Adamack et al. (2012). White shrimp juveniles would not have been affected by these freshwater conditions since most movement of juvenile white shrimp into marshes occurs later in the summer and fall. Reductions in brown shrimp production from river water releases would be expected to occur over an additional area where salinities dropped below 5 parts per thousand in 2010 when compared to prior years (Figure 4.6-14). This area was determined by interpolating thousands of salinity values throughout the estuary and comparing 2010 salinities to those in the years prior to the spill (2006 to 2009) (McDonald et al. 2015; Rouhani & Oehrig 2015b).

Estimates of prior year conditions are intended to represent salinity conditions that were likely to have occurred in 2010 had the release of river water not occurred as part of the response action. For each

200 square meter grid cell in the salinity model, the maximum number of consecutive days of low salinity (i.e., below 5 parts per thousand) between April 27 and September 15 was calculated for each year between 2006 and 2010. For each grid cell, the maximum number of consecutive days was averaged for years 2006-2009 to represent the "historical baseline condition" for that location. Each grid cell that experienced more than 30 consecutive days of low salinity above the historical baseline condition was considered affected by fresh water in 2010. The threshold of 30 days was selected to maximize the difference between average salinities inside and outside the resulting fresh water polygon in 2010, thereby representing the greatest low salinity impact (Rouhani & Oehrig 2015b).

Freshwater conditions may have affected animals in addition to shrimp. Rose et al. (2014) suggested negative impacts of river water releases on estuarine dependent fishes and invertebrates, but did not quantify the effects in terms of lost production. Rose et al. (2014) included in their study a specific model of the river water releases during the *Deepwater Horizon* (i.e. "Oil Spill" scenario). The results of Rose et al. (2014) suggest that the effect demonstrated for brown shrimp likely extend to other fauna.

Injury Quantification

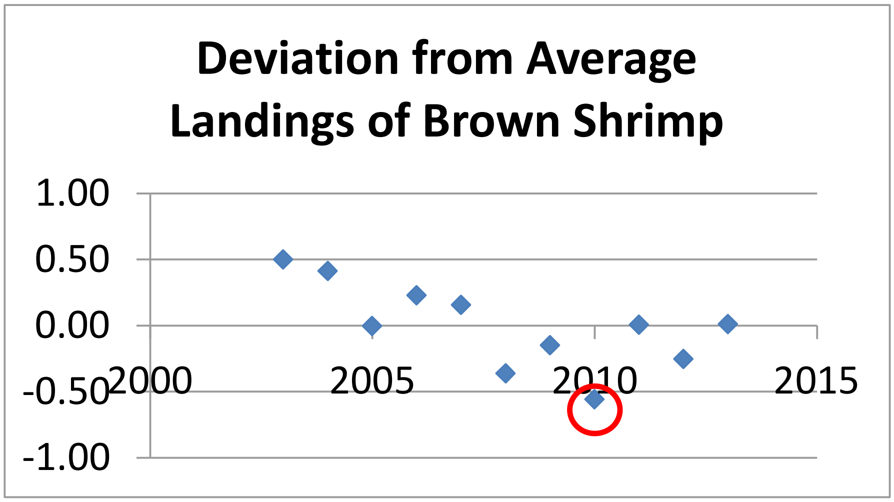
Reductions in juvenile penaeid shrimp growth observed along oiled shorelines would translate directly into fewer adult shrimp, since predation and other mortality is size-dependent (Adamack et al. 2012; Powers & Scyphers 2015). The faster the shrimp grow, the better their chances of avoiding predators and surviving to reach adult life stages. Reduced shrimp growth was converted to lost shrimp production using several factors. They include: field observations of growth in reference and oiled conditions; literature values of juvenile penaeid shrimp densities on the marsh surface and adjacent subtidal habitat; the total area of marsh and adjacent sediment used by juvenile penaeid shrimp with oiling conditions similar to those where shrimp growth was reduced; growth curves; and size-dependent survival relationships (Powers & Scyphers 2015).

Injury was calculated for an area to include a 1-meter wide swath of marsh surface and a 50-meter wide area of adjacent sediment, which is the area where shrimp have been observed in prior studies. Growth "penalties" are assessed for the period shrimp are exposed to marsh edge, after which time shrimp are assumed to forage in open waters away from the marsh edge. The effect of oiling would equate to a total loss of 1,176 MT wet weight of brown shrimp from the marsh system over 2010 and 2011. During the same period, approximately 913 MT of white shrimp production was lost due to oiling (Powers & Scyphers 2015).

Sources of uncertainty in this calculation include variations in interior zone widths, variation in the growth of shrimp in each treatment group, uncertainty in the length of shoreline miles oiled, uncertainties in baseline densities of shrimp, and uncertainties in growth assumptions. Losses of brown and white shrimp production total an estimated 2,089 MT (wet weight) over 2 years. This can be compared to an annual harvest in Louisiana of 17,700 MT wet weight of penaeid shrimp. Although harvest was closed after the spill, brown shrimp landings were low in 2010 (Figure 4.6-27).

Oiling effects persisted for at least 2 years (i.e., into fall 2011) along 179 miles (288 kilometers) of heavier oiled and heavier persistently oiled shoreline in Louisiana and Mississippi, including mainland herbaceous marsh, mangrove marsh, and back-barrier salt marsh shorelines (Table 4.6-13). Since PAH

concentrations remained elevated in marsh soils into 2012 and 2013, reductions in shrimp production likely persisted later than 2011 in areas that experienced heavier persistent oiling (Powers & Scyphers 2015). This injury would continue for as long as heavier persistent oiling conditions are present (or until the oiled marsh edge areas erode).



Source: Powers and Scyphers (2015).

Figure 4.6-27. Deviation from average landings of brown shrimp in Louisiana. Landings in 2010 were the lowest during the time period 2003 to 2013.

Table 4.6-13. Oiling reduced juvenile penaeid shrimp growth over a total of 179 miles (288 kilometers) of shoreline in Louisiana and Mississippi. Outlined boxes represent shoreline lengths where injury occurred. Numbers within highlighted boxes are summed to calculate total shoreline length affected.

MISSISSIPPI	Mainland Salt/Brackish Marsh		Back barrier Salt/Brackish Marsh		Delta/Inland Fresh/Intermediate Marsh		Mangrove/Marsh Complex	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	18	29	8	12	0	0	0	0
HEAVIER OILING	0	0	2	3	0	0	0	0
HEAVIER PERSISTENT OILING	0	0	0	0	0	0	0	0
LOUISIANA	**Mainland Salt/Brackish Marsh**		**Back barrier Salt/Brackish Marsh**		**Delta/Inland Fresh/Intermediate Marsh**		**Mangrove/Marsh Complex**	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	355	571	7	11	50	80	28	45
HEAVIER OILING	116	187	11	18	36	58	8	13
HEAVIER PERSISTENT OILING	39	62	0	0	3	5	3	5

The approach to quantifying shrimp losses from oiling is illustrated in Figure 4.6-28.

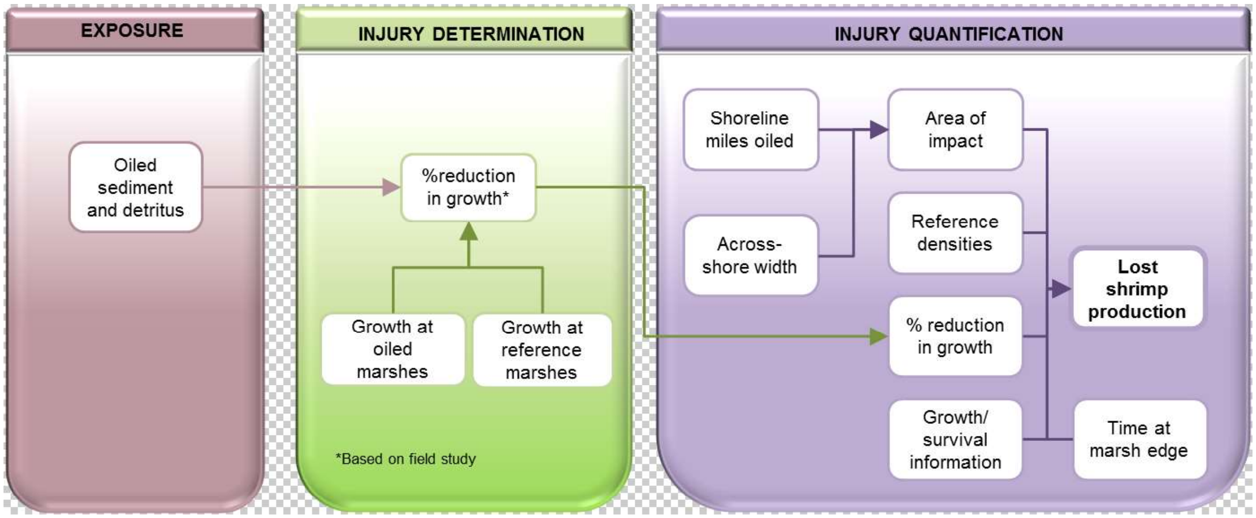


Figure 4.6-28. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to shrimp occurred.

The loss of substantive production of penaeid shrimp is an indicator of degraded marsh conditions throughout a large area of the northern Gulf of Mexico during and after the *Deepwater Horizon* oil spill. The tight linkage between quality and quantity of healthy marsh and penaeid shrimp production is well established for the Gulf of Mexico (Minello et al. 2003; Minello & Zimmerman 1991b; Rozas & Minello 2009).

4.6.4.5.4 Forage Fish

Key Points

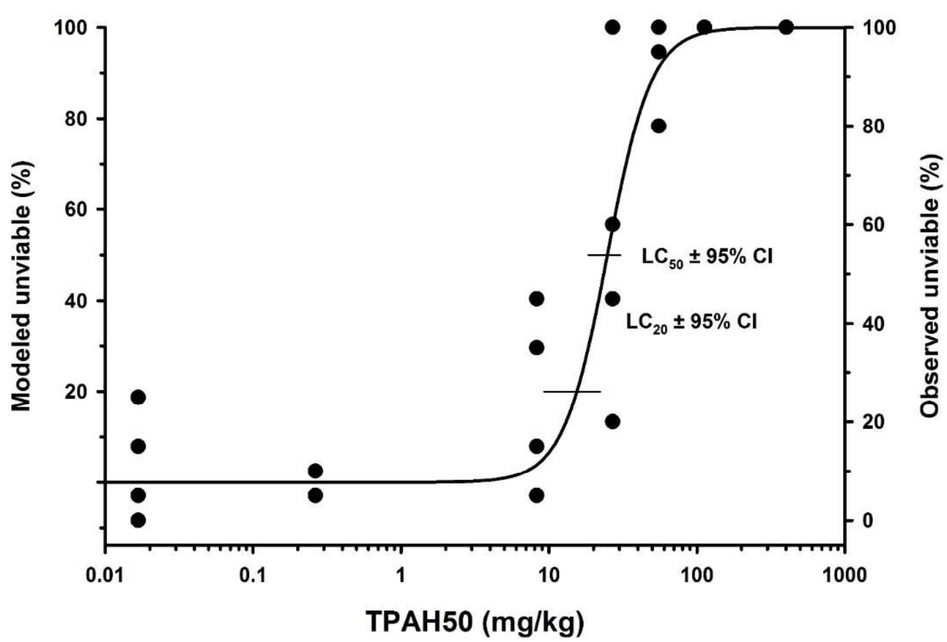
- Gulf killifish *(Fundulus grandis*) is an important part of the food web. Among the largest of the Gulf forage fish, it is preyed upon by wildlife, birds, and many sport fish, including flounder, spotted sea trout, and red snapper.

- Marsh soil conditions at heavier persistently oiled Louisiana mainland herbaceous salt marsh reduced successful hatching of *Fundulus* eggs by 68 to 99 percent when compared to conditions where no oil was observed.
- An estimated total of 84.7 MT wet weight of *Fundulus* was lost where soils exceeded toxic TPAH50 concentrations. This effect occurred over 39 miles (62 kilometers) of oiled shoreline in 2010.
- Injury in these areas will persist into the future as long as PAH concentrations in soils remain elevated above concentrations associated with reduced hatch success (approximately 15 parts per million TPAH50).

Gulf killifish *(Fundulus grandis)* is an important connector of energy derived from the marsh surface to open Gulf waters (Ross 2001). They are among the largest of the Gulf forage fish, preyed upon by wildlife, birds, and many sport fish, including flounder, spotted sea trout, and red snapper (Ross 2001). Though adult *Fundulus* are known to be tolerant of high temperatures and other stressors, their importance to marsh food webs make them a common choice for evaluating the effects of contaminants. Eggs and larvae of *Fundulus* are expected to be more sensitive to the effects of contamination than adults. *Fundulus* lay their eggs on the marsh surface. These eggs adhere to marsh grass and debris in the water column or near the muddy bottom, where they would be exposed to oil. *Fundulus* are known to use the entire flooded surface of mainland herbaceous, back barrier, mangrove, and delta *Phragmites* marsh habitats (Rozas & LaSalle 1990).

Injury Determination

Marsh soil conditions at heavier persistently oiled mainland herbaceous salt marsh shorelines in Louisiana reduced successful hatching of *Fundulus* eggs by 68 to 99 percent when compared to conditions where no oil was observed (Figure 4.6-29). As part of the coastal wetland vegetation study mentioned above (Hester et al. 2015), TPAH50 concentrations were measured in marsh surface soil samples taken at three distances from the vegetation edge between fall of 2010 and fall of 2013. The Trustees conducted laboratory studies to evaluate the effect of MC252 oil on *Fundulus* egg hatchability. Fertilized eggs were placed above sediments spiked with weathered MC252 oil over a range of concentrations representing those found at oiled marsh sites. Eggs exposed to oiled sediments were significantly less likely to hatch than those exposed to clean sediments (Morris et al. 2015).



Source: *Morris et al. (2015).*

Figure 4.6-29. Relationship between TPAH50 concentration in marsh soil and hatching success of *Fundulus* eggs. As concentrations of TPAH50 increase (horizontal axis), the percentage of fish eggs that do not hatch increases (vertical axis). At low concentrations, most fish hatch within 15 to 18 days. At high concentrations, many eggs did not hatch at all.

Other studies have confirmed that *Deepwater Horizon* oil has affected marsh forage fish. Dubansky et al. (2013) found signs of oil exposure (enzyme marker induction) and gill abnormalities (an increase in hyperplasia) in fish taken from oiled sites in 2010. Their laboratory exposures of Gulf killifish embryos to field-collected sediments from Grande Terre and Barataria Bay, Louisiana, also resulted in developmental abnormalities (e.g., failure to hatch, lower growth, slower heart rate, and increased yolk sac and pericardial edema) when compared to exposure to sediments collected from a reference site (Dubansky et al. 2013).

Injury Quantification

Reductions in *Fundulus* egg hatching success associated with exposure to oiled marsh soils would translate directly into fewer adult *Fundulus* available as prey for higher tropic levels. Reduced hatching success was converted to lost production of *Fundulus* adults using literature values of densities of fish eggs on the marsh surface, survival to the adult stage, the number of spawning events per year, average weight of adults, and a calculation of the total area over which hatching success and production was reduced (Powers & Scyphers 2015). A fecundity "penalty" was applied for areas exceeding TPAH50 concentrations in the laboratory toxicity test associated with reduced hatch success. Injury was calculated for an area to include a 6-meter wide swath of mainland herbaceous marsh surface edge habitat where soil TPAH50 concentrations exceeded concentrations shown to inhibit hatch success (Powers & Scyphers 2015). The approach to calculating lost *Fundulus* production is illustrated in Figure 4.6-30.

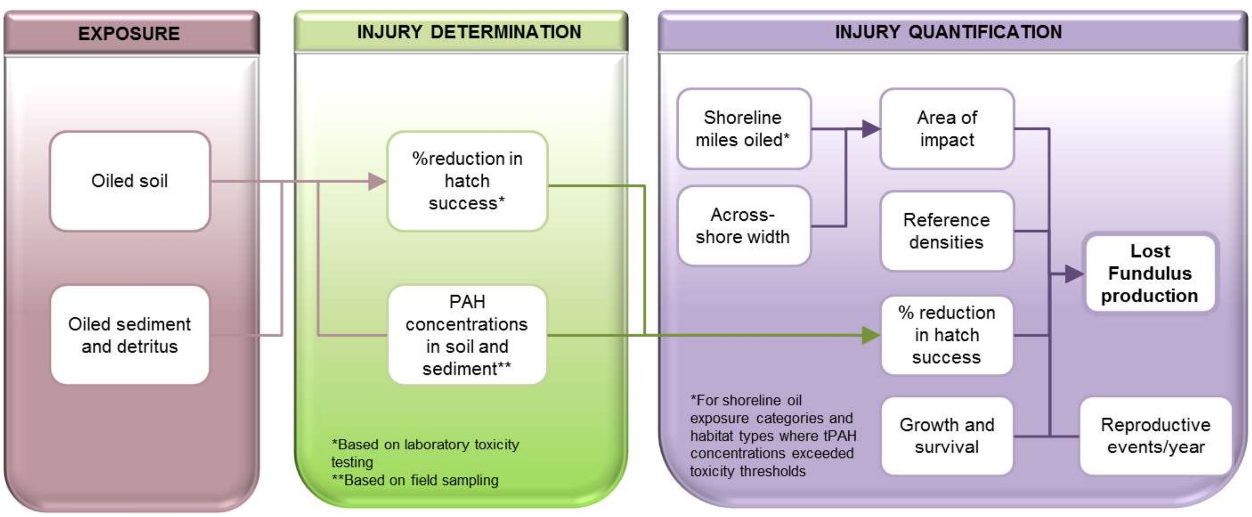


Figure 4.6-30. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to *Fundulus* occurred. Concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-29) to determine the reduction in hatch success. The length and width of each zone is used to calculate how many *Fundulus* eggs would not hatch. Literature values for growth and reproduction are used to quantify the weight of lost *Fundulus* over the time that TPAH50 concentrations exceed toxic thresholds.

An estimated total of 84.7 MT wet weight of *Fundulus* was lost due to marsh oiling in mainland herbaceous salt marsh in Louisiana where concentrations of TPAH50 in marsh soils exceeded toxic

effects thresholds. This effect occurred over 39 miles (62 kilometers) of heavier persistently oiled shoreline from 2010 to 2013 (Powers & Scyphers 2015). Sources of uncertainty in this calculation include variations in TPAH50 concentrations in each zone, variations in zone widths, variation in response of the animals in the laboratory toxicity test, uncertainty in the length of shoreline miles oiled, uncertainties in baseline densities of *Fundulus*, and uncertainties in growth, survival, and reproduction assumptions (Powers & Scyphers 2015). Injury in these areas will persist into the future as long as TPAH50 concentrations in heavier persistently oiled marsh soils remain above concentrations associated with reduced hatch success (approximately 15 parts per million TPAH50). The injury to *Fundulus* also demonstrates how changes in fecundity associated with the oil spill can affect fish populations (Powers & Scyphers 2015).

Table 4.6-14. *Fundulus* hatching was impaired in 2010 over a total of 39 shoreline miles (62 kilometers). The outlined box represents shoreline lengths where injury occurred.

LOUISIANA	Mainland Salt/Brackish Marsh		Back barrier Salt/Brackish Marsh		Delta/Inland Fresh/Intermediate Marsh		Mangrove/Marsh Complex	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	355	571	7	11	50	80	28	45
HEAVIER OILING	116	187	11	18	36	58	8	13
HEAVIER PERSISTENT OILING	39	62	0	0	3	5	3	5

4.6.4.5.5 Southern Flounder

Key Points

- Flounder are a key predator in marsh ecosystems. Other predatory fish species that utilize the marsh terrace environment would experience deleterious effects similar to those experienced by flounder.
- TPAH50 concentrations at heavier persistently oiled Louisiana mainland herbaceous shorelines reduced juvenile flounder growth by 31 to 90 percent.
- An estimated total of 40 MT wet weight of flounder was lost where TPAH50 concentrations in marsh soils exceeded 12.8 parts per million. This effect occurred over 39 miles (62 kilometers) of oiled shoreline in 2011.
- Reduced flounder production persists through 2013 and would be expected to continue in heavier persistently oiled marshes until soil TPAH50 concentrations drop below 12.8 parts per million.

In addition to serving as critical habitat for the production of invertebrate fisheries (e.g., penaeid shrimp), invertebrate prey species (e.g., amphipods), and forage fish species (e.g., *Fundulus*), nearshore areas provide key habitats for high-level predators (Peterson & Turner 1994). Southern flounder

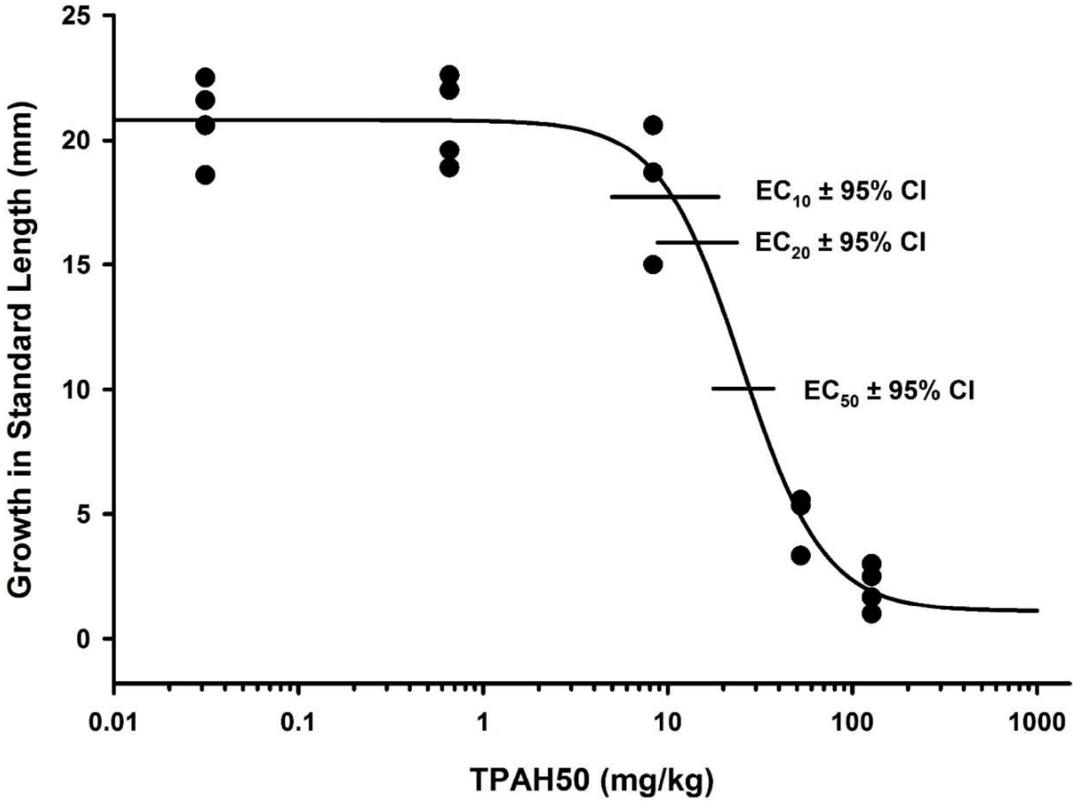
(*Paralichthys lethostigma*) use the surfaces of flooded shallow salt marsh, brackish marsh, mangrove, delta *Phragmites*, and other coastal habitats throughout the northern Gulf of Mexico (Burke 1995). Their close association with sediment makes them vulnerable to PAH in marsh soils and submerged sediments. Southern flounder spend most of their lives associated with bottom sediments. Male flounder grow more slowly and reach smaller sizes than females (Fitzhugh et al. 1996). During their first year of life, young flounder settle in bays and estuaries in the late winter and early spring, where they move onto flooded marsh surfaces to feed. Juvenile southern flounder eat small fish (including *Fundulus*), crustaceans (including amphipods and juvenile penaeid shrimp), and polychaetes. Adult southern flounder leave the bays during the fall to spawn in open waters of the Gulf of Mexico (Rogers et al. 1984).

Flounder are a key predator in marsh ecosystems; however, they are not the only bottom-oriented fish that fills this broad ecological niche. Other predators have similar habitat use (e.g., red drum, spotted sea trout): they also keep a close association with structured habitats (the most abundant such habitat in the northwestern Gulf of Mexico being herbaceous salt marsh), and they also feed on prey that flourish in marsh soil and adjacent marsh submerged sediments (Peterson & Turner 1994). The high productivity of predators in nearshore marsh environments has been closely linked to the frequent tidal and wind-driven inundation of marsh habitats. These extended hydroperiods of inundation allow fish and mobile invertebrates to forage in the refuge of a structured environment with high prey biomass (J. Oehrig et al. 2015). This behavior of fish and mobile invertebrates also exposes them to oil that has been deposited in marsh soils.

Injury Determination

Oil stranded on marsh shorelines beginning in the early summer of 2010. By this time, southern flounder juveniles would have grown substantially, though they were not yet adults (Rogers et al. 1984). In the spring of 2011, when the adult fish that survived 2010 oiling conditions spawned, TPAH50 concentrations at heavier persistently oiled mainland herbaceous shorelines in Louisiana reduced juvenile flounder growth by 31 to 90 percent (Figure 4.6-31) when compared to conditions where no shoreline oiling was observed (Powers & Scyphers 2015). Conditions toxic to flounder continued into 2012 (a 90 percent reduction in growth) and 2013 (a 32 percent reduction in growth).

As part of the coastal wetland vegetation study (Hester et al. 2015) mentioned above, TPAH50 concentrations were measured in marsh surface soil samples taken at three distances from the vegetation edge between the fall of 2010 and the fall of 2013. The Trustees conducted laboratory studies to evaluate the effect of MC252 oil on juvenile flounder growth over 32 days of exposure (Brown-Peterson et al. 2015). Juvenile flounder of lengths 1.8 to 3.5 centimeters were placed on sediments spiked with weathered MC252 oil over a range of concentrations representing those found at oiled marsh sites. Juvenile flounder exposed to oil put on less weight and reached smaller sizes than fish exposed to clean sediment. Gill and liver abnormalities were also observed in juvenile fish exposed to *Deepwater Horizon* oil contaminated sediments (Brown-Peterson et al. 2015).



Source: *Morris et al. (2015).*

Figure 4.6-31. Relationship between TPAH50 concentration in marsh soil and growth of juvenile southern flounder. As they are exposed to higher concentrations of TPAH50 (horizontal axis), juvenile southern flounder grew less (vertical axis) than fish placed on clean sediment. At low TPAH50 concentrations, most fish grew about 20 millimeters longer over the 32 days. At high concentrations, the flounder that survived the test grew less than 5 millimeters longer.

Injury Quantification

The dramatic reduction in growth observed in juvenile southern flounder exposed to conditions at heavier persistently oiled marsh sites would translate into fewer adult southern flounder since small fish suffer higher levels of predation and death due to lingering abnormalities (Powers & Scyphers 2015). Because male flounder have shorter life spans than female flounder (4 years vs. 8 years), the consequences would be more pronounced for male flounder. This is because female flounder have faster growth rates and longer time to potentially recover from earlier stunted growth periods.

Reduced flounder growth was converted to lost flounder production using literature values of juvenile flounder densities on the marsh surface, the total area of marsh used by juvenile flounder with oil conditions similar to those where growth was reduced, and growth/survival relationships (Powers & Scyphers 2015). Injury was calculated for an area to include a 5-meter wide swath of marsh surface soil, which is where flounder have been observed in other studies. Concentrations in submerged sediment adjacent to the marsh edge were not high enough to be toxic to flounder (Powers & Scyphers 2015).

Growth "penalties" were assessed for the period that flounder are exposed to the marsh edge, until they move offshore as adults. The approach used to calculate production losses is illustrated in Figure 4.6-32.

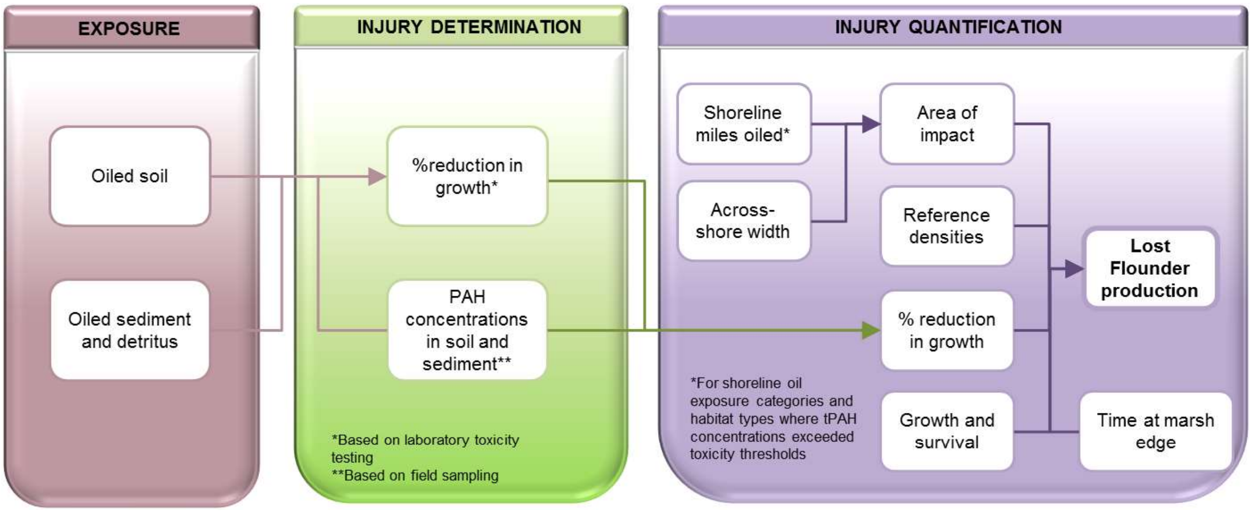


Figure 4.6-32. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to flounder occurred. Concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-31) to determine the reduction in growth. The length and width of each zone and literature values for growth and production were used to calculate lost flounder production over the time that the concentrations exceed toxic thresholds.

An estimated total of 40 MT wet weight of flounder was lost due to oiling of mainland herbaceous salt marsh in Louisiana where TPAH50 concentrations in marsh soils exceed 12.6 parts per million—the toxicity threshold concentration used for this analysis. This effect occurred from 2011 to 2013 over 39 miles (62 kilometers) of heavier persistently oiled mainland herbaceous shoreline in Louisiana (Table 4.6-15) (Powers & Scyphers 2015). Sources of uncertainty in this calculation include variations in TPAH50 concentrations in each zone, variations in zone widths, variation in response of the animals in the laboratory toxicity test, uncertainty in the length of shoreline miles, uncertainties in baseline densities of flounder, and uncertainties in growth, survival, and reproduction assumptions. Reduced flounder production persists through 2013 and would be expected to continue in heavier persistently oiled marshes until soil TPAH50 concentrations drop below 12.6 parts per million (Powers & Scyphers 2015).

Table 4.6-15. Oiling over 39 miles (62 km) of shoreline reduced juvenile flounder growth. The outlined box represents shoreline lengths where injury occurred. Numbers within the outlined box are summed to calculate total shoreline length affected.

LOUISIANA	Mainland Salt/Brackish Marsh		Back barrier Salt/Brackish Marsh		Delta/Inland Fresh/Intermediate Marsh		Mangrove/Marsh Complex	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	355	571	7	11	50	80	28	45
HEAVIER OILING	116	187	11	18	36	58	8	13
HEAVIER PERSISTENT OILING	39	62	0	0	3	5	3	5

Because a range of predatory fish species utilize marsh ecosystems (Peterson & Turner 1994), it is safely assumed that other species that utilize the marsh terrace environment would experience deleterious effects similar to those experienced by southern flounder. In addition, the reduction in prey due to lost benthic biomass (e.g., amphipods) from oiling and from summer river water release likely reduced growth further under natural conditions and served to reduce overall fitness of this wide range of predatory species that the public highly values (Powers & Scyphers 2015).

4.6.4.5.6 Red Drum

Key Points

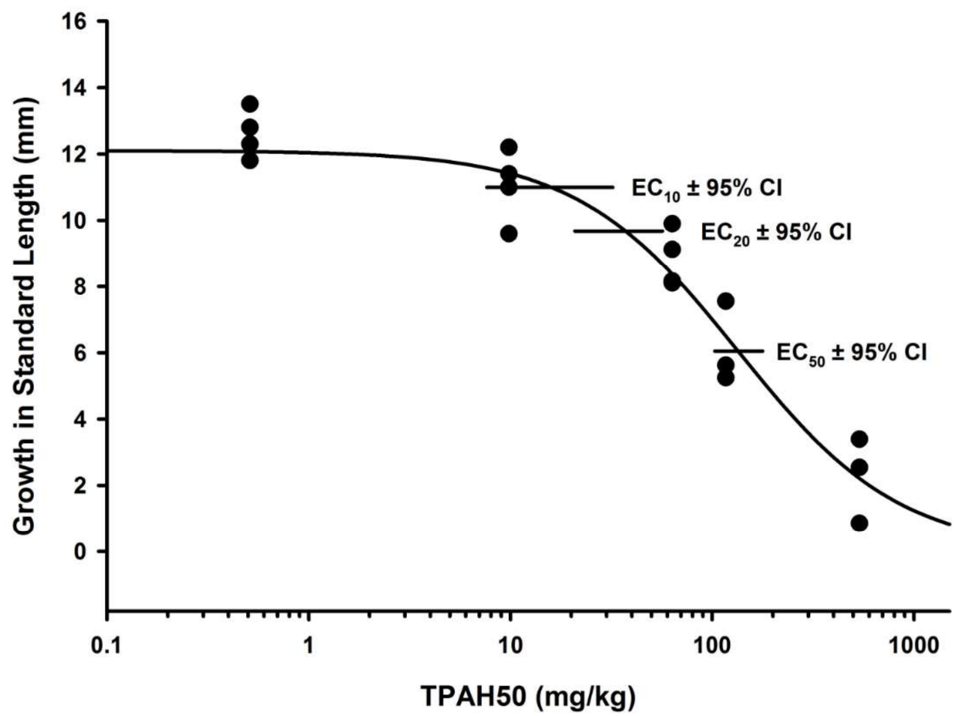
- Red drum are a long-lived predatory fish species in marsh/estuarine ecosystems, prized by recreational fishermen.
- TPAH50 concentrations at heavier persistently oiled Louisiana mainland herbaceous shorelines reduced juvenile red drum growth by up to 47 percent in 2010.
- An estimated total of 639 MT wet weight of red drum was lost where TPAH50 concentrations in marsh soils exceeded 31 parts per million. This effect occurred over 39 miles (62 kilometers) of oiled shoreline in 2011.
- Reduced red drum production persists through 2013 and would be expected to continue in heavier persistently oiled marshes until soil TPAH50 concentrations drop below 31 parts per million.

As a nearshore species, the red drum (*Sciaenops ocellatus*) is distributed over a wide range of habitats including estuaries, river mouths, bays, sandy bottoms, mud flats, sea grass beds, oyster reef, and surf zones. When adults reach 4 to 6 years old, they generally spawn near estuary inlets during late summer and fall (Wilson & Nieland 1994). After a brief planktonic period (4 to 6 weeks), currents carry young drum to estuaries and near-shore areas where they settle into structurally complex habitats like marshes for shelter and forage while they grow (Levin & Stunz 2005). When they were adjacent to marsh shorelines in the late summer and early fall of 2010, red drum were exposed to oil (Powers &

Scyphers 2015). As they mature (2 to 3 years), red drum tend to leave the close association with structured habitats and move into open coastal waters. Juvenile red drum eat small crustaceans and marine worms, and later, small fish; and they are eaten by birds of prey and larger fish.

Injury Determination

Oil stranded on marsh shorelines beginning in the early summer of 2010. By the fall of 2010, drum juveniles had settled adjacent to the marsh where they were exposed to oil. TPAH50 concentrations at heavier persistently oiled mainland herbaceous shorelines in Louisiana in 2010 reduced juvenile drum growth by 47 percent when compared to conditions where no shoreline oiling was observed (Figure 4.6-33) (Powers & Scyphers 2015). In 2013, heavier persistently oiled marsh conditions were still high enough to reduce drum growth by 21 percent. As part of the coastal wetland vegetation study (Hester et al. 2015) mentioned above, TPAH50 concentrations were measured in marsh surface soil samples taken at three distances from the vegetation edge between fall 2010 and fall 2013. The Trustees conducted laboratory studies to evaluate the effect of MC252 oil on juvenile drum growth over 13 days of exposure (Powers & Scyphers 2015). During these studies, 3.5 centimeter long juvenile drum were placed on sediments spiked with weathered MC252 oil over a range of TPAH50 concentrations representing those found at oiled marsh sites. Juvenile drum exposed to oil put on less weight and reached smaller sizes than fish exposed to clean sediment (Powers & Scyphers 2015).



Source: *Morris et al. (2015).*

Figure 4.6-33. Relationship between TPAH50 concentration in marsh soil and growth of juvenile red drum. As they are exposed to higher TPAH50 concentrations (horizontal axis), juvenile red drum grew less (vertical axis) than fish placed on clean sediment. At low TPAH50 concentrations, most fish grew about 12 millimeters longer over the 13-day study duration. At high concentrations, the drum that survived the test grew less than 4 millimeters longer.

Injury Quantification

The reduction in growth observed in red drum exposed to conditions at heavier persistently oiled marsh sites would translate into fewer adults since small fish suffer higher levels of predation. Reduced drum growth was converted to lost drum production (weight of adult equivalents) using literature values of juvenile drum densities measured in the marsh edge system (5 meters from the edge) and growth/survival relationships. Based on measured soil TPAH50 concentrations, growth reductions were predicted from toxicity test results. Concentrations in submerged sediment adjacent to the marsh edge were not high enough to be toxic to drum. However, the marsh edge ecosystem is inundated for extended periods of time, leading to exposure of red drum to high TPAH50 concentrations in the marsh soils (Powers & Scyphers 2015). Growth "penalties" were assessed for the period that juvenile drum show a high affinity for structured environments (6 months) and exhibit limited mobility. The approach used to calculate production losses is illustrated in Figure 4.6-34.

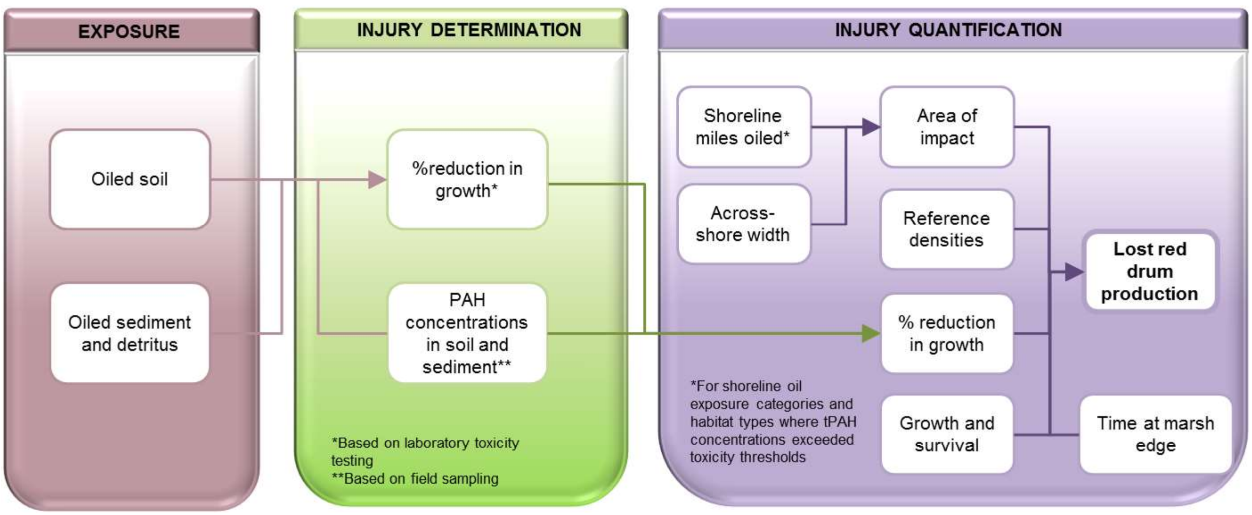


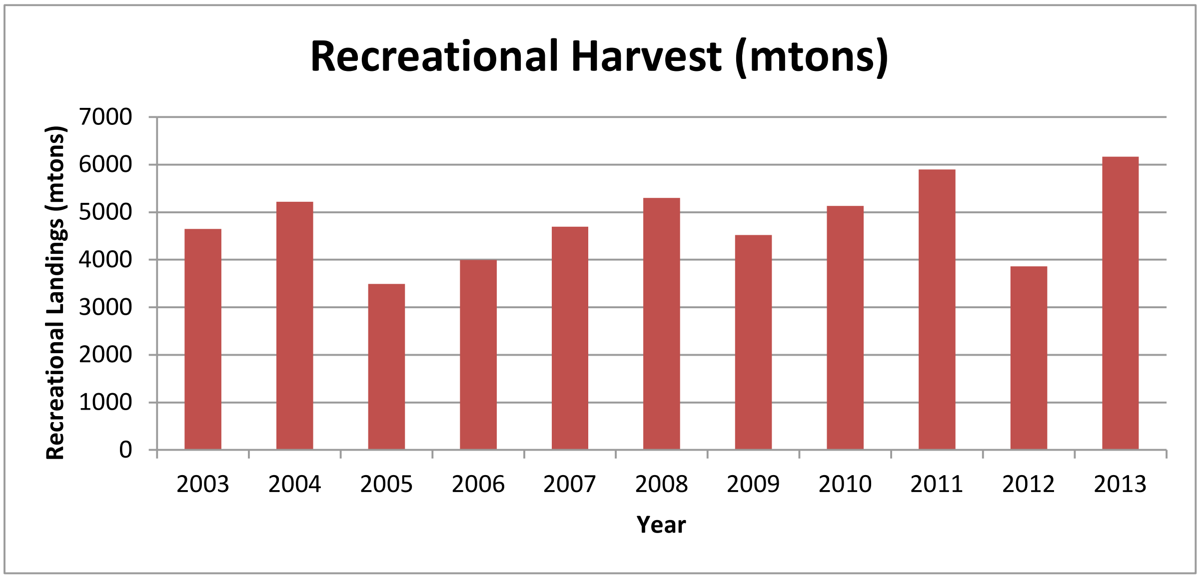
Figure 4.6-34. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to red drum occurred. TPAH50 concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-33) to determine the reduction in growth. The length and width of each zone and literature values for growth and production were used to calculate lost red drum production over the time that TPAH50 concentrations exceed toxic thresholds.

An estimated total of 639 MT wet weight of red drum was lost due to oiling of mainland herbaceous salt marsh in Louisiana where TPAH50 concentrations in marsh soils exceed 31 parts per million. This effect occurred over 39 miles (62 kilometers) of oiled mainland herbaceous shoreline in Louisiana between 2010 and 2013 (Table 4.6-16). Sources of uncertainty in this analysis include variations of concentrations of PAHs in each zone, variations in zone widths, variation in responses of the animals in the toxicity test, uncertainty in the length of shoreline miles oiled (Table 4.6-16), and uncertainties in baseline densities of drum, and growth, survival, and reproduction assumptions (Powers & Scyphers 2015).

Table 4.6-16. TPAH50 concentrations reduce growth to juvenile red drum over 39 miles (62 kilometers) of Louisiana mainland herbaceous marsh. The outlined box represents shoreline lengths where injury occurred.

LOUISIANA	Mainland Salt/Brackish Marsh		Back barrier Salt/Brackish Marsh		Delta/Inland Fresh/Intermediate Marsh		Mangrove/Marsh Complex	
Wetland Exposure Class	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)	Length (mi)	Length (km)
LIGHTER OILING	355	571	7	11	50	80	28	45
HEAVIER OILING	116	187	11	18	36	58	8	13
HEAVIER PERSISTENT OILING	39	62	0	0	3	5	3	5

Red drum are a critical predator in estuarine ecosystems and a highly targeted species in the recreational fishery (NMFS 2012). The importance of red drum to the recreational fishing community and the economy it supports cannot be overstated. Because red drum are long lived (more than 40 years) (Wilson & Nieland 1994), decreases in the production of the 2010, 2011, 2012, and 2013 year cohorts will reduce the number of fish appearing in the fishery for a period of 40 years. This reduction would be most notable during the first time period juvenile red drum become available for harvest at the age 2-3 years. For fish that were juveniles in the fall of 2010, this would be expected to occur in 2012. Notable, recreational landings of red drum in 2012 were anomalously low (Figure 4.6-35).



Source: *NOAA Fisheries Recreational Landings.*

Figure 4.6-35. Recreational harvest of red drum since 2003. Note that the harvest for 2012 is lower than immediately preceding years. 2–3 year old adult fish caught in 2012 would have been present in oiled marshes as juveniles during 2010.

4.6.4 Estuarine Coastal Wetlands Complex Injury Assessment

4.6.4.5.7 Fiddler Crabs

Key Points

- Fiddler crabs are highly abundant marsh residents that greatly influence ecological marsh processes through their burrowing and feed activities.
- Shoreline oiling reduced fiddler crab burrow density by at least 25 percent and burrow diameter by 21 to 44 percent when compared to sites that were not oiled.
- Reductions in number and size of fiddler crab burrows occurred over the length of shoreline where oiling conditions were heaviest, and indicate reduced fiddler crab numbers and biomass.
- Oiling caused a shift in fiddler crab species composition, which would result in lower fiddler crab biomass and a change in the prey base for predators.

Fiddler crabs are prolific burrowers in marsh substrates and process large amounts of sediments and organic material during feeding (Bertness 1985; Montague 1982). Through their burrowing and feeding activities, fiddler crabs greatly influence ecological marsh processes by potentially modifying marsh vegetation, sediments, organic material, nutrient cycling, microbial communities, and meiofauna (Bertness 1985; Hoffman et al. 1984; Montague 1982). In salt marshes, fiddler crab burrows can increase soil drainage, soil oxidation-reduction potential, and decomposition of belowground biomass, thereby indirectly increasing plant biomass (Bertness 1985; Montague 1982). Fiddler crabs are important prey items for larger macroinvertebrates, fish, reptiles, birds, and mammals that use salt marsh and mangrove habitats (Daiber 1982; Grimes et al. 1989). Crabs, including fiddler crabs, have sustained significant adverse effects years after exposure to oil, due to direct toxicity, smothering, and limited access to the marsh surface (Burger et al. 1991, 1992; Culbertson et al. 2007; Krebs & Burns 1977; Michel & Rutherford 2013).

Injury Determination

Shoreline oiling is estimated to have reduced fiddler crab burrow density by at least 25 percent and burrow diameter by 21 to 44 percent when compared to sites that were not oiled (Zengel et al. 2015c). Oiling also caused a shift in fiddler crab species composition from mostly *Uca longisignalis* to mixtures of *Uca longisignalis* and *U. spinicarpa*. This shift in species composition was likely mediated by loss of vegetation due to oiling, because *U. spinicarpa* is normally associated with less-vegetated habitats (Zengel et al. 2015c). To reach these conclusions, Trustees compiled results from four field studies that examined fiddler crab burrow density, burrow diameter, and species composition between 2010 and 2013. Study sites were located in Louisiana mainland herbaceous salt marshes dominated by *Spartina alterniflora* and included marsh edge sites and sites located in interior zones (Zengel et al. 2015c).

Injury Quantification

While reductions in burrow density and size were observed between 2010 and 2013, the number and production of lost fiddler crabs were not estimated due to differences in design parameters between the studies compiled for the analysis. Reductions in number and size of fiddler crab burrows occurred over the length of shoreline where oiling conditions were heaviest, and indicate reduced fiddler crab numbers and biomass. The change in fiddler crab species composition would also result in lower fiddler crab biomass, as *U. spinicarpa* is the smaller of the two species (Zengel et al. 2015c). The shift in fiddler crab species composition would be expected to persist as long as changes in vegetation persist and were observed throughout the study period of 2010 to 2013 (Zengel et al. 2015c).

4.6.4.5.8 Insects and Spiders

Key Points

- Insects and spiders are important members of the coastal food web as they provide food for a variety of species, including birds, fish, and amphibians.
- A decrease in terrestrial arthropod abundance and a shift in species composition was detected in oiled Louisiana coastal wetland marsh.
- Terrestrial arthropods were likely killed by direct oiling, toxic effect of chemicals in the oil, or a decrease in suitable habitat.

Insects and spiders (terrestrial arthropods) are an important component of the marsh ecosystem. They occur at high densities and are rich in species diversity (Denno et al. 2005; Wimp et al. 2010). Approximately 100 documented species of terrestrial arthropods are associated with coastal salt marshes dominated by *Spartina alterniflora* (McCall & Pennings 2012; Wimp et al. 2010). Types of arthropods that live in *Spartina* marsh habitats include predators (spiders and ants), pollinators (bees), parasites (wasps), herbivores (katydids and seed bugs), and detritivores (springtails). Insects and spiders are also important members of the coastal food web as they provide food for a variety of species, including birds, fish, and amphibians (Wimp et al. 2010). Herbivores and omnivores play a large part in the community because they both support higher trophic levels in the food web and can regulate plant community structure (Jiménez et al. 2012). Populations of herbivorous insects found in coastal marsh communities such as planthoppers can greatly influence plant community structure (Jiménez et al. 2012). Springtails, which are also found in the *Spartina* marsh, play a major role in the soil ecosystem by contributing to the decomposition of plant litter (Bardgett & Chan 1999; Rusek 1998). Terrestrial arthropods were likely killed by direct oiling, toxic effect of chemicals in the oil, or a decrease in suitable habitat (Pennings et al. 2014).

Injury Determination

Terrestrial arthropod communities were affected by oiling in the marsh as a result of the *Deepwater Horizon* oil spill. McCall and Pennings (2012) surveyed insect communities in unoiled and oiled sites in *Spartina* marsh. At the oiled sites, investigators surveyed insects within oiled—but still living—vegetation. If a heavily oiled, dead patch of vegetation was present at a given site, investigators walked

1 to 2 meters into the adjacent live vegetation from the edge of the dead patch before collecting insects. The investigators found that in 2010, the total surveyed terrestrial arthropod community was suppressed by 50 percent in oiled sites, relative to their reference sites. In 2011, the total surveyed insect communities adjacent to heavily oiled sites appeared to be similar to those of reference sites (McCall & Pennings 2012). This study shows that even in areas with living vegetation, oiling had an effect on adjacent terrestrial insect communities. Because the vegetation was not severely oiled in the study sites, terrestrial insect communities may have recovered more quickly than they would have in areas where the vegetation was severely impacted by oiling (i.e., within the heavily oiled dead vegetation patches that McCall and Pennings (2012) did not survey). Indeed, as Pennings observed: “Because terrestrial arthropods appear to be more sensitive to oil exposure than salt marsh plants, many scenarios of oil exposure could create salt marshes that appear healthy to the casual observer but that are, in fact, devoid of terrestrial arthropods and the ecosystem functions that they support” (Pennings et al. 2014).

Hooper-Bui et al. (2012) and Soderstrum et al. (2012) also performed insect surveys in unoiled and heavily oiled locations in Louisiana. Similarly, they found a decrease in terrestrial arthropod abundance in oiled sites compared to unoiled sites in 2010 and 2011. Further, in 2012, Hooper-Bui et al. (2012) observed a complete absence of mature ant colonies in heavily oiled areas, whereas similar areas with “no oil observed” had many colonies present.

Injury Quantification

Oil reduced abundance of insects using marshes (especially ants). While a formal quantification of total loss of insect production or miles of affected habitat is not possible at this time, it would nevertheless be expected that similar effects on insects and spiders would occur in other similarly oiled *Spartina* salt marshes (Hooper-Bui et al. 2012).

4.6.4.5.9 Nearshore Oysters

Key Points

- Nearshore oysters provide refuge to marine life through reef formation and play an important ecological role in stabilizing marsh shorelines.
- Shoreline oiling and cleanup actions significantly reduced the presence of nearshore oysters in the adjacent intertidal zone over approximately 155 miles (250 kilometers).
- An estimated 8.3 million adult equivalent oysters were lost due to marsh oiling along shorelines where oyster cover was removed by oiling or cleanup actions. An additional estimated 5.7 million oysters per year (adult equivalents) are unable to settle because of the loss of oyster shell cover. Had these oysters not been killed, they would have produced a total of 1.3 million pounds of oyster meat (wet weight) over their 5-year lifespans. Recovery of these nearshore oysters is not expected to occur without intervention or restoration actions.

An important sub-population of oysters (*Crassostrea virginica*) inhabits the nearshore zone fringing the marsh edge and forming emergent reefs or smaller hummocks. These nearshore oysters, like their

subtidal counterparts, play an important ecological role through their filtration activities with critical benefits for nutrient cycling (Powers et al. 2015b). Oysters remove sediments, phytoplankton, and detrital particles, potentially reducing turbidity and improving water quality (Dame & Patten 1981). Because they are not harvested, nearshore oysters provide a stable source of larvae to oysters in deeper waters (Murray et al. 2015). The complex habitat formed by the gregarious settlement of oysters also provides critical refuge for benthic invertebrates, fishes, and mobile crustaceans (Coen et al. 2007; Meyer & Townsend 2000; Peterson et al. 2003a). Nearshore oyster reefs can also reduce erosion and stabilize coastal shorelines through sediment trapping and accretion and adding hard substrate adjacent to marsh edges (Meyer et al. 1997; Piazza et al. 2005; Scyphers et al. 2011).

Injury Determination

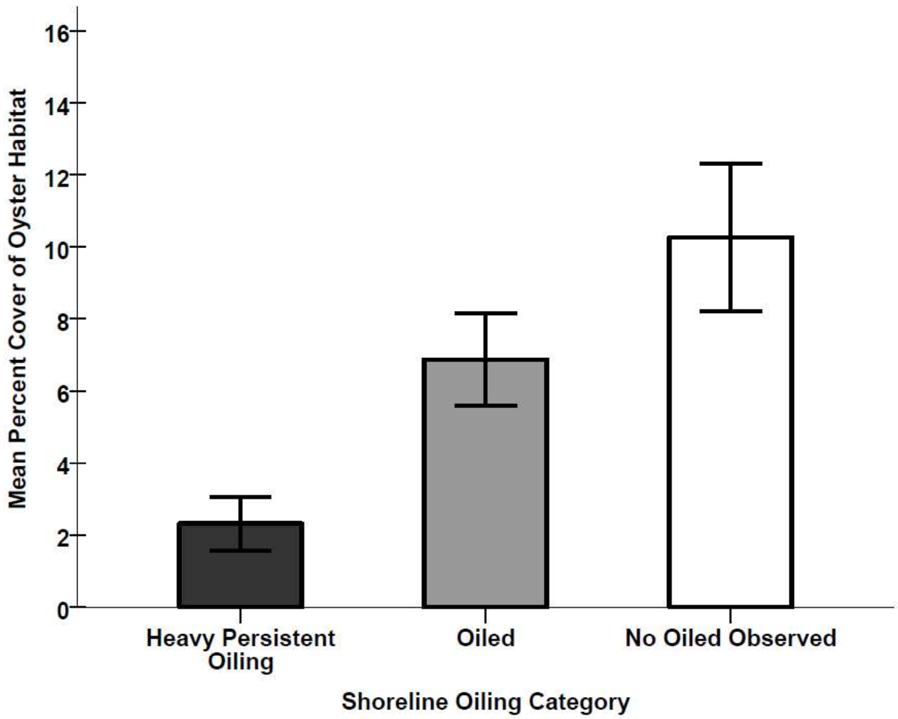
Shoreline oiling and related cleanup actions significantly reduced cover of fringing oysters within 50 meters of marsh shorelines (Powers et al. 2015b). Lowest percent cover values were recorded in areas adjacent to marshes that experienced heavy persistent oiling (2.3 ± 0.7 percent), followed by areas that experienced more moderate and less persistent oiling (6.9 ± 1.3 percent) and reference shorelines (10.3 ± 2.1 percent) (Figure 4.6-36). The proportion of sites with no oysters (with percent cover of oyster habitat <0.5 percent) was also highest adjacent to marshes that experienced heavy persistent oiling (56 percent), followed by lesser degrees of oiling (43 percent) and reference sites (24 percent).

Cleanup activities (e.g., raking, washing, or laying oil boom adjacent to the marsh) also affected percent cover of oyster habitat. For oiled sites with documented onsite or nearby cleanup activities, percent cover was significantly lower than oiled areas that did not have cleanup actions within 328 feet (100 meters) of sampling sites. The mean oyster percent cover at treated sites was 4.1 ± 0.9 percent compared to 10.1 ± 2.8 percent at untreated sites (Figure 4.6-37). The injury resulting from the *Deepwater Horizon* oil spill in the summer of 2010 was largely a function of an acute disturbance that occurred during or within a year after the oil spill (assuming approximately 2 years for oyster growth from spat to market size). By destroying oyster cover through smothering or through physical destruction during cleanup activity, much less shell and hard surface remained for future larvae to settle on. Although the disturbance was relatively short-lived, the consequences of the losses to oysters are substantial: they have no to very little predicted natural recovery, and include loss of essential fish habitat, reduced nutrient cycling, and decreased shoreline stability (Powers et al. 2015b).

These findings are the result of NRDA field studies conducted in 2012 and 2013 to determine whether percent cover of nearshore oyster habitat and oyster abundance differed as a function of shoreline oiling and cleanup activities (Powers et al. 2015b). Overall, 187 sites from Terrebonne Bay, Louisiana, to Mississippi Sound, Alabama, were sampled in 2013 after a pilot effort in 2012. Sites (200-meter long stretches of shoreline) were classified by shoreline oiling classification and mapped to estimate oyster cover, as indicated by the presence of shell substrate. For the purposes of evaluating nearshore oysters, the four shoreline oiling categories described in Zachary Nixon et al. (2015b) were reduced to three: heavy persistent oiling, oiled, and reference ("no oil observed").[4] The heavier and lighter oiling

[4] "No oil observed" is a shoreline category intended to describe areas where oiling was not observed during linear shoreline surveys. The SCAT survey and NRDA rapid assessment survey were the primary datasets used to inform the oiling categories

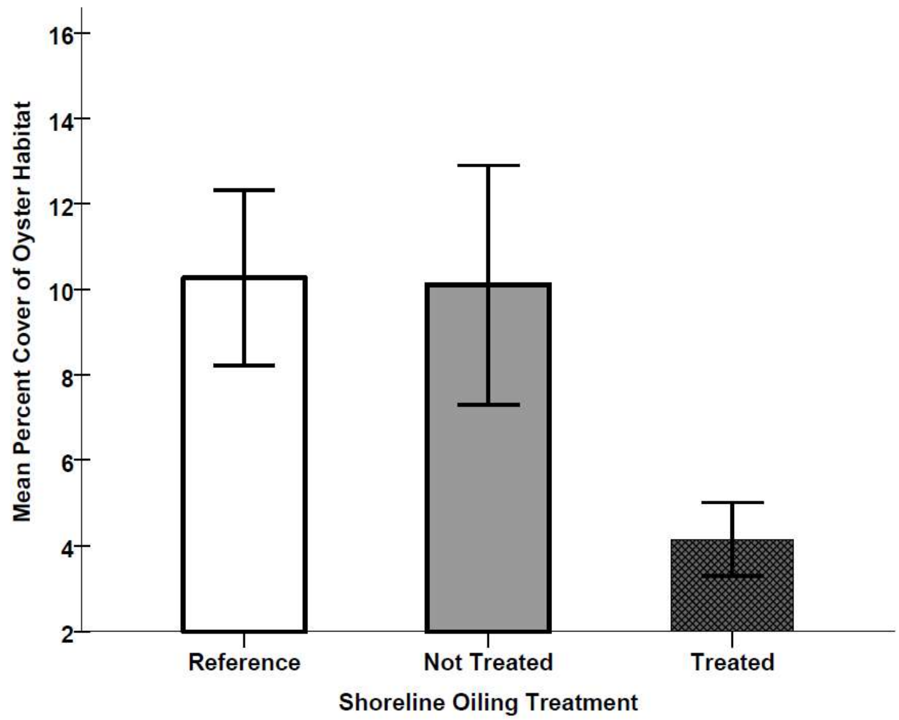
categories were combined into an "oiled" category to distinguish effects of heavy persistent oiling, such as heavy fouling and smothering, from those sites that experienced more subtle oiling effects. In addition to cover measurements, abundance of three life stages of oysters was measured at multiple locations at each site. Some sites sampled in this study were in areas affected by river water releases (Powers et al. 2015b).



Source: *Powers et al. (2015b).*

Figure 4.6-36. Percent cover of oyster by oiling category (mean ± 1 standard error) from Terrebonne Bay, Louisiana, to Mississippi Sound, Alabama. This figure demonstrates the effect of oiling on nearshore oysters. Oiled areas had lower oyster cover (percent of area) than non-oiled areas. Areas that experienced heavier persistent oiling had the lowest observed oyster cover.

and estimate oiled shoreline miles for evaluating exposure to wetland and beach animals. If neither survey detected oil in an area, that area is described as "no oil observed." However, in some instances, oil came ashore after a segment was surveyed. Other field sampling events later found oiling in some of these areas designated "no oil observed," and some areas likely experienced oil that was never detected.



Source: *Powers et al. (2015b).*

Figure 4.6-37. Relationship between oyster cover and whether shoreline cleanup occurred near the site. This figure shows that areas treated by cleanup activities ("Treated" category), had much less oyster cover (mean percent cover ± 1 standard error) than in areas that were oiled but not treated by cleanup actions ("Not Treated" category). Percent cover of oyster habitat at "Reference" sites (where cleanup activities were not prescribed) is also much greater than at "Treated" sites.

Injury Quantification

Nearshore oyster cover was dramatically reduced over an estimated total of 155 miles (250 kilometers) of shoreline (Roman 2015). This is calculated from the length of northern Gulf marsh shoreline where oiling and shoreline cleanup actions occurred and the proportion of unoiled sites where oyster cover was detected. Reduction of oyster cover along this shoreline translates directly into fewer adult oysters that would be produced over time adjacent to marsh habitats. Though these nearshore oysters are not harvested, their loss eliminates myriad services to humans and the ecosystem, including as a source of larvae for adjacent subtidal—and harvestable—oyster beds (Roman 2015).

Reduced oyster cover was converted to lost production of adult oyster equivalents using percent cover and density measurements of each size class from unoiled areas, literature values on survival and growth to the adult stage, and a calculation of the total area over which oyster cover was reduced. Injury was calculated for an area that includes a 50-meter wide swath of nearshore sediment adjacent to oiled shorelines; however, the bulk of fringing oyster cover is located within 3 meters of the marsh edge (Roman 2015). In addition to the oysters killed by oil or response actions, the loss of oyster shell cover prevents future larvae from settling in this area. Using estimates of post-spill settlement at reference locations from the NRDA sampling, the Trustees estimated the loss of juvenile oysters that would have been expected to settle on the lost shell (Roman 2015).

An estimated total of 8.3 million adult equivalent oysters were lost due to marsh oiling along shorelines where oyster cover was removed or reduced by oiling or cleanup actions (Roman 2015). The number of adult equivalent oysters lost was calculated by adding up numbers killed in each size class and adjusting spat and seed numbers for the proportion that would have been expected to survive to adults. These oysters, had they not been killed, would have produced a total of 1.3 million pounds of oyster meat over their 5-year lifespan. Approximately 40 percent of that total represents an estimate of the weight of the oysters directly killed, and the remaining 60 percent represents additional growth of adult oysters over the rest of their lifespan that did not occur because they were killed. This loss occurred between the time that oil reached the shorelines and the time when the majority of cleanup activities were completed (by the end of 2011) (Roman 2015). The loss of oyster shell cover also means that an estimated 5.7 million oysters per year (adult equivalents) would be unable to settle and grow in nearshore areas.

The approach to calculating these losses is illustrated in Figure 4.6-38. Uncertainty in this analysis comes from variation in oyster cover and abundance measured in the field, uncertainty in the number of shoreline miles oiled, and uncertainty in literature-based assumptions about growth and survival between juvenile and adult life stages. Recovery of these oysters is not expected to occur without intervention or restoration actions.

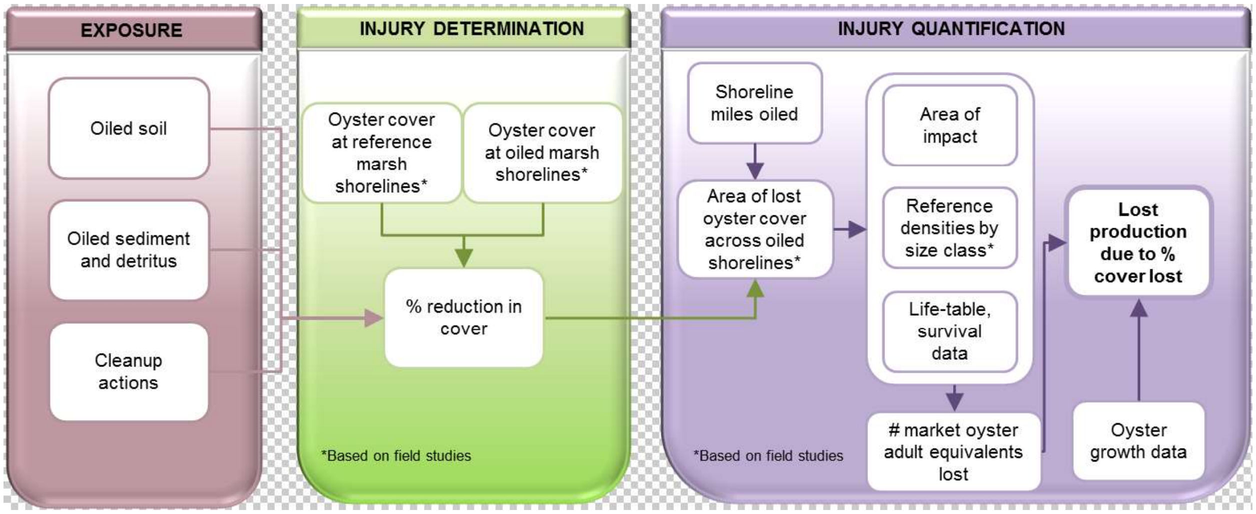


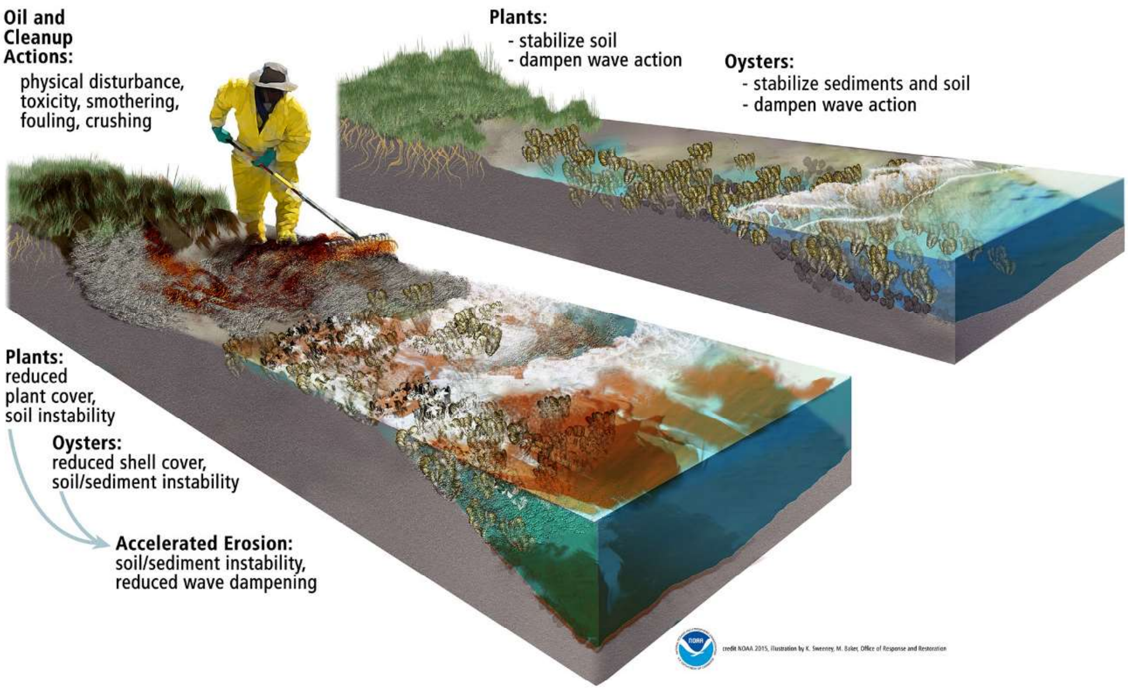
Figure 4.6-38. Illustration of information used to determine how injury to nearshore oysters occurred. Oyster cover in nearshore environments was mapped at oiled sites and unoiled (or reference) sites during field studies and extrapolated to the entire vegetated shoreline to determine the total amount of oyster cover lost due to a combination of shoreline oiling and cleanup actions. Field measurements of abundance of oysters in each size class were used to calculate how many oysters would have been lost per square meter. Growth and survival information from other studies were combined with numbers in each size class to convert to numbers of adult equivalent oysters. These numbers were converted to lost production over time using a mean growth foregone rate of 1.27 grams ash-free dry weight per 75 millimeter oyster.

4.6.4.6 Shoreline Erosion

Key Points

- Multiple studies demonstrated that the *Deepwater Horizon* spill resulted in increased rates of coastal erosion.
- Erosion rates approximately doubled along at least 108 miles (174 kilometers) of shoreline over at least 3 years.
- Wetland loss due to erosion cannot recover naturally.

Many factors have contributed to coastal wetland loss in Louisiana over the last 50 years. To explore relationships between the *Deepwater Horizon* spill and erosion of wetland shorelines over a much shorter time span (since 2010), the Trustees focused on evaluations of environmental factors that were affected by oiling or cleanup actions. Figure 4.6-39 illustrates potential mechanisms for oil and cleanup actions to enhance erosion of wetland shorelines. Shoreline oiling has been demonstrated to reduce wetland plant cover (Hester et al. 2015) and nearshore oyster cover (Powers et al. 2015b), and has been associated in prior studies with enhanced shoreline erosion (McClenachan et al. 2013; Silliman et al. 2012). Physical disturbance associated with oil spill cleanup actions has the potential to disrupt plant cover, soil stability, and nearshore oyster cover, which all could contribute to enhanced wetland shoreline erosion.



Source: *Kate Sweeney for NOAA.*

Figure 4.6-39. Trustees explored relationships between shoreline oiling, cleanup actions, plant oiling, oyster cover, and erosion. Mechanisms that could enhance erosion as a result of spill impacts include a loss of soil stability (which could be affected by loss of plant cover, physical trampling, or other disturbance) and changes to the roughness of the bottom adjacent to the shoreline (for example, through the loss of oyster cover) which could increase wave energy over small spatial scales.

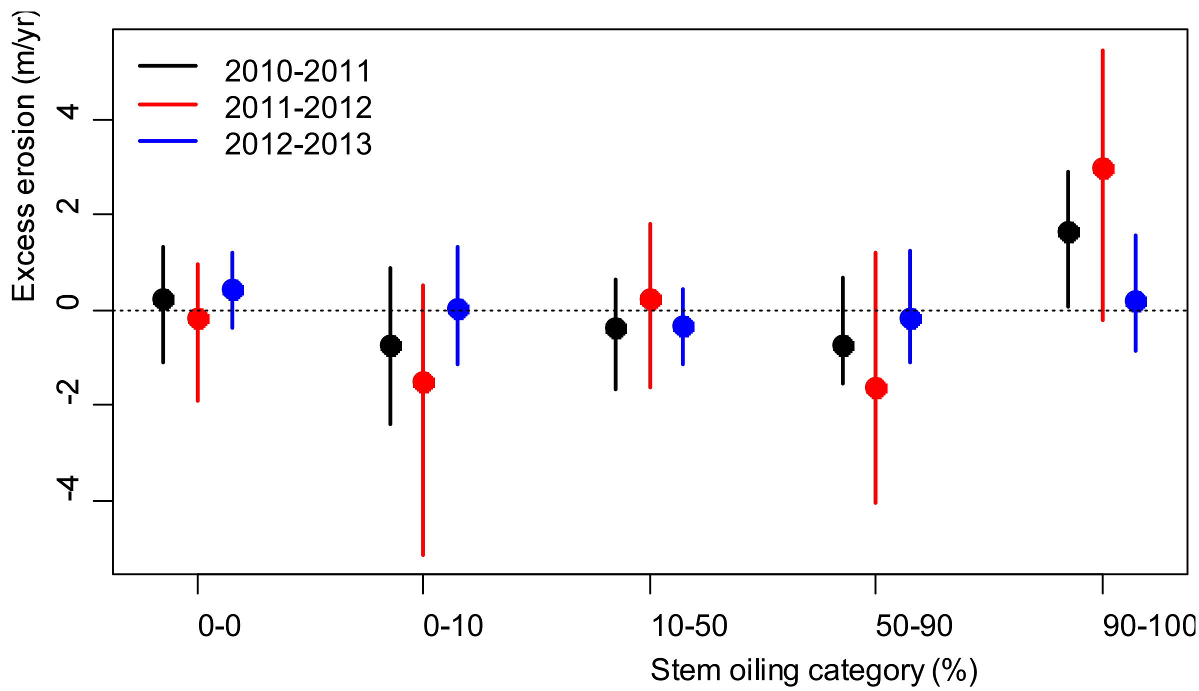
4.6.4.6.1 Injury Determination

In coastal Louisiana, subsidence, storm events, wind driven waves, human activities, and other factors contribute to a high rate of baseline erosion and can confound an assessment of the contribution of oiling or cleanup to erosion (Britsch & Dunbar 1993; Couvillion et al. 2011; Penland et al. 2001). To establish the relationship between oil, shoreline cleanup, and erosion, and to account for other factors that contribute to erosion in the region, various multivariate analyses and statistical approaches were conducted using field measurements and other available data. Using three different approaches—high resolution aerial image analysis of paired study sites, field measurements collected as part of the Coastal Wetland Vegetation (CWV) study, and assessments of nearshore oyster cover—the Trustees determined that oil and associated cleanup actions contributed to increased erosion in coastal wetlands.

The analysis of paired images compared treated, heavily oiled coastal wetland sites to untreated less oiled coastal wetland sites in Barataria Bay. This analysis found an increase in erosion at oiled, treated sites of 0.41 meter/year from the fall of 2010 to the spring of 2013 (Gibeaut et al. 2015). Paired sites were selected to account for factors such as wave energy and wind direction. Cleanup actions at these sites included approaches to remove or reduce oil and oiled debris and approaches to speed the weathering and degradation of residual oiling in order to accelerate coastal wetland recovery (Zengel et

al. 2015a). The heavy persistent oiled sites retreated at a rate of 1.36 meter/year, and the control sites retreated at a rate of 0.94 meter/year. Comparison with decadal-scale shoreline change analysis serves to place the current, short-term changes in context (Gibeaut et al. 2015). The long-term retreat rate is statistically indistinguishable from the current study's control sites, but the impact sites have rates statistically higher (Gibeaut et al. 2015).

The next study looked at the relationship between erosion and a broader range of oiling conditions over a larger geographic area within Louisiana (Silliman et al. 2015). Trustees conducted a number of analyses to evaluate the relationship between vegetation oiling (as measured by stem oiling) and erosion. Using the stem oiling categories developed for the CWV study, the Trustees compared erosion rates at 77 sites in coastal Louisiana mainland herbaceous marshes, which is the habitat at highest risk of erosion. These 77 sites were divided between five stem oiling categories (0 percent, 0-10 percent, 10-50 percent, 50-90 percent, and 90-100 percent). Cleanup did not occur at any of these sites. These sites included a range of wave exposures (Nixon 2015) to ensure that the analysis of erosion injuries considered the influence of wave energy on shoreline loss. The Trustees compared the average erosion for each stem oiling category during three 1-year periods (2010 to 2011; 2011 to 2012; and 2012 to 2013), factoring in the distribution of oiling categories over similar ranges of wave energy. The results (Figure 4.6-40) indicate that erosion rates approximately doubled in the more highly oiled locations (90-100 percent stem oiling) relative to sites with no stem oiling (Snedaker et al. 1996). After adjusting for the influence of wave energy, the erosion rate in sites with 90-100 percent stem oiling was 1.6 m/yr larger than expected (Silliman et al. 2015). The Trustees' analysis, in conjunction with the published studies of Silliman et al. (2012) and Lin et al. (2014), demonstrate that marsh oiling resulted in increased rates of erosion and loss of Louisiana coastal salt marsh.

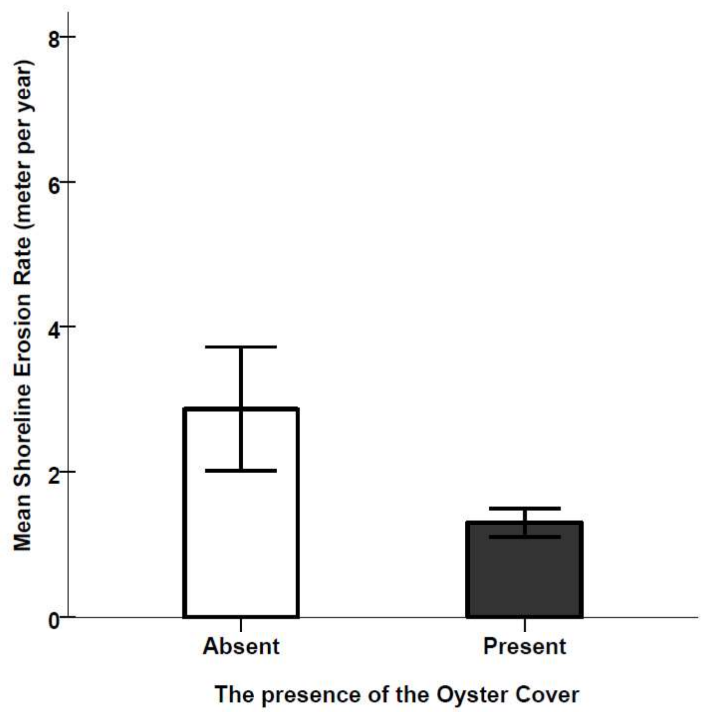


Source: *Silliman et al. (2015).*

Figure 4.6-40. Average excess erosion rate (meters/year) for each stem oiling category in 2010-2011, 2011–2012 and 2012–2013 for all Louisiana CWV sites. Excess erosion is the difference between the average erosion rate for each group of sites and the expected erosion rate based on the wave energy for that group of sites. The vertical lines are the central 95 percent randomization distributions for excess erosion in each stem oiling category. When the vertical line does not cross 0, the p-value for the comparison of that stem oiling category to the overall erosion rate is less than 0.05. These data demonstrate that (1) the erosion rates are highest in the 90-100 percent category for the first 2 years (2010-2011 and 2011-2012) and (2) those erosion rates are higher than expected for the range of wave energies for those sites.

In addition to directly contributing to elevation gain of marshes and bay bottoms through growth, shell production, and feces/pseudofeces production, oyster reefs also reduce shoreline erosion (Bahr & Lanier 1981). Shoreline erosion may be reduced by two mechanisms: the direct reduction in wave height and water current velocities by the friction of the oyster reef's rough, elevated rigid structure and through the trapping of sediment and stabilization of marsh edge substrate. The analysis of injury to nearshore oysters found that shoreline loss was more than twice as high (2.9 versus 1.3 meters/year) in areas lacking nearshore oyster cover (a difference of 1.6 meters/year) (Powers et al. 2015b). Overall, 79 nearshore oyster sampling stations were co-located with sites in Alabama, Mississippi, and Louisiana that were included in an evaluation of coastal wetland vegetation; and synoptic data on shoreline erosion were collected at these sites. Because so little was known about pre-spill nearshore oyster distribution, no pre-existing information was available about the likelihood of finding oyster cover at these 79 sites, though they were all located within habitats and salinity conditions thought to be supportive of oyster presence. Confirmation that oysters would have been present in oiled areas comes from the distribution of oyster cover at reference (unoiled) sites and the fact that oiled and unoiled sites are similar in the factors necessary to support oysters and other factors that could enhance erosion (Powers et al. 2015b).

The presence of nearshore oyster habitat was associated with significantly reduced shoreline erosion in the adjacent marsh (Powers et al. 2015b). At two of the sites, erosion over the 3-year period was extremely high. If those sites are considered to be statistical outliers i.e., more than two standard deviations above the mean) and removed from the analysis, the rate of excess erosion drops to 0.6 meters per year (Figure 4.6-41). The degradation and loss of nearshore oyster habitat resulting from shoreline oiling and associated treatment activities can disrupt strong facilitation between oysters and marsh vegetation and demonstrates a previously unreported ecosystem level consequence of oil spills (Powers et al. 2015b).



Source: *Powers et al. (2015b).*

Figure 4.6-41. This figure compares shoreline erosion rates over a 4-year period at sites where oysters were present versus absent (mean ± 1 weighted standard error). The erosion rate was significantly higher in areas where oysters were not present.

4.6.4.6.2 Injury Quantification

The analysis of paired sites that represent a combination of heavy persistent oiling and shoreline cleanup actions indicates that a higher erosion rate due to shoreline oiling and treatment occurred over at least 6 miles (10 kilometers) of heavy persistently oiled shoreline in Barataria Bay over 2.6 years (Gibeaut et al. 2015). Uncertainty in this analysis comes from variability within sites in amount of erosion observed and the length of shoreline over which the effect can be estimated. This method only analyzed sites that were heavily persistently oiled and treated, and therefore applies to a relatively small length of shoreline.

The relationship between plant oiling and erosion indicates that accelerated erosion occurred over 47 to 67 miles (76 to 109 kilometers) of shoreline where Louisiana mainland herbaceous salt marsh vegetation

experienced greater than 90 percent plant oiling between 2010 and 2013 (Silliman et al. 2015). Uncertainty in this estimate comes from variability in observations of transect lengths over time and the length of shoreline over which the effect can be estimated. The dataset used for this analysis is applicable only to Louisiana mainland herbaceous salt marsh, but it is possible that the effect would have been observed in other habitats if more sites had been sampled.

Decreased cover of intertidal oysters is associated with increased rates of erosion (Powers et al. 2015b). This effect was observed throughout the area sampled (i.e., Louisiana, Mississippi, and Alabama). This analysis indicates that excess erosion attributable to oiling, cleanup actions, and resulting destruction of nearshore oyster cover occurred over 108 miles (174 kilometers) of oiled coastal wetlands between 2010 and 2013 (Roman 2015). This effect was observed along all oiled shorelines in Alabama, Mississippi, and Louisiana. Uncertainty in this analysis is associated with variations in oyster cover measurements, observations in transect lengths over time, and uncertainty in estimates of the lengths of oiled shoreline.

In summary, erosion attributable to the *Deepwater Horizon* spill approximately doubled along at least 108 miles (174 kilometers) of coastal wetlands over at least 3 years (Roman 2015). While all three studies evaluated erosion in Louisiana, it is not possible to determine how much overlap or separation there is between conditions at the sites in the three different analyses. Eroded areas will not recover naturally, as land mass is permanently lost (Powers et al. 2015b).

4.6.4.7 Wetland Response Injury

Key Points

- More than 497 miles (800 kilometers) of boom was stranded in marshes, injuring vegetation and birds.
- Removal of stranded boom also affected the wetlands. Vegetation was crushed by airboats, walking boards, foot traffic, and dragging of the boom across the wetland surface.
- The footprint of stranded boom totaled approximately 52 acres (210,000 square meters), which does not include the greater area of wetland swept by the boom when moved by storm waves.

4.6.4.7.1 Injury Determination

As described in Chapter 2 (Incident Description), several different types of response activities took place in the marsh. The Trustees evaluated the impacts of such activities on accelerated erosion (see Section 4.6.4.6). Here, the Trustees analyze marsh impacts as a result of boom that was deployed in the water, displaced due to wave and storm action, and stranded upon the marsh. Hard and soft boom stranded in various ways on the shoreline, depending on boom type, water levels, wave conditions, and shoreline type and slope (Michel & Nixon 2015). Along some shorelines, boom was pushed deep into the wetland, crushing and breaking vegetation as it swept across the wetland platform and eventually settled.

Figure 4.6-42 shows examples of the harm caused by stranded boom. In many locations, the impact of boom on vegetation was pronounced enough to be seen in aerial imagery. For example, Figure 4.6-43 shows a location with stranded boom in wetlands in July 2010, and the same location in October 2010.

Comparison of the images shows visible scarring of wetland habitat with dead vegetation where the boom had been. Boom also stranded on shell berms, which are important bird-nesting habitats; and this boom stranding potentially damaged nests, crushed eggs and chicks, and limited the re-occupation of these areas for nesting post-storm (Figure 4.6-42). Stranded boom also acted as physical barriers to animals that must move between areas landward of the boom and the water line to feed, access water, and escape from predators. In some locations, boom had the opposite effect as intended: rather than protecting the marsh edge from oil, the boom trapped oil up against marsh (Michel & Nixon 2015).

Removal of stranded boom caused wetland impacts. Vegetation was crushed by airboats, walking boards, foot traffic, and dragging of the boom across the wetland surface (Michel & Nixon 2015). Even under the best of conditions, boats operating at the wetland edge will harm this fragile, highly erosional platform because (1) the boats will bounce up and down with wave action and (2) crews will constantly get on and off. Often, grappling hooks were thrown to attach to stranded boom within throwing distance. The grappling hooks would gouge the wetland surface and uproot plants during missed throws and failed attempts to get the boom to the boat (Michel & Nixon 2015).

Source: *NOAA Deepwater Horizon SCAT Program.*

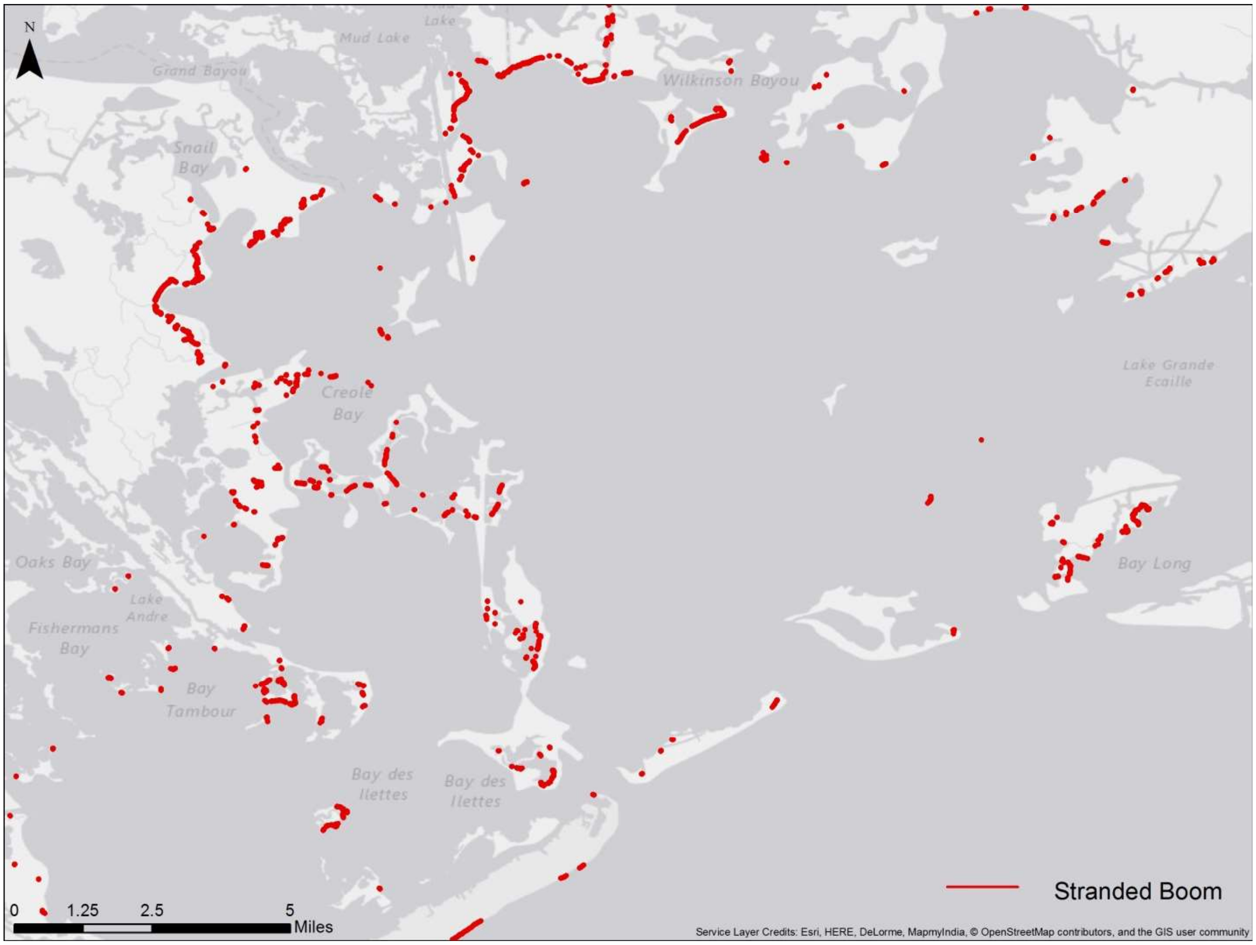
Figure 4.6-42. Stranded boom on wetlands in St Bernard and Plaquemines Parish, July and August 2010. Vegetation was crushed under the boom.

Sources: *Left image: NAIP (2010) Data available from the U.S. Geological Survey. Right image: ERMA (2015).*

Figure 4.6-43. Impacts of stranded boom. Left photograph: Stranded boom digitized on aerial imagery, July 2010, Barataria Bay, Louisiana. Right photograph: Visible scarring after boom was removed, October 2010.

4.6.4.7.2 Injury Quantification

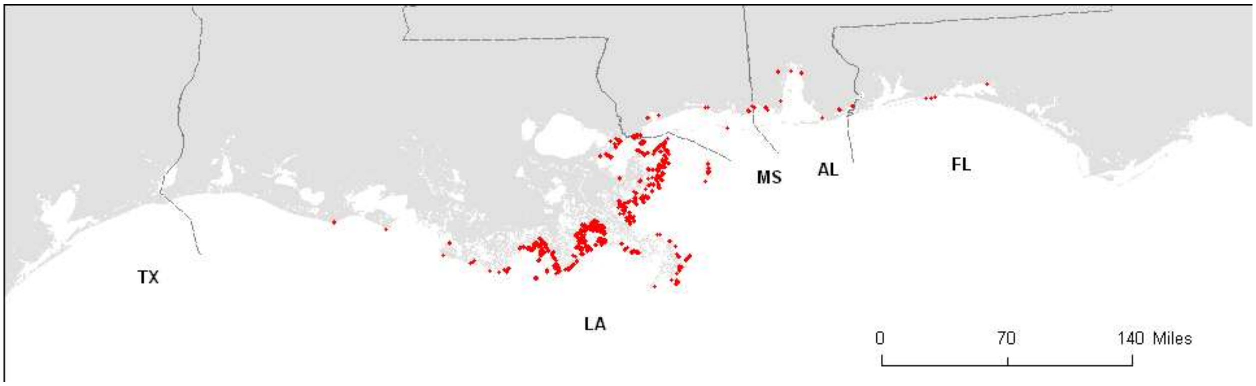
As an illustrative example of the extent of boom stranded in wetlands, the Trustees digitized stranded boom in Barataria Bay from a set of aerial images collected during overflights conducted in late July 2010 (ERMA 2015). For just this one snapshot in time, approximately 19 miles of boom were identified as stranded in Barataria Bay wetland habitat (Figure 4.6-44).



Source: *ERMA (2015).*

Figure 4.6-44. Stranded boom in Barataria Bay, in late July 2010. Stranded boom was digitized from National Agriculture Imagery Program (NAIP 2010) aerial imagery.

Based on a review of different response records, the Trustees determined that upwards of 497 miles (800 kilometers) of boom was stranded in wetlands (Figure 4.6-45) (Michel & Nixon 2015). This amount represents roughly 20 percent of the total boom deployed during response (USCG 2011). The vast majority of the deployed boom was in Louisiana. Based on the length of stranded boom and its average width, the footprint of this boom totaled approximately 52 acres (210,000 square meters). This footprint underestimates the area of injury because it does not include the area of wetland swept by the boom as the boom washed into the wetland and was moved by storm waves (Michel & Nixon 2015). The quantified footprint as well as the area swept by the boom represents injury to wetland vegetation and associated fauna. The boom area also represents injury to nesting and loafing birds (Michel & Nixon 2015). Based on a review of aerial photographs taken over time since the spill, the Trustees estimate that vegetation killed by overlying stranded boom may need up to a year to recover new growth after the boom was removed.



Source: Michel and Nixon (2015).

Figure 4.6-45. Map of estimated locations of stranded boom in coastal wetland habitats, based on aerial imagery and on-the-ground surveys. The vast majority of stranded boom was in Louisiana.

4.6.4.8 Integration of Coastal Wetland Injury Quantification

Table 4.6-17 summarizes injury to coastal wetland resources from shoreline oiling.

Table 4.6-17. Coastal wetland resources assessed and injury findings, including the maximum percent change relative to reference conditions, the year of maximum impact or the year studied, the zone or width of impact, the miles (kilometers) of affected shoreline, the time period observed, and the expected recovery time. Length of affected shoreline is based on 2008 shoreline. Actual length of affected shoreline in Louisiana may exceed values reported in the table by up to 40 percent in some areas. (NA: not analyzed)

Model Species/Injury	Max percent change relative to reference condition	Year of max impact/year studied	Zone/width of impact	Miles (km) of shoreline affected	Observed Time Period	Expected Recovery Time
Mainland Herbaceous **above ground biomass**	53%	2010	Edge, Interior	350-721 (563-1,161)	2010-2013	2-8 years
Mainland herbaceous **total live cover**	35%	2010	Edge, Interior	350-721 (563-1,161)	2010-2013	2-8 years
Amphipod mortality	96%	2010	Edge, Interior	155 (249)	2010-2013	more than 4 years
Periwinkle abundance	90%	2011	Edge, Interior	39 (62)	2011	more than 10 years
White Shrimp growth (oil)	46%	2011	Intertidal, Edge	179 (288)	2011	more than 2 years
Brown Shrimp growth (oil)	56%	2011	Intertidal, Edge	179 (288)	2011	more than 2 years
Brown Shrimp production (freshwater)	60%	2010	Intertidal	NA	2010	1 year

***Fundulus* hatch success**	99%	2010	Edge	39 (62)	2010-2013	more than 4 years
Flounder growth	90%	2011	Edge	39 (62)	2011-2013	more than 3 years
Red drum growth	47%	2010	Edge	39 (62)	2010-2013	more than 4 years
Fiddler crab burrow density	>25%	2010	Edge, Interior	NA	2010-2013	more than 4 years
Insects (total arthropod community)	50%	2010	Edge/Interior	NA	2010-2012	more than 1 year
Nearshore oyster cover	99.5%	2013	Intertidal	155 (250)	2013	no recovery

4.6.4.9 Synthesis of Coastal Wetland Resource Assessments

4.6.4.9.1 Synthesis of Conclusions and Key Aspects of the Injury for Restoration Planning

Widespread injury occurred across the estuarine coastal wetland complex and included subtle effects (e.g., reduced growth and egg hatch success) to lethal effects (e.g., death). These effects occurred to diverse species that use these coastal habitats for some or all of their lifecycle. Injuries occurred to plants and amphipods at the base of the food web and to high-level predators such as southern flounder. The diversity of these injuries indicates that the marsh habitat supporting these species has been severely degraded (Powers & Scyphers 2015). In planning restoration actions, the Trustees considered the totality of the coastal wetland injury summarized below.

Mainland herbaceous salt marshes across Louisiana, Mississippi, and Alabama were impacted. Louisiana salt marshes experienced reductions in live aboveground biomass and live plant cover ranging from 11 to 53 percent compared to reference conditions over a total of 350 to 721 miles (563 to 1,161 kilometers) (Zachary Nixon et al. 2015a). Recovery time estimates range from 2 years after the spill for lighter oiling categories to 8 years after the spill for heavier oiling categories (Zachary Nixon et al. 2015a). Mainland salt marsh vegetation in Mississippi and Alabama was also adversely affected by the oil spill based on significant reductions in live aboveground biomass (Hester et al. 2015). Louisiana mangrove-marsh habitat also sustained oil-related impacts based on multiple indicators of the reduced vegetative extent of mangroves due to plant oiling (Willis & Hester 2015a). The marsh edge, which serves as a critical transition between the emergent marsh vegetation and open water habitat, suffered the most serious injuries (Hester et al. 2015; Powers & Scyphers 2015). However, vegetation on the marsh platform behind the edge was also oiled and injured (Hester et al. 2015). During the response, over 52 acres (21 hectares) of marsh were affected by stranded boom (Michel & Nixon 2015).

Substantial decreases in secondary production (50 to 96 percent decline) were estimated for amphipods, periwinkles, brown and white shrimp, southern flounder, and red drum in marsh areas that experienced heavy persistent oiling compared to shoreline areas that had no observed oil; and reduced secondary production also occurred in areas with intermediate levels of oiling (Powers & Scyphers 2015; Zengel et al. 2015b). Independent analyses performed by State Trustees (Blancher et al. 2015) support

the observation of reduced secondary productivity for these and additional species in the marsh edge environment.

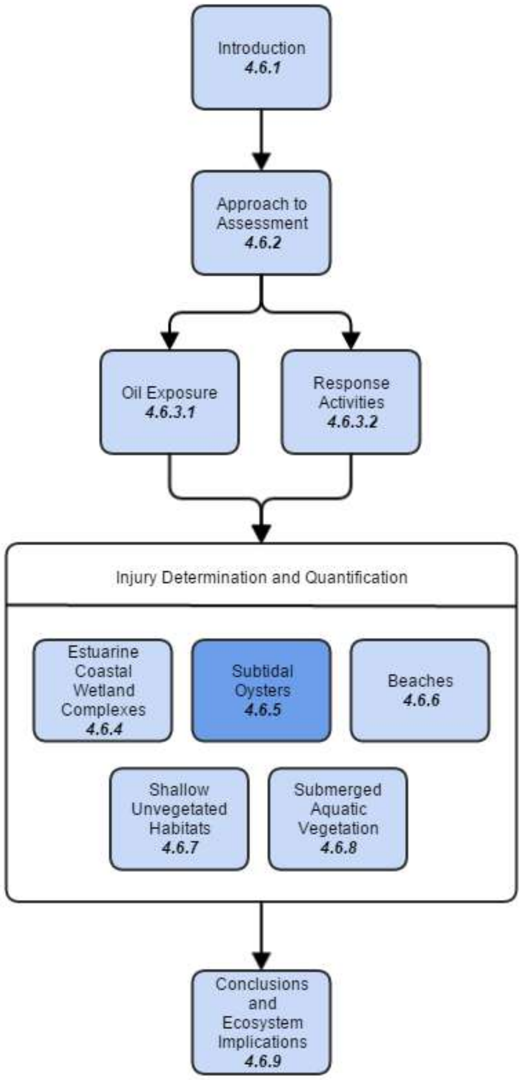
Oyster habitat adjacent to marsh areas was reduced by 60 percent in areas of heavy persistent oiling compared to reference areas (Powers et al. 2015b). In addition to loss of habitat and production, fecundity was substantively reduced (99 percent reduction in egg hatching success) for *Fundulus*—the one coastal wetland species for which fecundity studies were performed (Powers & Scyphers 2015). The expected duration of the estimated losses of ecosystem function varied by taxa and range from 2 years (lightly oiled vegetation) to permanent losses (nearshore oysters) (Zachary Nixon et al. 2015a; Powers & Scyphers 2015; Roman 2015; Zengel et al. 2015b; Zengel et al. 2015c). In many cases, the marsh fauna will not recover until PAH concentrations on the marsh edge decline substantially. These fauna are a few representative species and reflect only a small proportion of what likely happened to the broader faunal community. In addition, shoreline erosion rates approximately doubled over at least 108 miles (174 kilometers) more than 3 years after the spill, and this loss is permanent (Roman 2015).

As described in Chapter 5 (Sections 5.5.2 and 5.5.9), the Trustees plan to compensate for these injuries by restoring a suite of coastal habitats stretching across the northern Gulf of Mexico.

4.6.5 Subtidal Oyster Assessment

Key Points

- Subtidal oysters provide a multitude of ecosystem services, including improved water quality and habitat for economically and ecologically important marine species.
- Subtidal oyster abundance in coastal Louisiana was reduced by summer river water releases, which caused direct mortality and subsequent reproductive failure.
- Between 4 and 8.3 billion subtidal oysters (adult equivalents) are estimated to have been lost due to direct mortality and a consequent lack of reproduction. Over three generations, which represents a minimum recovery time, the killed oysters would have produced a total of 240 to 508 million pounds of fresh oyster meat.
- The dramatic decreases in oyster densities and the associated reproductive injury imperils the sustainability of oysters in the northern Gulf of Mexico.



Within most estuaries in the Northern Gulf of Mexico and Atlantic Ocean, the Eastern Oyster (*Crassostrea virginica*) forms reefs that provide ecosystem services that benefit human societies; these ecosystem services include, but are not limited to, enhanced estuarine habitats, improved water quality, and shoreline stabilization (Grabowski et al. 2012). For instance, oysters enhance the recruitment and growth of economically valuable and ecologically important finfish and crustaceans, thereby increasing these species' productivity (Breitburg et al. 2000; Coen et al. 1999; Grabowski et al. 2005; Harding & Mann 2001; Peterson et al. 2003a; Soniat et al. 2004; Tolley & Volety 2005). Oyster reefs concentrate bottom deposits of feces that promote bacterially mediated denitrification, thereby counteracting anthropogenic nitrogen loading (Carmichael et al. 2013; Kellogg et al. 2013; Newell et al. 2002; Piehler & Smyth 2011; Smyth et al. 2013). By filtering water and enhancing light penetration, oysters promote other valuable estuarine habitats such as submerged aquatic vegetation (Carroll et al. 2008; Everett et al. 1995; Newell 1988; Newell & Koch 2004; Wall et al. 2008). In combination with their nearshore counterparts (discussed in Section 4.6.4.5.9, Nearshore Oysters), they stabilize shorelines and protect against storm surge and erosion (Meyer et al. 1997; NRC 2014; Piazza et al. 2005; Scyphers et al. 2011). Nearshore and subtidal oysters seem to form a single larval pool (Murray et al. 2015).

4.6.5.1 Approach to the Assessment

All approaches to estimating *Deepwater Horizon*-related subtidal oyster injury used field-collected abundance data, either from Louisiana's oyster fisheries-independent monitoring program (which focuses only on public subtidal oyster grounds and not on nearshore or leased areas) or from the *Deepwater Horizon* NRDA. Abundance and cover of subtidal oysters were evaluated during multiple field sampling events beginning in 2010 and continuing into 2014. Larval settlement and recruitment in subtidal areas have also been monitored throughout the region since 2010. Because oysters are sensitive to salinity fluctuations, the release of river water during the response was a major concern for oyster health. Although oysters are exposed to freshwater from natural events, such as storms, the opening of the salinity control structures exposed oysters to low salinities for much longer time frames and during different seasons (late spring/summer) than in normal years. The river water releases happened at the peak time for oyster growth and reproduction and had more severe consequences than salinity fluctuations during the fall and winter. Buoyant larvae were likely exposed to oil in surface waters during the summer of 2010 (Powers et al. 2015a).

The relationship between oil exposure and abundance of subtidal oysters was examined by evaluating relationships between abundance of each age class (adult, juvenile, and spat) and oyster tissue PAH concentrations, oiling (as measured in terms of co-located sediment TPAH50), and oil-on-water (days, frequencies, and presence/absence).

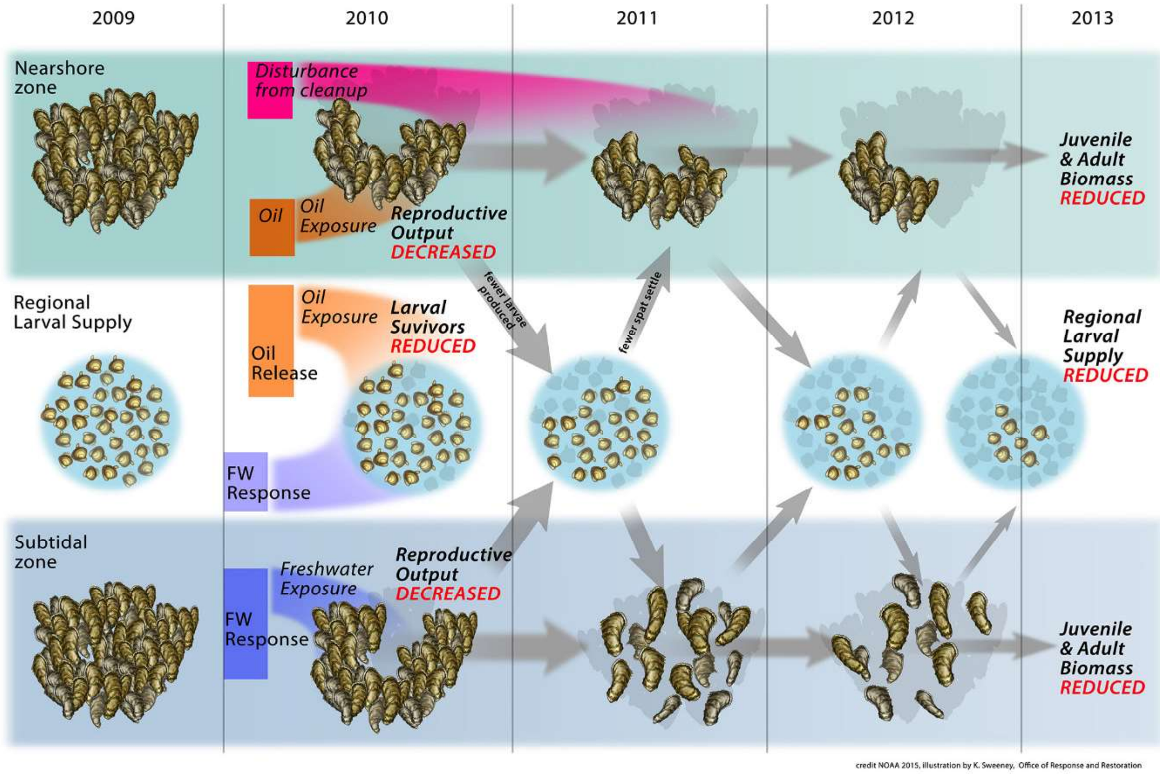
Exposure to *Deepwater Horizon*-related river water releases was characterized using modeled daily average salinity predictions derived from extensive local salinity measurements at monitors and sampling stations (McDonald et al. 2015). The Trustees applied three approaches to quantify subtidal oyster injury resulting from exposure to water released from the Davis Pond and Caernarvon salinity control structures in 2010:

- The first approach ("NRDA Spatial") assessed oyster density differences between areas highly exposed to freshwater and areas less exposed, using data collected under the *Deepwater Horizon* NRDA in 2010.
- The second approach ("Fisheries Temporal") quantified injury by comparing abundance in 2010 to those of prior years, both for the area of impact and basin-wide, using annual fisheries-independent data collected by the state of Louisiana. This monitoring data has limited spatial coverage, especially in Barataria Bay, but Trustees applied the abundance observations from these stations throughout the areas evaluated (area of impact and basin-wide).
- The third approach ("NRDA/Nestier") combined NRDA 2010 abundance data with a dose-response function derived from annual LDWF Nestier tray studies to identify areas likely to have experienced decreased survival due to freshwater from the Davis Pond and Caernarvon salinity control structures. The percent cover of oyster resource in the area of injury (measured as part of NRDA field studies) was used to calculate the total expected loss of oysters for each basin (Roman & Stahl 2015a).

Larval transport modeling was conducted to determine connectivity between nearshore and subtidal oyster habitat and between basins to assist in interpreting observations of spat settlement since 2010

(Murray et al. 2015). Oyster larvae release locations and timing in this model were intended to represent areas affected by shoreline oiling and cleanup actions, areas and timing of river water releases, and areas with oiling in surface waters.

4.6.5.2 Conceptual Model and Pathways for Oil and Response Actions to Affect Subtidal Oysters



Source: *Kate Sweeney for NOAA.*

Figure 4.6-46. This figure illustrates: (1) how the *Deepwater Horizon* spill harmed oyster populations in the Gulf of Mexico; and (2) how the connections between oyster populations in the nearshore and subtidal zones and the regional larval supply could result in ongoing suppression of the oyster population following the initial injury. The top row (pink) shows disruption of cleanup actions in 2010 and 2011 to nearshore oyster cover. The top row (orange) illustrates oil killing nearshore oysters in 2010. The bottom row (blue) shows influence of salinity control structures on oysters in the subtidal zone. The middle row integrates effects of freshwater and oil and cleanup actions to illustrate that fewer surviving adult oysters in the nearshore and subtidal zones would produce fewer larvae in later years. Oyster larvae present in 2010 were also exposed to oil in the water.

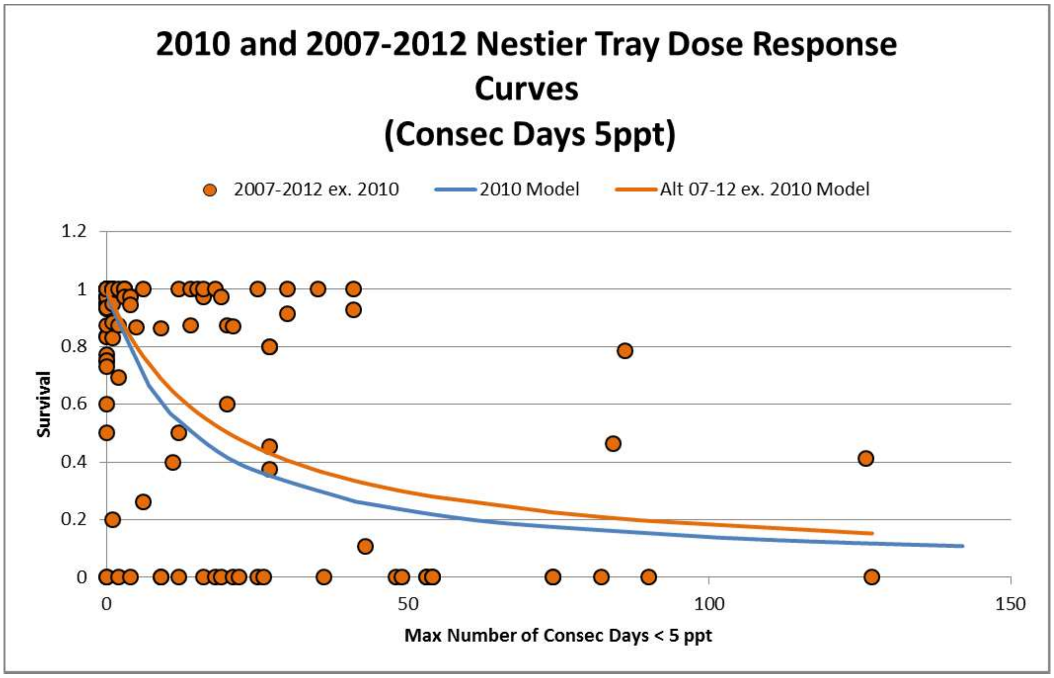
As indicated in Figure 4.6-46, oysters could be affected in many ways: by river water released as part of *Deepwater Horizon* response actions (Martınez et al. 2012), by shoreline oiling or physical nearshore response actions, and by exposure of larvae to oil in surface waters during the late spring and summer of 2010. Reductions in oyster abundance and cover in the subtidal and nearshore zones would be expected to reduce the spawning stock available to repopulate oyster reefs throughout the region (Grabowski et al. 2015a). Effects on subtidal oysters are discussed in the following section. Effects on

nearshore oysters are discussed above with other marsh fauna. Ongoing and future effects on regional oyster recruitment as a result of injury to subtidal and nearshore oysters are also summarized below.

4.6.5.3 Injury Determination

4.6.5.3.1 Impacts of River Water Releases on Subtidal Oysters

The abundance of subtidal oysters in coastal Louisiana was reduced by summer river water releases conducted as part of response actions to the *Deepwater Horizon* spill (Grabowski et al. 2015b; Powers et al. 2015a). As discussed in Section 4.6.3.2.2 (River Water Releases), the timing, volume, and duration of the low salinity water from these response actions were unusual compared to the years prior to the spill (2006 to 2009), leading to very large areas that experienced atypical salinity conditions in the summer of 2010 (Rouhani & Oehrig 2015b). When average daily salinity conditions dropped below 5 parts per thousand for more than 30 consecutive days between April and September, substantial numbers of oysters were killed, as shown by over a decade of data collected in these zones by the state of Louisiana (see Nestier tray dose response curve in Figure 4.6-47) (Powers et al. 2015a; Rouhani & Oehrig 2015a, 2015b). Observations from NRDA sampling have confirmed this. Oyster abundance in 2010 was very low in many areas within the areas affected by these river water releases and dropped to zero over most of these areas in 2011 (Powers et al. 2015a).



Source: *Rouhani and Oehrig (2015a).*

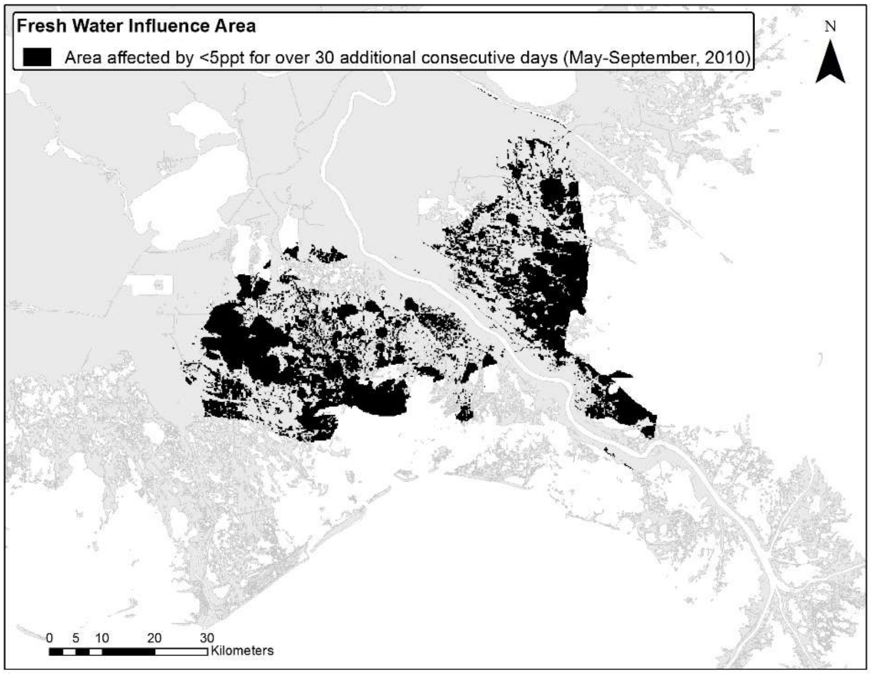
Figure 4.6-47. This graph shows the relationship between exposure to freshwater and oyster survival. As the number of consecutive days with salinity below 5 parts per thousand increases (horizontal axis), survival of oysters drops (vertical axis) and more oysters die. For example, a survival of 0.400 means that out of 100 oysters, 40 would be alive and 60 would die. The curves are derived from exposure and survival data from annual Nestier Tray oyster studies conducted in the Barataria Bay and Black Bay/Breton Sound basins by the Louisiana Department of Wildlife and Fisheries.

Furthermore, annual NRDA sampling of both oyster settlement and abundance shows that these initial injuries have severely harmed oyster reproduction in the years since the spill, decreasing prospects for recovery (Grabowski et al. 2015a). The injury determination addresses impacts to subtidal oyster abundance from both the initial freshwater-related injury and the resulting reproductive failure.

The design of the NRDA subtidal oyster studies was intended to evaluate abundance of oysters throughout the area where oil was observed on shorelines and surface waters. While toxicity studies demonstrated that exposure to oil in water from the *Deepwater Horizon* spill could also have potentially harmed oysters (Morris et al. 2015), confirmation of such exposure is limited (Jacob Oehrig et al. 2015). In addition, statistical analyses attempting to relate oyster densities with NRDA-collected data on oiling (measured in terms of co-located sediment TPAH50) and oil-on-water (days, frequencies, and presence/absence) did not support a discernable association between exposure to oil and subtidal oyster densities (Powers et al. 2015a).

To understand how response actions (river water releases) affect oyster abundance, daily salinity conditions in 2010 were compared to those of prior (baseline) years (2006 to 2009) (McDonald et al. 2015; Powers et al. 2015a; Rouhani & Oehrig 2015b). The period 2006 to 2009 was chosen to represent baseline conditions because it was after Hurricane Katrina and is most likely to represent conditions that would have occurred in the absence of the *Deepwater Horizon* spill. The number of consecutive days with average salinity below 5 parts per thousand is an important variable in determining oyster abundance because oyster survival declines dramatically the longer oysters are exposed to freshwater (Figure 4.6-47) (Powers et al. 2015a; Rouhani & Oehrig 2015a, 2015b).

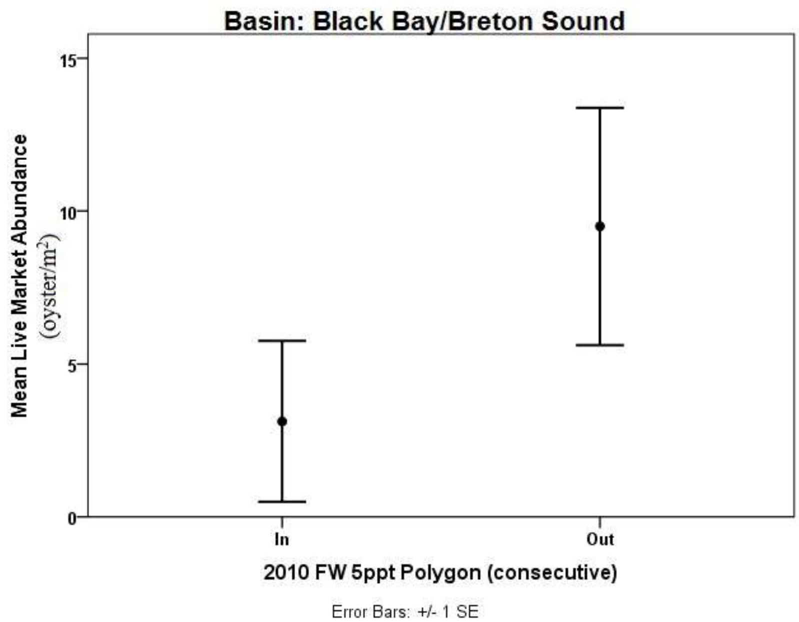
The area of freshwater impact was determined by interpolating thousands of salinity values throughout the estuary and comparing 2010 salinities to those in the years prior to the spill (2006 to 2009) (McDonald et al. 2015; Rouhani & Oehrig 2015b) (Figure 4.6-48). For each 200 square meter grid cell in the salinity model, the maximum number of consecutive days of low salinity (i.e., below 5 parts per thousand) between April 27 and September 15 was calculated for each year between 2006 and 2010. For each grid cell, the maximum number of consecutive days was averaged for 2006 to 2009 to represent the location's "historical baseline condition." Each grid cell that experienced more than 30 consecutive days of low salinity above that experienced in the historical baseline was considered affected by fresh water in 2010. The threshold of 30 days was selected to maximize the difference between average salinities inside and outside the resulting fresh water polygon in 2010, thereby representing the greatest low salinity impact. The difference in number of days below 5 parts per thousand in 2010 and prior years confirms that conditions in 2010 are well outside typical conditions for the years immediately preceding the spill (Rouhani & Oehrig 2015b). Note that the upper portions of the estuaries are white in Figure 4.6-48 because they are usually below 5 parts per thousand in baseline years and 2010 salinity conditions were not anomalous for those areas.



Source: Rouhani and Oehrig (2015b).

Figure 4.6-48. Locations in Barataria Bay and Black Bay/Breton Sound basins with more than 30 consecutive days with salinity below 5 parts per thousand in 2010 when compared to the number of consecutive days below 5 parts per thousand during historical baseline years. Note that the upper portions of the estuaries are white because they are below 5 parts per thousand in baseline years and 2010 salinity conditions were not anomalous for those areas. The widespread decrease in salinity is coincident with discharge records from the summer river water release structures, which also demonstrate the atypical nature of the massive late spring/summer river water release (see Figure 4.6-13).

Abundance and cover of subtidal oysters were evaluated during multiple NRDA field sampling events beginning in 2010 and continuing into 2014 (Powers et al. 2015a; Roman & Stahl 2015b). Trustees determined that oysters were injured in Barataria Bay (affected by the outfall from the Davis Pond structure) and in the Black Bay/Breton Sound basin (affected by the Caernarvon structure). The Trustees assessed injury using multiple methods and datasets. In all cases, the Trustees compared measured densities after the spill to a baseline or reference condition to estimate the density reduction attributable to the release of river water from these structures in 2010 (e.g., see Figure 4.6-49). All approaches to estimating *Deepwater Horizon*-related subtidal oyster injury used field-collected abundance data, either from Louisiana's oyster fisheries-independent monitoring program or from the *Deepwater Horizon* NRDA. All approaches also showed that the freshwater released in response to approaching oil in 2010 led to reduced oyster densities in Barataria Bay and Black Bay/Breton sound (Powers et al. 2015a).



Source: *Powers et al. (2015a).*

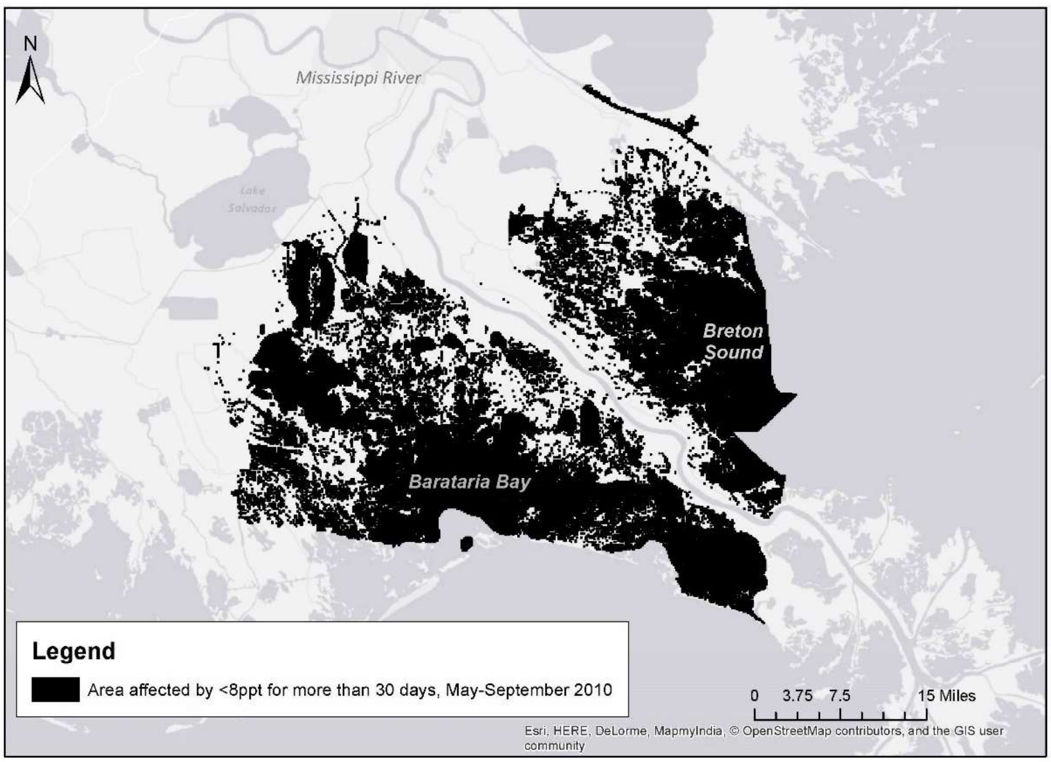
Figure 4.6-49. This graph compares the abundance of live market sized oysters per square meter (vertical axis) in Breton Sound areas affected and areas not affected by the river water releases (horizontal axis), as observed during the NRDA field study in 2010. The areas affected are those shown in black in Figure 4.6-48, where salinities dropped below 5 parts per thousand for an unusually long period. The average in the affected areas (n=4) is lower than in unaffected areas (n=4).

In addition to the NRDA analysis of injury to subtidal oyster abundance, the Trustees (Grabowski et al. 2015b) evaluated fisheries-independent data on oyster abundance collected by Gulf state monitoring programs to evaluate trends over time (i.e., before and after the spill). The Trustees found significant declines in all size-classes of oysters in the area most heavily impacted by the spill between eastern Louisiana through Mississippi, most notably in Black Bay/Breton Sound (Grabowski et al. 2015b; Powers et al. 2015a). Fisheries-independent monitoring data provide the ability to examine trends over time and are most powerful when multiple stations representing sub-regional conditions are repeatedly sampled over many years.

4.6.5.3.2 Reproductive/Recruitment Effects (Subtidal and Nearshore Oysters)

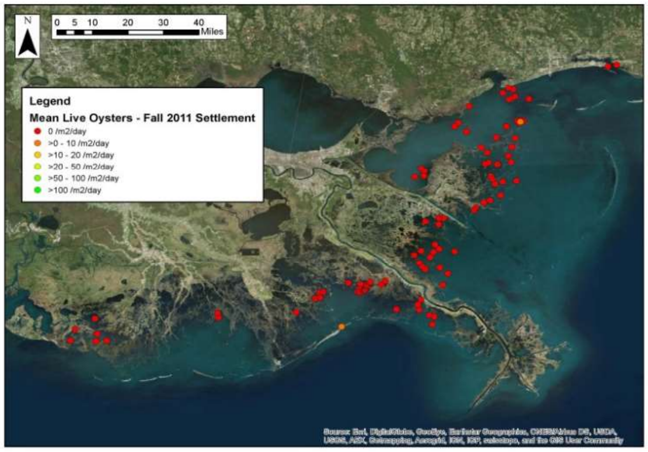
Whether oyster reefs are sustainable over time depends on the balance between factors that decrease numbers (e.g., mortality, predation, sinking into soft mucky sediments, and harvest) and factors that maintain or expand reef structures (e.g., reproduction and larval settlement, growth, and new shell production). The oyster reproductive cycle includes release of eggs and sperm into the water, fertilization of eggs, development through several larval stages, and recruitment (Kennedy 1996). This last stage refers to the settling of spat (juvenile oysters) onto already present live and dead shell surfaces or other suitable material (Kennedy 1996). The availability of clean shell for spat settlement is always a major factor contributing to oyster reef sustainability and commercial oyster fisheries. Extended periods of failure of any part of the reproductive cycle can lead to sedimentation of existing reefs, removing substrate for settlement and reducing oyster cover over time.

With reduced numbers of juvenile and adult oysters in subtidal areas in 2010, fewer larvae were produced in 2011 and beyond. In addition to the area where salinity dropped low enough to kill oysters in 2010, Figure 4.6-50 shows that a large area of oyster habitat (black-shaded area in the figure) experienced a prolonged period of daily salinities below 8 parts per thousand (Rouhani & Oehrig 2015b), conditions under which surviving oysters would have produced fewer eggs. This combined effect then led to reduced spat settlement in 2011 (Figure 4.6-51) (Grabowski et al. 2015a). With reduced oyster numbers, the decreased activity of suspension feeding would contribute to a decrease in the availability of shell free of sediment and fouling organisms, thus potentially reducing substrate available to spat and further exacerbating recruitment problems for oysters. Reduced oyster cover in nearshore areas has also contributed to a lack of recruitment and recovery throughout the region, because of the interconnectedness of the nearshore and subtidal larval pool (Murray et al. 2015). These reproductive effects have continued at least into 2014 (based on NRDA studies of oyster densities) with the result that reduced larval production, spat settlement, and spat substrate availability are compromising the long-term sustainability of oyster reefs throughout the northern central Gulf of Mexico, but especially in Barataria Bay and Black Bay/Breton Sound (Grabowski et al. 2015a).



Source: *Rouhani and Oehrig (2015b).*

Figure 4.6-50. Locations in Barataria Bay and Black Bay/Breton Sound basins with more than 30 consecutive days below salinity thresholds of less than 8 parts per thousand in 2010. These locations represent the influence of freshwater releases in response to the *Deepwater Horizon* spill.



Source: *Grabowski et al. (2015a).*

Figure 4.6-51. This map shows measurements of oyster spat settlement in the northern Gulf of Mexico in the fall of 2011. Settlement rate is measured as quantity of oyster spat per square meter per day. Red dots indicate very low to zero oyster spat settlement. Oyster larvae swim near the surface of the water. As they age, the larvae sink and must attach to shells or other hard surfaces to complete their development.

Larval settlement/recruitment in subtidal areas has been monitored throughout the region since 2010. To help interpret observations of spat settlement, larval transport modeling was conducted to determine connectivity between nearshore and subtidal oyster habitat and between sub-basins (Murray et al. 2015). Circulation modeling demonstrates that nearshore oysters (affected by oiling as described above) and subtidal oysters form a common regional larval pool and identifies connections between oyster supply and settlement within and among basins (Murray et al. 2015). The loss of subtidal oysters killed by the river water releases and nearshore oysters killed by oil and cleanup actions have reduced the adult oyster spawning stock available to maintain healthy populations throughout the region (Grabowski et al. 2015a; Powers et al. 2015a; Powers et al. 2015b). Fisheries-independent data comparing spat abundance before and after the spill confirm this trend (Grabowski et al. 2015b), particularly for areas east of the Mississippi River.

The oysters that successfully spawned in the late spring and summer of 2010 released larvae into areas where surface waters were covered by oil slicks (see Section 4.2, Natural Resource Exposure). The Trustees conducted toxicity testing to determine whether *Deepwater Horizon* oil would affect oyster egg fertilization and larval health. In these tests, exposure to oil significantly reduced growth, survival, and settlement success of oyster larvae (Morris et al. 2015). However, due to difficulties determining the number of oyster larvae that encountered toxic concentrations, the assessment of injury to oyster reproduction from losses to spawning stock and oyster cover do not include losses due to toxic effects of oil exposure on larvae released in 2010.

4.6.5.3.3 Confounding Factors

The Trustees conclude that the multi-year reproductive injury is due primarily to the *Deepwater Horizon*-related mortality in 2010 (subtidal) and 2010-2011 (nearshore). While it is true that 2011 was a high flow year for the Mississippi River that led to the opening of two major spillways (Bonnet Carre and

Morganza), salinity observations and modeling indicate minimal overlap between areas experiencing unusually low salinities in 2010 and 2011 (Rouhani & Oehrig 2015b). Furthermore, the small area where overlap exists is unlikely to include significant oyster habitat (McDonald et al. 2015). In the analysis of the relationship between oyster abundance and environmental factors, salinity differences from prior years were far more important in explaining observed oyster abundance than temperature variations from prior years and other variables (e.g., disease prevalence, precipitation, and harvest closures) (Powers et al. 2015a).

Injury Quantification

Between 1.1 and 3.2 billion subtidal oysters (adult equivalents) are estimated to have been directly killed in 2010 in areas affected by low salinity waters—an area of 118,000 acres or 479 square kilometers of oyster habitat (Powers et al. 2015a). The number of adult equivalent oysters lost was calculated by adding numbers killed in each size class and adjusting spat and seed numbers for the proportion that would have been expected to survive to adults. Over their 5-year lifespan, oysters that had not been killed would have produced a total of 69 to 195 million pounds of fresh oyster meat (wet weight). Approximately 60 percent of that total represents an estimate of the weight of the oysters directly killed, and the remaining 40 percent represents additional growth of adult oysters over the rest of their lifespan that did not occur because they were killed. The growth portion is less than the weight of the direct kill because oyster harvesting limits the potential future growth of subtidal oysters. These losses were evaluated using field measurements of oyster abundance after the spill, historical information from the states on oyster abundance, mapping of oyster cover conducted by NRDA field studies, field observations of oyster survival over time from Nestier tray studies, salinity modeling to interpret oyster abundance observations over space and time, and literature information on survival between life stages (Powers et al. 2015a). Figure 4.6-52 illustrates the conceptual approach used to calculate these losses for the preferred method (i.e., using Nestier tray data to estimate death due to river water releases).

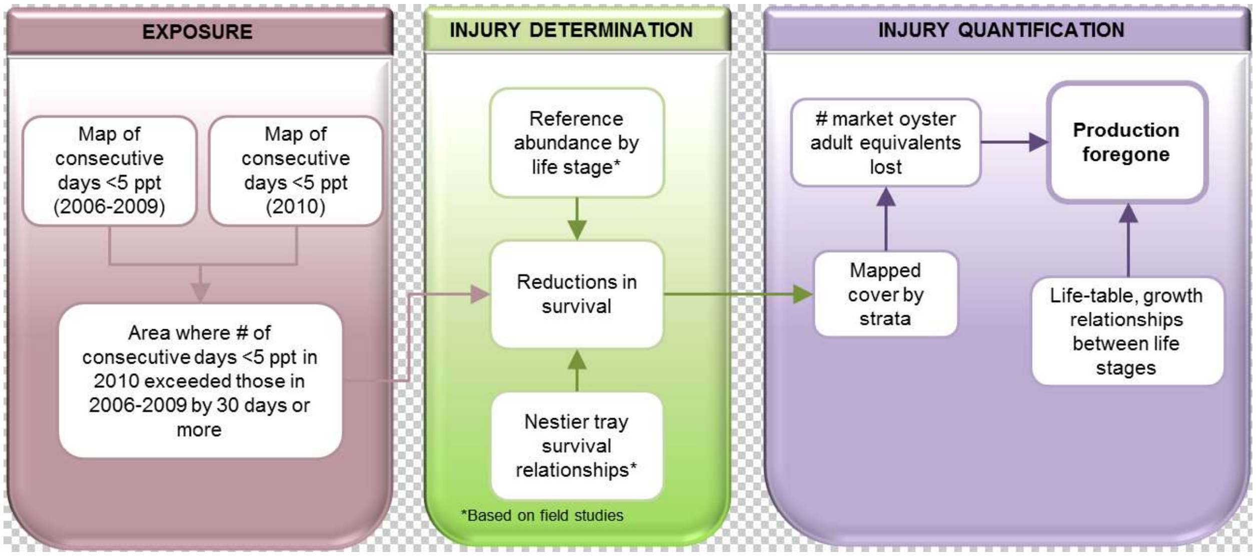


Figure 4.6-52. This diagram illustrates the analysis used to determine and quantify injury to oysters living in subtidal areas. Left box: Thousands of salinity and temperature measurements were gathered and used to identify areas where salinity was unusually low in 2010 compared to prior years. Center box: The relationship between exposure to low salinity and proportion of oysters that would die was determined from Nestier tray data and applied to the abundance of oysters present before the spill (see also Figure 4.6-47). Right box: Data on the mapped percent cover of oysters and the abundance of oysters were used to calculate the number of oysters in the affected area before the spill, the number of oysters killed, and the number (and weight) of oysters that would have grown to adult stages if the spill had not happened.

The Trustees applied three approaches to evaluate subtidal oyster injury resulting from exposure to river water released from the Davis Pond and Caernarvon salinity control structures in 2010 (Powers et al. 2015a; Rouhani & Oehrig 2015b):

- The first approach ("NRDA Spatial") assessed oyster density differences between areas highly exposed to freshwater and areas less exposed, using data collected under the *Deepwater Horizon* NRDA in 2010.
- The second approach ("Fisheries Temporal") quantified injury by comparing abundance in 2010 to those in prior years, both for the area of impact and basin-wide, using annual fisheries-independent data collected by the state of Louisiana.
- The third ("NRDA/Nestier") combined NRDA 2010 abundance data with the relationship between exposure to freshwater and expected oyster death, derived from annual LDWF Nestier tray studies (Figure 4.6-47), to identify areas likely to have experienced decreased survival due to freshwater from the Davis Pond and Caernarvon structures.

Figure 4.6-52 illustrates the approach to developing the Trustees' "most likely" estimate of oysters killed by river water releases. This approach is preferred because it uses observed relationships between salinity and oyster death in the basins of interest and observations of oyster abundance taken from those same areas. The percent cover of oyster shell in the area of injury and abundance of oysters

outside the areas of freshwater influence were used to calculate the number of oysters lost in each basin. Percent cover estimates were derived from NRDA sponsored oyster mapping studies and recent studies conducted by the state of Louisiana (Powers et al. 2015a; Roman & Stahl 2015a).

Sources of uncertainty in calculating the number of subtidal oysters killed by freshwater come from variations in salinity during the different baseline years (and uncertainty around what salinities would have been in 2010 had the spill not occurred), variations in responses to salinity between Nestier trays exposed to similar salinity conditions, variations in observations of oyster cover over the areas of concern, variation in abundance of oysters outside the area of freshwater influence as representative of pre-spill oyster abundance, and uncertainty in literature-derived assumptions about oyster growth and survival between life stages (McDonald et al. 2015; Powers et al. 2015a; Roman & Hollweg 2015; Rouhani & Oehrig 2015a; Stahl et al. 2015).

Oyster samples collected in 2013 and 2014 indicated that spat settlement has yet to recover to pre-spill levels. In order to evaluate how long this loss might persist, the Trustees evaluated the reproductive consequences of losing such a large area and extent of oyster spawning stock (Grabowski et al. 2015a). In addition to the direct kill of spawning oysters in nearshore and subtidal areas, the Trustees estimate that an additional 2.8 to 5.1 billion adult equivalent oysters will have been lost between 2010 and 2017 (which represents three generations of oysters and 7 years after spill-related injury began) due to these oysters' lost reproductive potential (Grabowski et al. 2015a). The number of adult equivalent oysters lost from reproductive potential was calculated by determining the number of eggs that were not produced by the oysters directly killed by oil and freshwater, along with the eggs not produced by surviving oysters (Figure 4.6-53). These foregone eggs were converted to adult equivalent oysters foregone using information on the proportion that would be expected to survive and grow to adult stages. Over three generations (7 years after the spill), these oysters would have produced a total of 170 to 310 million pounds of fresh oyster meat (wet weight) assuming typical survival and a 5-year maximum lifespan (Grabowski et al. 2015a, Roman and Hollweg 2015[5]).

Because oyster cover is still present in the areas affected by river water releases, substrate is still available for oyster spat to eventually settle on. Field measurements of oyster spat settlement combined with modeled oyster larvae movement, shoreline oiling, and mapping of areas affected by freshwater indicate that Barataria Bay, Breton Sound, and Mississippi Sound are areas where oyster reproduction has been most severely affected by the spill (Grabowski et al. 2015a). Oysters lost from "reproduction foregone" were calculated using estimates of oysters directly killed and estimates of oyster densities in areas experiencing reproductive suppression as described above; and these estimates were combined with literature values for sex ratios, fecundity, fertilization rates, and expected survival between life stages (Grabowski et al. 2015a). The approach to calculating these losses is illustrated in Figure 4.6-53. It should be noted that while the reproductive output of <u>subtidal</u> oyster populations may gradually recover over 7 years without intervention (because oyster shell cover is still present in these areas), the

[5] Assumptions include that the weight of one 75 millimeter market-sized oyster is 1.8 grams ash-free dry weight (afdw) in subtidal and nearshore habitats, and continued survival to 5 years contributes an additional 1.3 grams afdw per oyster in subtidal habitat (given harvest pressure) and an additional 2.4 grams afdw per oyster in nearshore habitat, given that they are not harvested.

reproductive output of missing nearshore oysters (8.3 million adult equivalent oysters per year [see Section 4.6.4.5.9, Nearshore Oysters]) would persist until restoration rebuilds spawning oysters in the intertidal zone, where oil and cleanup actions have eliminated oyster shell cover (Grabowski et al. 2015a; Powers et al. 2015b).

The total losses to subtidal oysters adds the losses of oysters directly killed by river water releases, the lost reproductive output from those oysters, and the reduced egg output of surviving oysters that experienced conditions between 5 and 8 parts per thousand salinity. When these three losses are combined, the loss to subtidal oyster populations is estimated at 4 to 8.3 billion adult equivalent oysters (Powers et al. 2015a). When adding the weight of the oysters killed to the lost weight of oysters that would have been produced in future generations, between 240 and 508 million pounds of wet oyster meat have been lost from the ecosystem over 7 years (Grabowski et al. 2015a; Powers et al. 2015a). These analyses do not include additional sublethal effects that may have occurred in the area of reduced salinity (e.g., reductions in growth in surviving oysters in the area experiencing salinities below 8 parts per thousand in 2010).

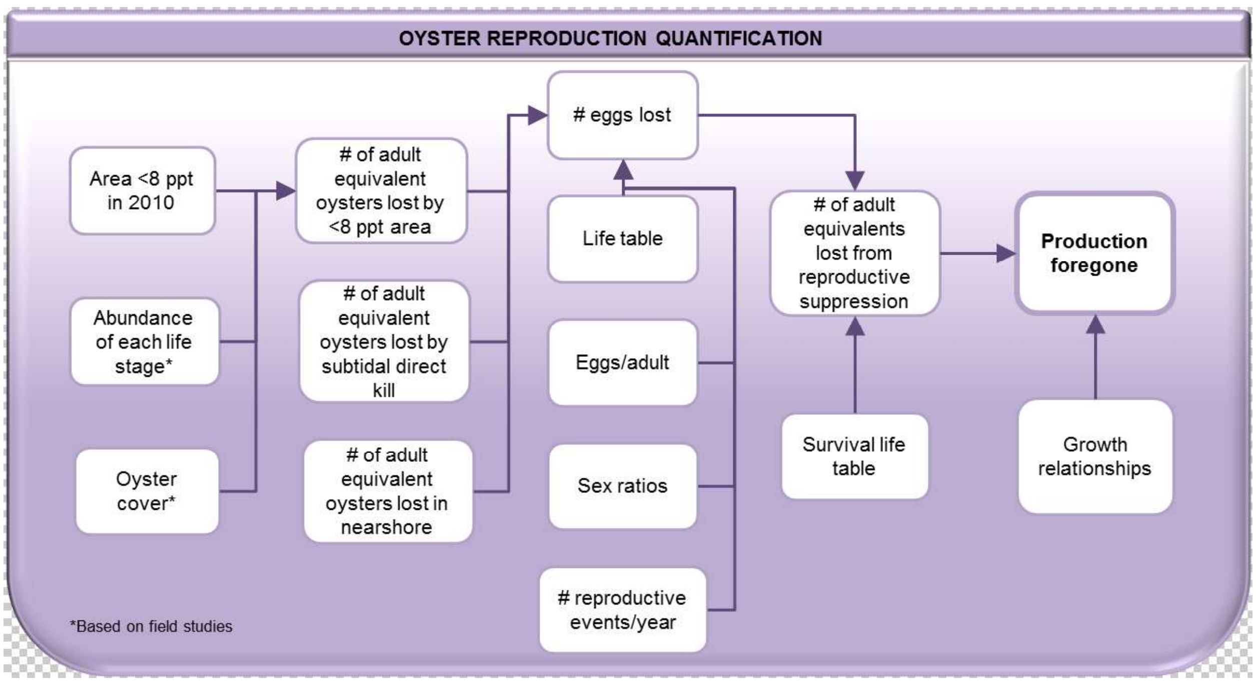


Figure 4.6-53. This diagram illustrates the calculation of lost oysters and production from injury to oyster reproduction. Oysters living in areas where salinity was lower than 8 parts per thousand in 2010 would not have reproduced; subtidal oysters killed by river water releases did not reproduce; and nearshore oysters killed by oil and response actions did not reproduce. The number of eggs lost was calculated using literature information on oyster life cycles. Lost eggs were converted to the number (and weight) of oysters that would have grown to adult stages if the spill had not happened.

Field measurements of oyster spat settlement combined with modeled oyster larvae movement, shoreline oiling, and mapping of areas affected by freshwater show that the *Deepwater Horizon* oil spill most severely affected oyster reproduction in Barataria Bay, Breton Sound, and Mississippi Sound

(Grabowski et al. 2015b; Murray et al. 2015; Powers et al. 2015a; Powers et al. 2015b). Oyster populations in the north central Gulf of Mexico will likely require substantive restoration activities to overcome the population bottleneck created by the oil spill and associated response activities (Grabowski et al. 2015a). Prior to the oil spill, the northern Gulf of Mexico had one of the few last populations of oysters that could withstand annual harvest (zu Ermgassen et al. 2012). It now appears evident that these reproductive effects have continued at least into 2014 (based on NRDA studies of oyster recruitment). These effects resulted in reduced larval production, spat settlement, and spat substrate availability that compromises the long-term sustainability of oyster reefs throughout the northern central Gulf of Mexico, with oyster reefs in Barataria Bay and Black Bay/Breton Sound showing especially severe impacts (Powers et al. 2015a). The *Deepwater Horizon* oil spill decreased oyster abundance and caused reproductive injury that imperils the sustainability of oysters in the northern Gulf of Mexico.

4.6.5.4 Conclusions and Key Aspects of the Injury for Restoration Planning

Substantial injury to subtidal oysters in the Northern Gulf of Mexico occurred as the result of the *Deepwater Horizon* spill and response actions. The Trustees took into consideration all aspects of the injury assessment in their restoration planning to offset the substantial losses that occurred to the subtidal oyster resource. Specifically, key elements of the subtidal oyster injury that informed the Trustees' restoration planning include:

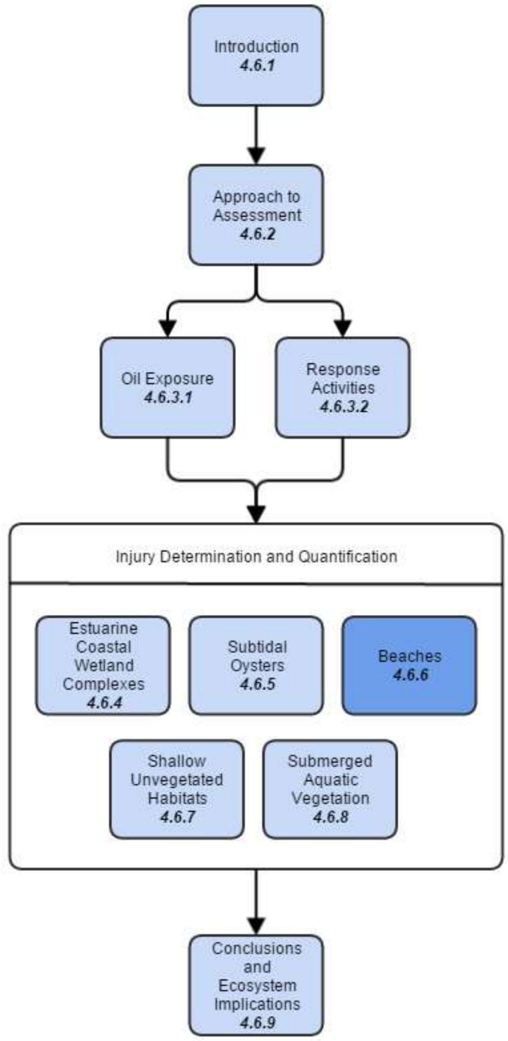
- As result of the oil spill and response activities, between 4 and 8.3 billion subtidal oysters (adult equivalents) are estimated to have been lost.
- The abundance of subtidal oysters in coastal Louisiana was dramatically reduced by summer river water releases conducted as part of response actions to the *Deepwater Horizon* spill. This injury is most pronounced in Barataria Bay and Black Bay/Breton Sound.
- The long-term sustainability of nearshore and subtidal oysters throughout the northern central Gulf of Mexico has been compromised, as a result of the combined effects of reduced larval production, spat settlement, and spat substrate availability due to the spill.

As described in Chapter 5 (Section 5.5.9), the Trustees have identified a suite of restoration approaches to offset these losses.

4.6.6 Beach Assessment

What Is in This Section?

- **Approach to the Assessment** (Section **4.6.6.1**): How did the Trustees assess the injury to sand beaches?
- **Exposure to Oil and Response Activities** (Section **4.6.6.2**): How, and to what extent, were sand beaches exposed to *Deepwater Horizon* oil and response activities?
- **Injury Determination** (Section **4.6.6.3**): How did exposure to *Deepwater Horizon* oil and response activities affect sand beaches?
- **Injury Quantification** (Section **4.6.6.4**): What was the magnitude of injury to sand beaches?
- **Inferences** (Section **4.6.6.5**): What impacts were inferred from the literature?
- **Conclusions and Key Aspects of the Injury for Restoration Planning** (Section **4.6.6.6**): What are the Trustees' conclusions about injury?



Sand beaches and their associated dunes are integral to the northern Gulf of Mexico ecosystem, playing many important economic, recreational, and ecological roles. Sand beaches and dunes provide habitat to a diversity of biota (McLachlan & Brown 2006). The casual observer sees sea oats, birds, fish in the surf, or occasionally a beach mouse or turtle on the beach. A critical underpinning to the sand beach food web is hidden within the sand and beach wrack, sometimes in burrows and sometimes without. These populations, consisting of hundreds to thousands of amphipods, crabs, shrimp, clams, snails, and worms per square meter, are a key reason that the more obvious animals such as birds and fish visit the beach: to feed on these organisms (Defeo et al. 2009; Peterson et al. 2006). Dune vegetation provides shelter and food resources for beach dwelling animals. This vegetation retains windblown sand that will renourish beaches after storms, and plant roots stabilize the beach. As a result of the *Deepwater*

Horizon spill, sand beaches and dunes across the northern Gulf of Mexico, stretching from Texas to Florida, were adversely affected by both oil exposure and the response activities that were undertaken to clean it up. In this section, the Trustees summarize their assessment of injury to sand beach habitat, which is described in more detail in Michel et al. (2015).

This section focuses on injuries to natural resources and ecological functions. Injuries to sand beaches associated with lost human uses are described separately in Section 4.10 (Lost Recreational Use).

4.6.6.1 Approach to the Assessment

Sand beaches along the barrier islands and shorelines of the northern Gulf of Mexico were exposed and injured as a result of both the direct effects of *Deepwater Horizon* oil and as a consequence of response activities undertaken to remove the oil. The Trustees took the following steps to determine and quantify the injury to sand beach and dune habitat (Figure 4.6-54):

- **Step 1 (box 1 in Figure 4.6-54).** The Trustees first characterized the exposure of sand beaches to *Deepwater Horizon* oil and response activities. This characterization took into consideration the repeated oiling of beaches that occurred over the duration of the *Deepwater Horizon* incident and the extensive efforts to find and remove oil from affected sand beach habitats. Oiling exposure was determined using information from field surveys performed as part of the U.S. Coast Guard's Unified Command's (UC's) and Trustees' efforts to respond to the spill. These data were then used to develop a shoreline oil exposure map discussed in detail above in Section 4.6.3.1.2. The Trustees also compiled all available information from the UC to determine the types, intensity, and frequency of response activities used to remove the oil from sand beaches.
- **Step 2 (box 2 in Figure 4.6-54)**. The Trustees then determined the nature and extent of injuries to sand beach habitat caused by the oil and response activities. The Trustees

Sand Beach Habitat Injury—Key Findings:

- Sand beaches are ecologically important in the northern Gulf of Mexico. They provide habitat to crabs, snails, worms, and other small organisms, which in turn are food for larger biota such as birds, fish, and turtles.
- Sand beaches across the northern Gulf of Mexico were oiled extensively as a result of the *Deepwater Horizon* spill, with the degree of exposure ranging from light to heavy oiling. Response activities similarly occurred extensively and repeatedly at sand beaches and dunes across the northern Gulf, causing additional injuries.
- The Trustees determined that there was injury to sand beach habitat across all degrees of oiling and across all types of response activity, with the injury severity determined by the degree of oiling and type and frequency of response activity.
- The Trustees concluded that at least 600 miles (965 kilometers) of sand beaches were oiled to some degree and 436 miles (701 kilometers) of sand beach habitat were injured by response activities, along sand beach coastlines stretching from Texas to Florida. The length and area (acreage) of injury by state, and for federal lands within each state, is summarized at the end of this section.

conducted an exhaustive review of existing literature to determine the effects of oil spills and the types of physical disturbances similar to the response activities conducted as part of the response efforts. This information was used to determine the nature of the biological and physical injuries and anticipated recovery rates that resulted from the spill and response activities.

- **Step 3 (box 3 in Figure 4.6-54).** Finally, the Trustees used this information to quantify the injury, in kilometers (miles) and area (hectares/acres) of sand beach habitat. The injury quantification took into consideration the initial degree of oiling and anticipated recovery and the impact of subsequent response activities.

Further description of the Trustees' exposure analysis and injury determination and quantification is provided below, and in Michel et al. (2015).

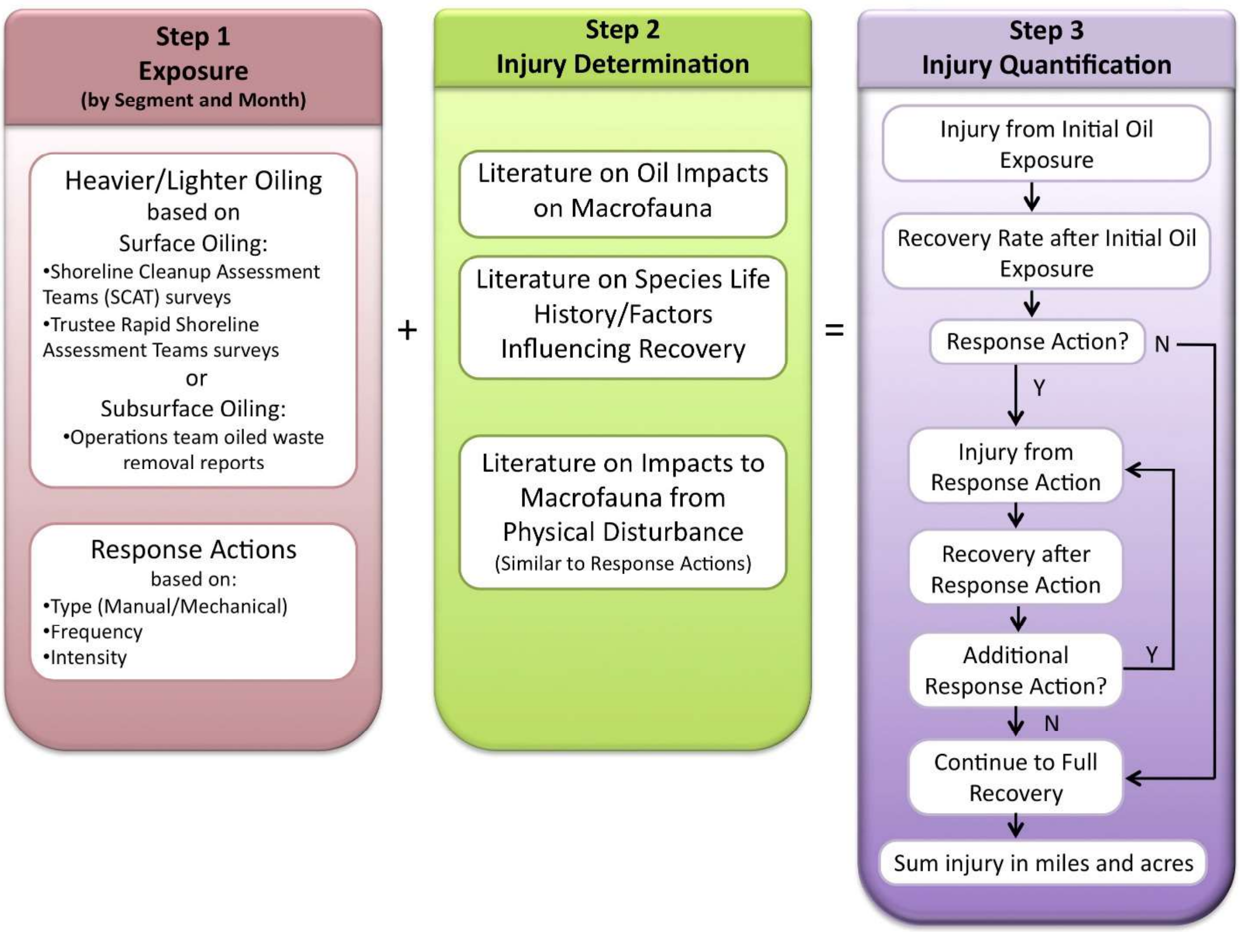


Figure 4.6-54. Approach to the sand beach exposure and injury assessment. The approach addresses impacts resulting from both oiling and response activities during the *Deepwater Horizon* spill.

4.6.6.2 Exposure to Oil and Response Activities

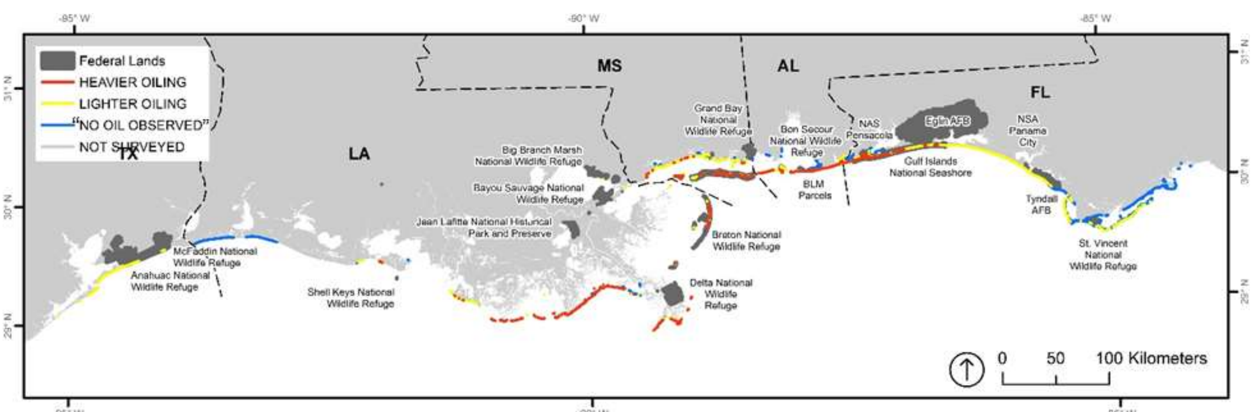
Oil exposure on sand beaches, and response activities in particular, varied widely by location and time. The Trustees characterized both the exposure to oil and response activities that occurred on sand beaches.

Key Exposure Findings:

- Over 600 miles (965 kilometers) of sand beach habitat were exposed to *Deepwater Horizon* oil in the northern Gulf of Mexico.
- 436 miles (701 kilometers) of sand beach and dune habitats were affected by response activities undertaken to clean up the oil.
- Approximately 100 million pounds (45 million kilograms) of oil waste materials were removed from sand beaches during response activities.

4.6.6.2.1 Oiling Exposure

As described above in Section 4.6.6.1 (Approach to the Assessment) and in Zachary Nixon et al. (2015b), the Trustees developed oil exposure categories for sand beaches, using a combination of information on (1) the maximum amount of surface oil observed by field teams in 2010 and (2) the amount of oiled materials removed from beaches. For the purposes of the sand beach assessment, the Trustees further grouped these into two categories: heavier and lighter oiling. Figure 4.6-55 shows the spatial distribution of these two oil exposure categories. The Trustees further calculated the total area (acreage) of exposed sand beach habitat by digitizing the width of the beach (from mean low water to the base of the dunes, seawall, or other feature) using high-resolution imagery and available topo-bathymetric data (Michel et al. 2015). They divided this width into supratidal and intertidal zone, based on photo interpretation of the high-tide line (Michel et al. 2015). Table 4.6-18 summarizes the total length and area of oiled beach in each state, and by federal lands managed by DOI and DOD within each state. Figure 4.6-56 shows the same information graphically for oiled shoreline lengths.



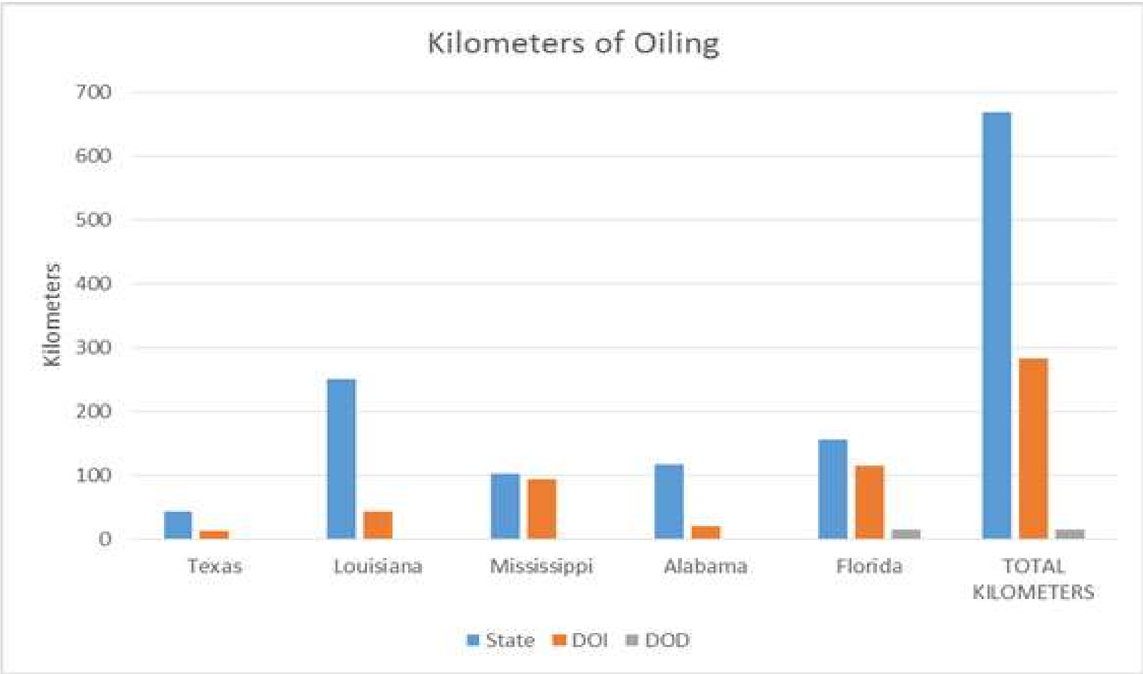
Source: *Michel et al. (2015).*

Figure 4.6-55. Map of the northern Gulf of Mexico, showing beaches that were oiled during the *Deepwater Horizon* spill. For the sand beach injury assessment, oiling exposure was grouped into two categories: lighter and heavier oiling. This map illustrates the extensive spatial extent of sand beach habitat that was oiled as a result of the *Deepwater Horizon* oil spill, from Texas to Florida.

Table 4.6-18. Length and area of oiling by state lands and Department of the Interior (DOI) and Department of Defense (DOD) lands, by state. Area is based on length and the measured width of the beach, as described in Michel et al. (2015). The top table is in miles and acres. The bottom table is in kilometers (km) and hectares (ha) (Michel et al. 2015).

	Texas		Louisiana		Mississippi		Alabama		Florida		Totals	
	Miles	Acres	Miles	Acres	Miles	Acres	Miles	Acres	Miles	Acres	Miles	Acres
State Lands	27	842	156	3,368	64	1,124	72	1,299	96	1820	415	8,437
DOI	8	197	27	632	57	1334	12	244	71	1710	176	4134
DOD	0	0	0	0	0	0	0	0	9	91	9	91
Total	35	1,039	182	4,001	121	2,458	84	1,543	176	3,621	600	12,662

	Texas		Louisiana		Mississippi		Alabama		Florida		Totals	
	km	ha	km	ha	km	ha	km	ha	km	ha	km	ha
State	43	341	250	1,363	102	448	116	526	155	737	667	3,414
DOI	13	80	43	256	93	546	20	99	114	692	283	1,673
DOD	0	0	0	0	0	0	0	0	15	37	15	37
TOTAL	57	421	293	1,619	195	995	136	624	284	1,465	965	5,124



Source: *Michel et al. (2015).*

Figure 4.6-56. Kilometers of shoreline oiling in each state and on federal lands (DOI/DOD).

4.6.6 Beach Assessment

4.6.6.2.2 Response Activities

The Trustees compiled information on response activities primarily from records kept by the UC, which had the primary authority for leading the *Deepwater Horizon* cleanup. The UC used information reported by SCAT teams on the location and amount of oil to direct cleanup and recovery operations. These activities were tracked during response. Throughout the period of the response (though more detailed recordkeeping began in June 2011), the UC kept records on the location and type of cleanup activities conducted and the amount of oil waste removed. For the purposes of directing and tracking response activities, the UC divided beaches into "segments" and "operation zones." The Trustees used the information recorded by the UC to characterize the type and extent of response activities that occurred for each sand beach response segment and operation zone, on a monthly basis. Specific examples of the types of UC reports that the Trustees used include: weekly tracking reports of the segments worked, methods used, number of workers, and amount of oiled waste recovered by tidal zone for the period from June 2011, to March 2014; Response Branch daily reports; daily ICS 209 reports, SCAT photographs, and special reports; and Shoreline Treatment Recommendations issued by SCAT (Michel et al. 2015).

> The overall scale and magnitude of the sand beach cleanup effort is illustrated by the amount of oily waste materials removed from sand beaches. For example, between May, 2010, and May, 2011, more than 76 million pounds of oily waste were removed from Louisiana beaches alone. This is roughly equivalent to the trash generated daily by 17 million people—or four times the population of Louisiana (EPA 2015). Table 4.6-19 summarizes the total amount of oily waste materials removed over the course of the multi-year cleanup effort.

Table 4.6-19. Oily material removed from sand beaches by oil spill response activities, by state (Michel et al. 2015).

State	Prior to June 2011 (kg)	June 2011–February 2014 (kg)
Texas	7,917[a]	0
Louisiana	34,501,478	6,883,846
Mississippi	1,762,847 (total) 1,552,668 (islands) 206,097 (mainland)	51,284
Alabama	1,165,369 (oiled debris to July 2011)	422,138
Florida	Not available	30,062

[a] *Equates to 10 percent of the total volume of oily solids disposed of by contractors from all Texas beaches (Texas Unified Command Memo 2011).*

The Trustees grouped different types of response activities into five Response Injury (RI) categories (Figure 4.6-57). These categories were based on the intensity and frequency of response action, as described further in Michel et al. (2015). Because of the unprecedented amount of oil that was stranded on sand beaches and the complicated temporal and spatial patterns of oiling, cleanup on sand beaches required several years of effort. During this time, many beaches were visited multiple times to clean up newly stranded oil or to clean up buried and then re-exposed oil (Michel et al. 2015). In addition, barriers were placed early in the response to prevent oil from entering sensitive habitats. These barriers included Hesco baskets, sand bags, sheet piling, and sand berms. In at least two locations (Cameron

Parish, Louisiana, and the back side of Dauphin Island, Alabama), barriers were placed on shorelines where no oil stranded. In these cases, the shoreline fauna were affected by response activities—during the placement and removal of barriers and the time when barriers were in place—even though those beaches were never oiled (Michel et al. 2015). The Trustees' injury assessment included both the impacts of repeated response visits and the impacts of response activity at beaches and dunes that were never oiled.

Shoreline Cleanup Activities Completed:

- August 2010—Texas
- June 2013—Florida, Alabama, and Mississippi
- February 2014—Louisiana, with 11 kilometers (7 miles) of beach in active monitoring
- 2015—Sporadic cleanups ongoing in Louisiana, Alabama, and Florida

Based on their compilation of response activity records, the Trustees determined that some type of response activity occurred along 701 kilometers (436 miles) of the approximately 965 kilometers (600 miles) of oiled sand beach shoreline (Table 4.6-20) (Michel et al. 2015). The severity of the impact across the 701 kilometers (436 miles) is described in Section 4.6.6.4.

Cleanup did not occur at all oiled sand beaches for several reasons, such as:

- Beaches that had sensitive habitats where further impacts were to be minimized.
- Beaches that had documented oiling, but at levels below cleanup thresholds set by the UC.
- Beaches where it was determined that the adverse effects of response activities would be greater than the effects of the oil itself.
- Beaches where oil was removed by natural processes before response actions could be taken.

The Trustees also determined that approximately 18 kilometers (11 miles) of beach habitat (272 acres or 110 hectares) were affected by the placement of barriers where oil never ultimately washed ashore (Michel et al. 2015).

Increasing severity of response injury

Response Injury (RI) Category 1

- Intermittent Manual Treatment/Augering/SOMs removal
- Manual only, lower frequency (<20 visits/month),
- includes vehicle traffic for transport of workers and waste;
- Mechanical augering;
- Submerged oil mats (SOMs) removal impacts in the intertidal zone

Response Injury (RI) Category 2

- Intensive Manual Treatment
- Mostly manual (but includes walk-behind sifters for any duration),
- higher frequency (>20 visits/month),
- includes vehicle traffic for transport of workers and waste

Response Injury (RI) Category 3

- Beach Grooming/Tilling/Very Intensive Manual
- Treatment at least twice in a month with a mechanical beach groomer that would sift the sand, down to a depth of 12 inches;
- All tilling operations;
- Intensive manual removal

Response Injury (RI) Category 4

- Excavation
- Treatment at least twice a month with a mechanical device and at least once in the month beach sediment was mechanically removed from the beach and sifted;
- Mechanical removal of clean sediments for manual removal of oiled sediments

Response Injury (RI) Category 5

- Intensive Mechanical Treatment
- Extensive deep (>12 inches) mechanical treatment, staging areas, and dredging

Source: *Michel et al. (2015).*

Figure 4.6-57. Types and descriptions of the five Response Injury categories for sand beach impact assessment.

Table 4.6-20. Length and area of shoreline that were assigned a Response Injury category (including placement of barriers) by state lands and Department of the Interior (DOI) and Department of Defense (DOD) lands, by state. The top table presents shoreline lengths in miles and acres; and the bottom table presents shoreline areas in kilometers (km) and hectares (ha) (Michel et al. 2015).

	Louisiana		Mississippi		Alabama		Florida		Totals	
	Miles	Acres	Miles	Acres	Miles	Acres	Miles	Acres	Miles	Acres
State Lands	129	2,635	46	863	70	1275	41	1,884	286	6,657
DOI	12	363	57	1,338	10	235	65	4,144	144	6,080
DOD	0	0	0	0	0	0	5	47	5	47
Total	**141**	**2,998**	**103**	**2,201**	**81**	**1,510**	**112**	**6,074**	**436**	**12,784**

	Louisiana		Mississippi		Alabama		Florida		Totals	
	km	Ha	km	ha	km	ha	km	ha	km	ha
State Lands	208	1,066	74	349	113	516	66	762	460	2,694
DOI	19	147	92	541	16	95	105	1,677	232	2,460
DOD	0	0	0	0	0	0	8	19	8	19
Total	**227**	**1,213**	**166**	**891**	**130**	**611**	**180**	**2,458**	**702**	**5,174**

4.6.6.3 Injury Determination

The Trustees determined both that there was injury to sand beach habitat and dunes and that the severity of the injury varied by degree of oiling and the specific type of response activity that occurred. This Section summarizes the Trustees' injury determination, which was based on evaluating impacts to sand beach macrofauna and to beach-nesting birds.

The Trustees determined there was injury to sand beach habitat due to oiling and response activities, based on an assessment of impacts to biological resources that utilize this habitat, including:

- Macrofauna (Section 4.6.6.3.1): Small invertebrates such as crabs, clams, amphipods, shrimp, and insects that live in the sand and beach wrack.
- Birds (Section 4.6.6.3.2) that use beaches as nesting habitat.

4.6.6.3.1 Macrofauna

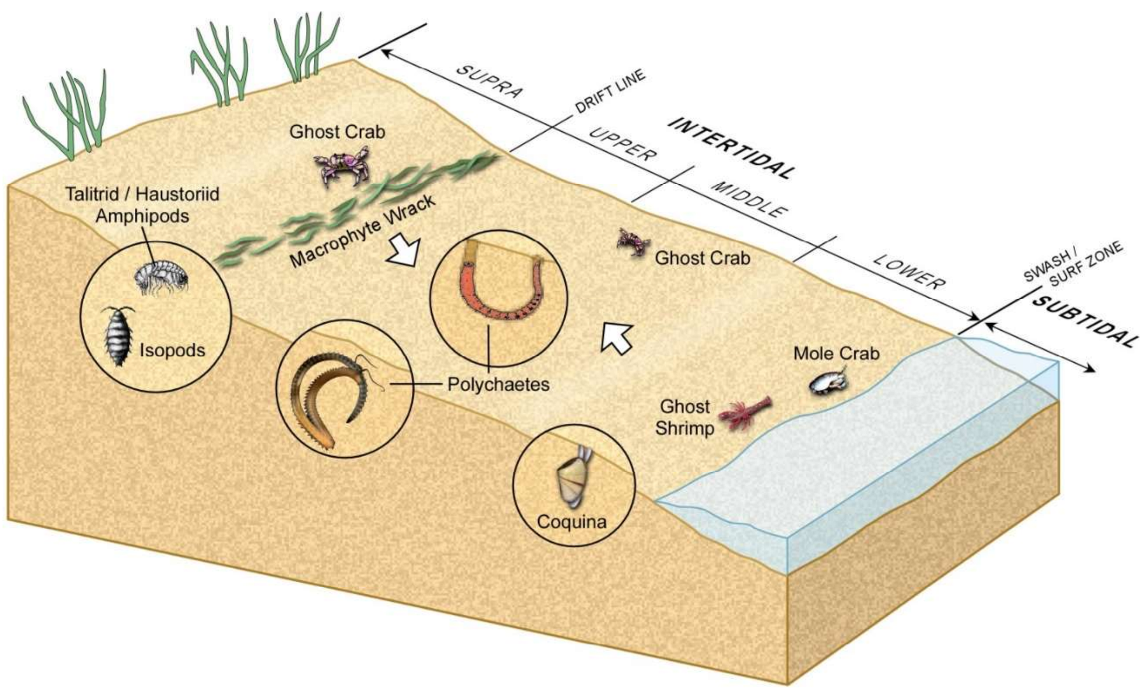
In their determination of injury, the Trustees assessed impacts to small organisms collectively called "macrofauna" that live in the sand and beach wrack. These organisms include amphipods, ghost and mole crabs, clams, ghost shrimp, and insects.

The reliance of these beach macrofauna on the quality of beach habitat and wrack to feed, reproduce, and shelter makes them key indicators for assessing beach health and function and its impairment by oil and spill response activities (Bessa et al. 2014; Hooper 1981; Junoy et al. 2005; Moffett et al. 1998; Peterson et al. 2000; Schlacher et al. 2007; Thebeau et al. 1981; Witmer & Roelke 2014). Beach macrofauna perhaps serve better than any other biological group as indicators of injury to a sand beach

and of recovery of the beach habitat and were used in this assessment to gauge the level injury caused by the *Deepwater Horizon* oil and spill response activities.

The Trustees focused on macrofauna in their determination of injury because these small organisms are a key component of sand beach ecosystems and are major food source for other biota, including birds and fish (Nel et al. 2014; Rothschild 2004). The oil and cleanup efforts directly affected the sand and wrack that supports macrofauna communities, which in turn support other wildlife such as shore birds.

The determination of injury was made for two distinct sub-habitats within sand beaches—supratidal (above mean high tide) and intertidal (between mean high and low tides) zones (Figure 4.6-58). This is because the macrofauna communities within these two zones are different in at least two regards. The zones have distinct recovery timeframes, as described in detail in Michel et al. (2015); and amounts of oiling and response activity differed across the two zones. While oiling occurred over the entire beach, the most intensive oiling occurred in the intertidal zone (Michel et al. 2015). By contrast (and perhaps somewhat counter-intuitively), the more intensive response actions actually took place in the supratidal zone. This is because oil often became buried in the supratidal zone, and therefore required more intensive response efforts to remove it than the oil that predominantly remained exposed at the surface in the intertidal zone (Michel et al. 2015).



Source: Michel et al. (2015).

Figure 4.6-58. Distribution of representative sand beach macrofauna within the supratidal (above mean high tide line, or "drift line") and intertidal (between mean high and low tide—or between the draft and surface zone) habitats. Wrack, which is a key element of the supratidal beach habitat, is also depicted.

The injury determination for both supratidal and intertidal macrofauna communities was based on a comprehensive review of the scientific literature on:

1. The life history of sand beach invertebrates (how and when different key species reproduce) and what factors may influence how impacted communities recover over time; and
2. How oil and different degrees of disturbances similar to the response activities affect these communities as they start to recover from the *Deepwater Horizon* incident.

The following text first describes the macrofauna communities that inhabit the supratidal and intertidal zones, their life histories, and factors that affect their recovery from an adverse impact to their habitat. The text then summarizes information from the literature on the impacts of oiling and response activities on macrofaunal communities.

Description of Supratidal and Intertidal Macrofauna Communities

> Beach wrack is a key component of sand beach ecosystems. Composed of decomposing vegetation, this material is a rich source of food and nutrients for beach organisms. Crabs, insects, and other macrofauna live on and eat the wrack; and many other larger beach inhabitants such as birds, fish, and mammals in turn eat the macrofauna.

The invertebrate communities in these two zones have very different species compositions, life histories, and habitat requirements that respond differently to the types of response activities applied in the intertidal and supratidal zones.

In the *supratidal* zone, organisms associated with beach wrack represent an important prey source for higher trophic levels, such as birds and fish (Dugan et al. 2003). Thus, the supratidal beach community depends on the presence of wrack as an essential habitat feature. However, if wrack is not present or has been removed from the beach, no wrack macrofaunal community can develop, even if reproductively capable adults are present (Dugan et al. 2003). The extensive removal of wrack—both oiled and unoiled—during the initial cleanup stages drove the initial injury to the sand beach supratidal habitat. The semi-terrestrial components of the supratidal community, such as amphipods, have limited dispersal of young (McLachlan & Brown 2006; Nelson 1993).This means that much of the recruitment of the supratidal community must come from local sources over relatively short distances. When beaches become heavily oiled and the resident wrack becomes oiled or is removed, recovery will only be successful once the wrack upon which they depend returns (Michel et al. 2015).

The *intertidal* community species and their life history, on the other hand, are very different from those found in the supratidal zone of the beach. The intertidal benthic community consists predominantly of marine species and is dominated by coquina clams, mole crabs, polychaetes, and haustorid amphipods. The majority of these species feed by filtering out food particles from the water in the swash zone above them, and these species rely on beach or surf diatoms as their primary source of nutrition. For many of these species, the mature adults live in the sediments and release fertilized eggs or larvae into the water column. The larvae drift passively for an interval of days to weeks, rarely months, while they develop. When they drift to appropriate habitat, the juveniles settle in the beach and start the cycle over again (McLachlan & Brown 2006). Thus, adults with this life history on a beach come from some distance up

current; the current direction can vary, but is generally from the east in the northern Gulf of Mexico (Georgiou et al. 2005; Stone & Stapor 1996).

Adverse Effects of Oiling and Response Activities

The Trustees evaluated data from other oil spills to characterize the disturbance and recovery of the supratidal and intertidal macrofaunal communities from the *Deepwater Horizon* spill.

The Trustees assessed the impact of oil on both tidal zone communities based on studies of two major spills with characteristics that were most similar to the *Deepwater Horizon* spill. Bejarano et al. (2011) summarized impacts from these spills.

- The 1979 Ixtoc I spill occurred on beaches with similar fauna as the northern Gulf of Mexico, with a similar type of oil (i.e., heavily weathered crude oil that had been transported long distances before stranding onshore), and with comparable ranges of oiling intensity (Kindinger 1981; Tunnell et al. 1982). Researchers of this spill compared pre- and post-oiling intertidal faunal communities at thirteen beaches and reported decreases of 85 to 97 percent in species at the more heavily oiled beaches. For the supratidal invertebrates, Hooper (1981) reported the complete loss of key amphipod species.
- The T/V *Prestige* spill of heavy oil off Spain in 2002 was also similar in terms of the type of oil and the extent and degree of oiling. Studies showed decreases in species abundances of 60 to 85 percent (de la Huz et al. 2005; Junoy et al. 2005). The geographic extent of the spill's shoreline oiling is the largest of any marine spill globally (Z. Nixon et al. 2015), affecting vast contiguous kilometers of sand beach shoreline. Many stretches of shoreline oiled repeatedly over many months and, in the case of buried oil and submerged oil mats acting as secondary sources, over many years (see Section 4.2, Natural Resource Exposure).

Studies of the effects to the sand beach community from historical oil spills are generally much smaller in terms of the volume spilled and affected shorter lengths of coastline. As a result, studies of earlier spills can only provide a minimum measure of likely impacts from the *Deepwater Horizon* spill.

Based on review of the literature on oil impacts on sand beaches, the Trustees estimated that:

- In the supratidal zone, locations with heavier oiling had a 100 percent decrease in macrofauna abundance when compared to unoiled locations; and locations with lighter oiling had a 20 percent decrease.
- In the intertidal zone, locations with heavier oiling had a 95 percent decrease in macrofauna abundance when compared to unoiled locations; and locations with lighter oiling had a 40 percent decrease.

The Trustees also review the literature to identify impacts to macrofaunal communities that likely occurred as a result of the types of response activities that took place on sand beaches (Table 4.6-21). This information was then used to assign response activities to a Response Injury (RI) category with increasing severity from 1 to 5. Michel et al. (2015) provide a more detailed summary of the impact to

sand beach communities from the types of physical disturbances that occurred during response activities; and that research was used to estimate the degree of invertebrate community disturbance and to assign each activity to a RI category.

Table 4.6-21. Expected impacts from different types of response-related activities that were conducted during the *Deepwater Horizon* response, based on studies reported in the literature. Refer to Michel et al. (2015) for a detailed discussion and references cited.

Disturbance Type	Impacts to Macrofaunal Communities
Foot Traffic	Consistent 10-fold decreases in ghost crab abundances between visited and unvisited beaches Reduced survival of softer-bodied crustacea and juvenile bivalves in the lower intertidal
Off-road Vehicle Traffic	Direct mortality of nocturnal animals such as ghost crabs during night operations Lower abundance, species richness, and diversity of intertidal macrobenthos due to direct crushing, which is increased when the vehicle turns due to increased shear Crushing and burying of wrack, which affected wrack-associated species
Wrack Removal	Depressed species richness, abundance, and biomass of wrack-associated fauna, Reduction of prey for higher trophic levels Reduced percent total organic matter in the upper beach zone Disappearance of air-breathing amphipods or sandhoppers
Mechanical Sifters	100 percent removal and mortality of animals that were larger than the screen size Alterative of beach sediment by removal of shell material Desiccation of animals in sand stockpiled to dry prior to sifting Changes in sand compaction that can increase erosion during wind storms
Tilling	Crushing of burrows Changes in sand compaction that can increase erosion during wind storms
Sand Excavation/ Dredging/ Staging Areas	Complete mortality of resident biota Increased sand compaction impacts burrowing behavior and reduces the abundance of burrowing fauna, leading to reduced substrate productivity and microhabitat suitability
Barriers	Faunal loss from disruption of movement by fauna, sediment, and detritus between tidal zones Crushing of burrows and fauna during placement and removal

Recovery

Data from previous oil spills have usually shown that it takes more than a year for the invertebrate community on sand beaches to recover from of the effects of an oil spill; recovery times have been documented to range between 0.5 to 5 years (Bejarano et al. 2011). However, it is clear from the scientific literature that the recovery of sand beach communities after an oil spill is not only dependent on the persistence of the oil and on beach dynamics and characteristics (e.g., shoreline type, beach geomorphology), but also depends on the specific macrofauna community composition, species-specific sensitivity to oil toxicity, and physical fouling by the oil. In addition, invertebrate community recruitment patterns and lifecycles can have a substantial effect on recovery timeframes on sand beaches (Michel et al. 2015).

Previous studies that have shown that beach macrofauna naturally recover to pre-disturbance abundances over relatively short time frames were mainly focused directly on the impacted beach

(Bejarano et al. (2011), and did not adequately characterize the full recovery of the larger surrounding ecosystem. The immediate accumulation of new individuals (immigrating from lagoon beaches or shallow subtidal areas) in a disturbed location does not necessarily represent recovery of ecosystem services within the broader area. The absence of the emigrating individuals from the location from which they originated means that the communities in the original location become lessened. Recovery for the entire beach ecosystem will not occur before the macrofauna population reproduces, providing new individuals that survive and grow, and thereby actually replace those that were killed by the disturbance (Michel et al. 2015).

Consequently, estimating actual recovery of macrofauna on injured oiled beaches requires an understanding of (1) recruitment distances patterns and (2) life history and reproductive capabilities of the affected macrofauna. For example, many of the heavily oiled beaches that also underwent intense response-related disturbances are isolated from large source populations and would have to rely on very small, local spawning populations for re-population, and this will slow their recovery rate.

Further discussion of sand beach macrofauna recovery timeframes subsequent to the *Deepwater Horizon* spill and response activities is provided in the injury quantification section below (Section 4.6.8.3.2).

4.6.6.3.2 Beach-Nesting Birds

Beaches provide multiple services to a variety of fauna. As another measure of injury to sand beach habitat, the Trustees evaluated the impacts of response activities on the use of sand beaches as bird-nesting habitat. Section 4.7 of this Chapter quantifies the avian injury that resulted from the *Deepwater Horizon* oil spill. This section does not quantify an injury to beach-nesting birds, but it does assess injury to sand beach habitat based on evaluating the loss of its function as bird-nesting habitat.

Many beaches in the northern Gulf of Mexico that were affected by the spill provide bird nesting habitat. The Trustees determined that response activities would have adversely affected the utilization of sand beach habitat by nesting birds; and that this adverse effect occurred at some level for all types of response actions, because of nesting birds' sensitivity to human disturbance. The nature of the impacts of response activities to beach-nesting birds was assessed by comparing the types of response activities that occurred on sand beaches to information in the literature on the impacts that comparable types of physical disturbances have had on bird nesting success. An illustrative analysis was performed for sand beaches in Barataria and Terrebonne Bays, Louisiana, and for selected representative bird species (see text box).

Louisiana beach-nesting bird species included in the analysis were:

- Snowy plover
- American oystercatcher
- Gull-billed tern
- Black skimmer
- Wilson's plover
- Brown pelican
- Sandwich tern
- Least tern

Response Information

The information on response activities used in the analysis included:

- The frequency of visits to beaches during nesting season (April to July).
- The duration of visits.
- The number of workers present for visits.
- The nature of activities undertaken during the visit.

This information was obtained from the same types of response records described in Section 4.6.6.2 (Exposure to Oil and Response Activities).

Types of Disturbances That Impact Nesting

The Trustees' literature review revealed that relatively minor human disturbances can result in measurable decreases in nesting and reproductive success for the bird species considered in the Trustees' illustrative analysis. For example, pedestrians walking by nests, boats driving by beaches, and vehicles driving on beaches have all been shown to be associated with nest abandonment, destruction of nests, and egg and chick mortality (see text box below). These types of activities are comparable to certain *Deepwater Horizon* response activities, such as manual beach cleanup operations and crews patrolling beaches looking for oil.

Adverse effects to eggs and chicks due to pedestrians and boating by beaches as reported in the literature:

- Direct destruction of eggs in nests by pedestrians walking on them
- Increased egg mortality
- Nest and colony site abandonment
- Reduced fledgling success
- Increased population of egg and chick predators
- Increased chick mortality
- Reduced time incubating causing increased predation or destruction of eggs

Adverse effects to eggs and chicks due to vehicles on beaches as reported in the literature:

- Increased egg mortality
- Increased chick mortality
- Reduced fledgling success
- Nest and colony site abandonment
- Reduced time incubating causing increased predation or destruction of eggs

Sources*: Anderson (1988); Anderson and Keith (1980); Cowgill (1989); George (2002); Lafferty et al. (2006); McGowan and Simons (2006); Ruhlen et al. (2003); Sabine et al. (2006); Safina and Burger (1983); Schulte and Simons (2014); Toland (1999); Virzi (2010).*

A review of the bird nesting literature further revealed that all bird species in the Trustees' analysis are tenacious nest attenders, seldom leaving the nest during daylight hours, unless disturbed (McGowan & Simons 2006; Molina et al. 2014; Page et al. 2009). This behavior protects eggs from predators and cools eggs on hot days. However, if a disturbance occurs and the adult birds depart, the nest is left unattended. Using equations available from the literature (Westmoreland et al. 2007), the Trustees estimated that a 4 gram egg left unattended and exposed under the summer sun would overheat and

lose viability after approximately 1.5 hours (see Ritter et al. (2015) for a description of these calculations). This is the egg size of most birds included in the illustrative analysis, including Wilson's and snowy plovers, the tern species, and American oystercatchers. An analysis of the response records showed that a significant proportion of cleanup activities on sand beaches were longer than the duration (1.5 hours) over which eggs lose viability in elevated temperatures. Adult birds flushed from their nests during such response activities would therefore have been sufficient to overheat eggs.

Frequency of Disturbances That Impact Nesting

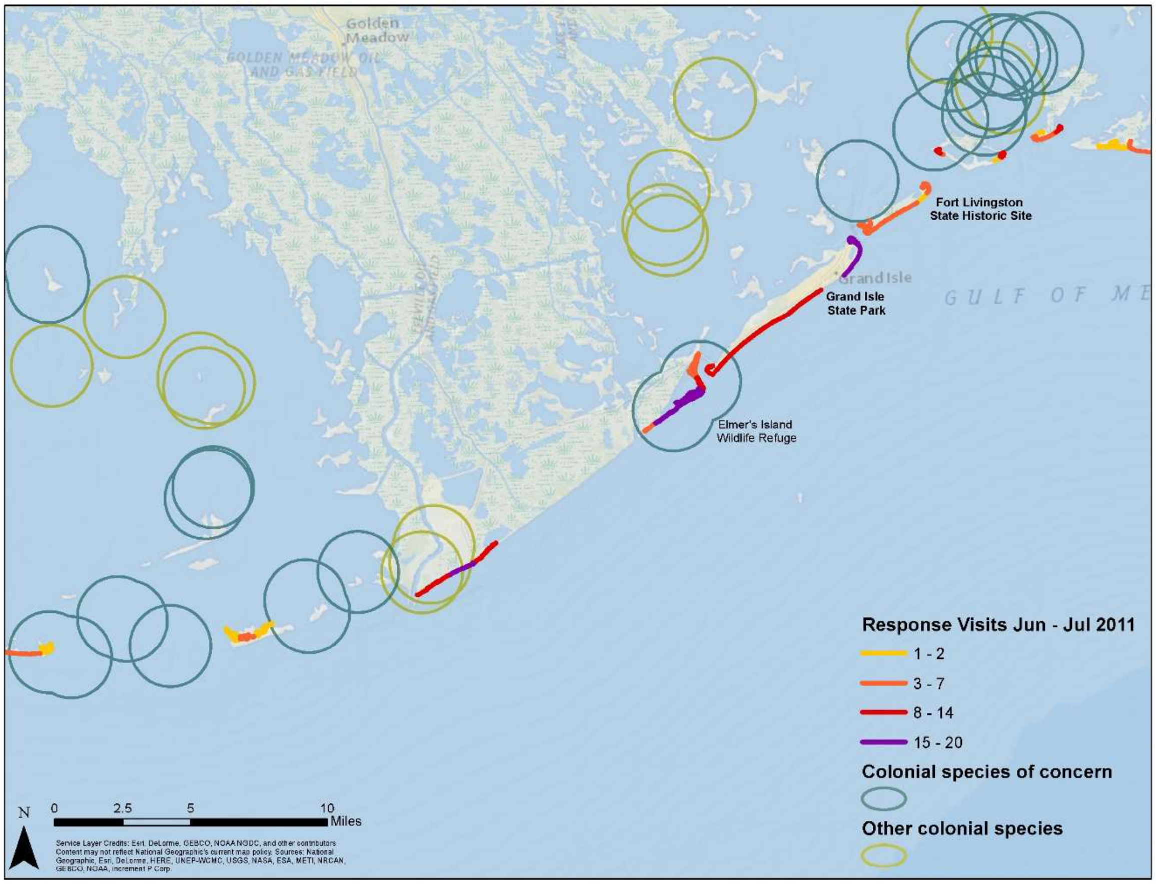
Further, even small numbers of response activity visits on a given beach could have pronounced negative effects on the reproduction of some species, if the activities took place during nesting season (April to July). After a nest failure, certain bird species—American oystercatchers, snowy plover, and brown pelican—may not re-attempt to nest or will do so only once or twice (American Oystercatcher Working Group et al. 2012; Page et al. 2009; Shields 2014). Ritter et al. (2015) review this literature in greater detail. Therefore, as few as one or two response visits to a beach when adult birds nest could result in the loss of the entire nesting season for some bird species.

The Trustees' Illustrative Analysis

The Trustees overlaid information on the frequency of response activities with known nesting locations on Barataria and Terrebonne Bay (the area considered in the Trustees' illustrative example). This analysis showed that many beaches with known nesting locations were disrupted multiple times during nesting seasons, with response activity visits lasting longer than an hour. Figure 4.6-59 shows an example of the overlap of nesting habitat and response activities for the 2011 nesting season in Barataria Bay. According to the map, many response visits occurred in June and July 2011, at beach habitat that is normally used by nesting birds, including state species of concern. In particular, at Elmer's island wildlife refuge, 15-20 response visits occurred during this time, which would have led to significant disruption of nesting behavior. See Ritter et al. (2015) for further details on the frequency of visits for the illustrative example.

Recognizing the potential for human disturbance to adversely impact nesting, the UC response activities were halted during nesting season on some beaches that are particularly important nesting habitat (e.g., Fourchon Beach in Lafourche Parish and Isle Dernieres in Terrebonne Bay). However, these protective measures were not in place in 2010; and when initiated in 2011, response activities were halted in sensitive habitats only for a subset of beaches where nesting and response activities overlapped. For example, response activities were not halted at Elmer's Island.

In summary, the Trustees determined that response activities had an adverse impact on sand beaches used as bird nesting habitat. Even intermittent and minor activities, such as manual cleanup using hand-held tools, potentially had a significant impact on sand beach functionality as nesting habitat.



Source: Ritter et al. (2015).

Figure 4.6-59. This figure shows the overlap between the locations of nesting colonies in Barataria Bay and the locations of response activities on sand beaches during a part of the 2011 nesting season (the months of June and July). Bird species include brown pelican, black skimmer, least tern, and Sandwich tern. The map shows that there were many response activity visits to beaches where birds nest (or attempt to nest). For example, between 15–20 visits to Elmer's Island occurred during this time. The published literature suggests that these visits would have had significant adverse impacts to birds attempting to nest, even if they involved "minor" response activities that did use heavy equipment.

4.6.6.4 Injury Quantification

This section presents the Trustees' quantification of injury to sand beaches. The Trustees' injury determination focused on the sand beach macrofauna community and was supported by the beach-nesting birds analysis. For injury quantification, the Trustees focused only on the impact of oil exposure and response-related disturbances to macrofauna.

The quantification was made for each beach segment where response activity occurred. Each segment was assigned an RI category. RI categories are briefly described in Figure 4.6-57 and more thoroughly described in (Michel et al. 2015). For each segment, injury was computed on monthly intervals. The computed injury accounted for the impact of each response visit that occurred, until conditions recovered to what would be expected had the *Deepwater Horizon* spill not occurred. Step 3 in Figure

4.6-54 outlines the injury quantification process used by the Trustees. The total quantification of length and area of injured sand beach habitat, by RI category, is provided in Table 4.6-22.

4.6.6.4.1 Injury Quantification in the Intertidal Habitat

As described above, based on a review of the literature, the Trustees determined that heavier oiling exposure would have likely resulted in a 95 percent decrease in macrofauna abundance and species composition and that lighter oiling would have likely resulted in a 40 percent reduction. The Trustees developed four different recovery rates for the intertidal community, depending upon the severity of the oiling, the severity of response activities (RI categories), and the proximity to healthy source populations (i.e., the sources of recruitment of new macrofauna to injured beaches). The development of these recovery rates is described in further detail in Michel et al. (2015).

The fastest recovery was assumed for those beaches to the east end of the Gulf of Mexico, where lighter oiling occurred and less intensive treatments were applied. The slowest recovery was assumed for the heavily oiled and isolated beaches west of the Mississippi River mouth. Recovery would occur by 4 to 6 years after the last response action for the heavier oiled beaches and from 2 to 3 years for the lighter oiled beaches. In other words, for Louisiana heavier oiled beaches where response activities continued into 2014, recovery is estimated to occur by 2020. Each segment was assigned a rate of recovery. However, every time a response activity occurred in any given beach segment, the recovery was reduced by the appropriate percent for that month in the Trustees' quantification analysis (Michel et al. 2015).

4.6.6.4.2 Injury Quantification in Supratidal Habitat

For supratidal zone communities, impacts on heavier oiled beaches could not be separated from the response injury because, in the first few months of oil removal activities, essentially all wrack was bagged and removed from the beach, whether individual patches of wrack were oiled or not (Michel et al. 2015). These beaches therefore experienced a 100 percent reduction in wrack-associated macrofauna. On lighter oiled beaches, the Trustees assumed that much less wrack was removed. Accordingly, the wrack community was reduced less, and it was assumed to be reduced by a factor of 20 percent.

Recovery in the supratidal zone occurred only after wrack arrived and persisted on beaches and established wrack-associated communities. These processes were estimated to take 3 years on heavier oiled beaches and 1 year on lighter oiled beaches (see Michel et al. (2015) for more details on the recovery rates and how they were set). Most of the oil in the supratidal zone was quickly buried, which triggered the need for its removal by sifting, excavation, tilling, and other processes, particularly on amenity beaches. Thus, in the Trustees' analysis, if response activities were conducted in the supratidal zone during the recovery period, the recovery would be reduced by the appropriate percent for each month when activity occurred (Michel et al. 2015).

In summary, the Trustees quantified the degree of injury in terms of length and area of injured sand beach habitat. Quantification was based on information on the degree of oil exposure and response-related disturbances for every beach response segment by month and recovery rates presented by Michel et al. (2015). Table 4.6-18 summarizes the oil exposure, and Table 4.6-22 shows the lengths and areas of the five different Response Injury levels for each state and for federal lands. In both tables, the

areas include supratidal and intertidal zones. These data represent the minimum injury; there were many gaps in the available documentation, particularly for mechanical treatment conducted prior to 2011 and for the locations of staging areas throughout all states.

The intensity and duration of response actions on sand beaches varied by state, year, season, and even by times of day. Most operations were manually conducted up until the fall of 2010/2011. After that, buried oil at selected Alabama and Florida beaches were removed with heavy equipment, with names such as “Big Dig” and “Deep Clean.” In 2010, sifting operations were conducted at night when temperatures were cooler and the oil was less likely to clog sieves; these operations would have added impacts to nocturnal animals, such as ghost crabs, beach mice, and sea turtles. In some instances, merely getting large equipment to and from the beach destroyed dune vegetation along the route (Wetland Sciences Inc. 2014).

Most beaches on Florida state lands underwent intensive manual removal, though up to 10 kilometers (6 miles) received some type of mechanical treatment. In Alabama, at least one-third of beaches were mechanically treated, and barriers were placed on Dauphin Island. In Mississippi, more than 40 percent of beaches received some type of mechanical treatment. Louisiana beaches experienced the most intensive response actions, where at least 74 kilometers (46 miles) had very intensive mechanical treatment over a 4-year period. For example, 54,534 kilograms (120,226 pounds) of oiled materials—the equivalent of 18 dump trucks—were removed from the easternmost end of Fourchon Beach in December 2013. Part of the back beach and tidal flat on the south side of South Pass was dredged to remove buried oil. Oiling and cleanup operations were conducted over two months (July and August) in the summer of 2010 along 57 kilometers (35 miles) of the Texas Gulf coast. Operations were generally limited to light manual operations with approximately 7,917 kilograms (17,454 pounds) of oiled waste materials removed. Response injury was not assessed for Texas due to the short duration of operations.

Table 4.6-22. Length and area of response injury by state including federal lands (DOI, DOD). Injury quantification as presented in Michel et al. (2015). For DOD lands, response injury was developed only for affected Florida beaches; no DOD lands with sand beach were impacted in the other Gulf states. The top table presents beach length and area in miles and acres; and the bottom table presents these data in kilometers (km) and hectares (ha).

	Louisiana		Mississippi		Alabama		Florida		Total Miles	Total Acres
State Land	**Miles**	**Acres**	**Miles**	**Acres**	**Miles**	**Acres**	**Miles**	**Acres**		
RI 1	38	498	0	0	0	0	0	0	38	498
RI 2	37	623	27	437	42	550	35	1,580	141	3,190
RI 3	16	423	19	427	11	248	3	137	48	1,235
RI 4	10	306	0	0	14	394	3	166	27	867
RI 5	20	596	0	0	0	0	0	0	20	596
Barriers	8	189	0	0	4	83	0	0	286	6,657
TOTAL	**129**	**2,636**	**46**	**863**	**70**	**1,275**	**41**	**1,884**	**286**	**6,657**
DOI Land										
RI 1	12	363	0	0	0	0	0	0	12	363
RI 2	0	0	42	744	6	93	34	2,035	81	2,872
RI 3	0	0	15	593	4	120	22	1,648	41	2,361

RI 4	0	0	0	0	1	22	9	461	1	484
RI 5	0	0	0	0	0	0	0	0	0	0
TOTAL	**12**	**363**	**57**	**1,338**	**10**	**235**	**65**	**4,144**	**144**	**6,080**
DOD										
RI 1	-	-	-	-	-	-	0	0	0	0
RI 2	-	-	-	-	-	-	5	47	5	47
RI 3	-	-	-	-	-	-	0	0	0	0
RI 4	-	-	-	-	-	-	0	0	0	0
RI 5	-	-	-	-	-	-	0	0	0	0
TOTAL	**-**	**-**	**-**	**-**	**-**	**-**	**5**	**47**	**5**	**47**

	Louisiana		Mississippi		Alabama		Florida		Total km	Total ha
State Land	**km**	**ha**	**km**	**ha**	**km**	**ha**	**km**	**ha**		
RI 1	61	202	0	0	0	0	0	0	61	202
RI 2	60	252	43	177	68	223	56	639	227	1291
RI 3	26	171	31	173	18	100	5	55	77	500
RI 4	16	124	0	0	23	159	5	67	43	351
RI 5	32	241	0	0	0	0	0	0	32	241
Barriers	13	76	0	0	6	34	0	0	460	2694
TOTAL	**208**	**1067**	**74**	**349**	**113**	**516**	**66**	**762**	**460**	**2694**
DOI Land	0	0	0	0	0	0	0	0	0	0
RI 1	19	147	0	0	0	0	0	0	19	147
RI 2	0	0	68	301	10	38	55	824	130	1162
RI 3	0	0	24	240	6	49	35	667	66	955
RI 4	0	0	0	0	2	9	14	187	2	196
RI 5	0	0	0	0	0	0	0	0	0	0
TOTAL	**19**	**147**	**92**	**541**	**16**	**95**	**105**	**1677**	**232**	**2460**
DOD	0	0	0	0	0	0	0	0	0	0
RI 1	-	-	-	-	-	-	0	0	0	0
RI 2	-	-	-	-	-	-	8	19	8	19
RI 3	-	-	-	-	-	-	0	0	0	0
RI 4	-	-	-	-	-	-	0	0	0	0
RI 5	-	-	-	-	-	-	0	0	0	0
TOTAL	**-**	**-**	**-**	**-**	**-**	**-**	**8**	**19**	**8**	**19**

4.6.6.5 Inferences

Determining and quantifying injury for sand beach habitat was largely based on a literature review on impacts of oil spills and cleanup activities to sand beach macrofauna. While scientifically robust, these studies were fairly limited in number, and were focused on spills of smaller magnitude and shorter intensity than the *Deepwater Horizon* oil spill. While the Trustees evaluated impacts to the sand beach macroinvertebrate community, disturbance to nesting shorebirds habitat on sand beaches could have been carried into higher trophic levels, affecting foraging and reproductive success. Even relatively

minor levels of response activity would have disrupted the use of sand beach habitat by nesting shorebirds that are known to be sensitive to physical disturbances. The Trustees determined that a potential loss of sand beach functionality for shorebird nesting habitat occurred as a result of response activities during the nesting season. This impact, while not fully quantified, would likely increase the overall magnitude of the response injury in shorebird nesting habitat.

Collectively, the Trustees determined that the *Deepwater Horizon* oil spill and subsequent response actions resulted in significant loss in ecosystem function of sand beach habitats in the northern Gulf of Mexico and that substantive restoration activities will be required to compensate for the loss.

4.6.6.6 Conclusions and Key Aspects of the Injury for Restoration Planning

Sand beaches were extensively injured as a result of the *Deepwater Horizon* oil spill and subsequent response actions. The Trustees considered the totality of the injury in planning restoration to offset the losses. In particular, key aspects of the injury to sand beaches that informed the Trustees' comprehensive restoration planning include:

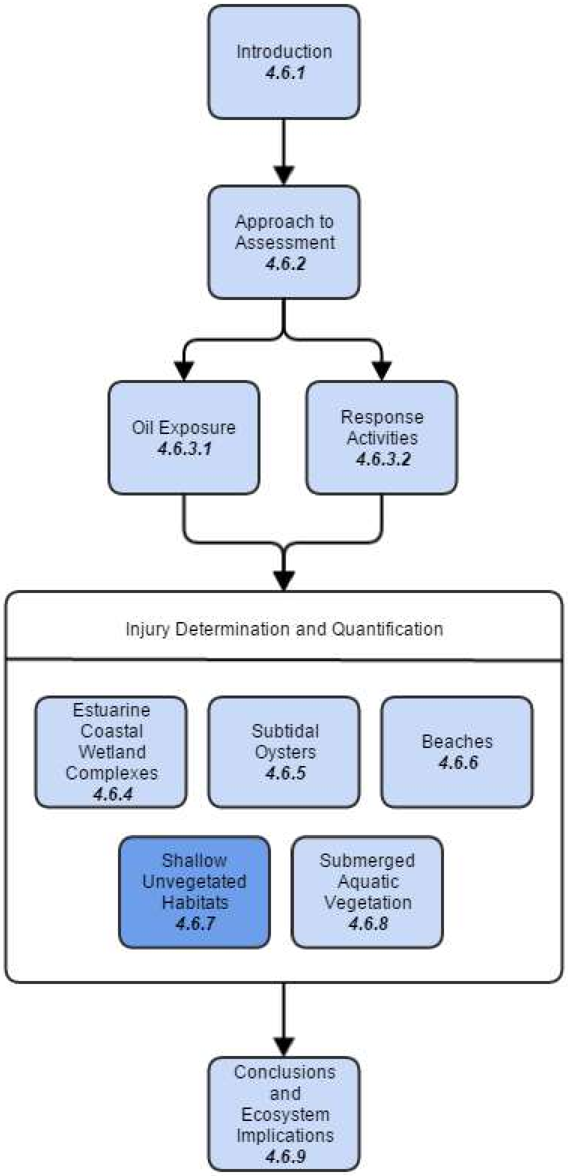
- 965 kilometers (600 miles) of sand beach and dune habitat along shorelines and back barrier islands across the northern Gulf of Mexico were injured as a result of the *Deepwater Horizon* oil spill.
- Injuries resulted from a combination of the direct effects of oil and ancillary adverse impacts of response activities undertaken to clean up the oil. Injuries included reduced abundance of crabs, amphipods, insects, and other macrofauna that live in the sand and wrack (decomposing vegetation that serves as habitat and food source for many beach organisms). Injuries also included impacts to other biota (e.g., beach mice) and a disruption of bird-nesting habitat.

As described in Chapter 5 (Sections 5.5.2 and 5.5.3), the Trustees have identified a suite of restoration approaches to offset these losses, including injuries that occurred on federally managed lands.

4.6.7 Shallow Unvegetated Habitats—Gulf Sturgeon Assessment

Key Points

- Gulf sturgeon is a threatened species and listed under the Endangered Species Act of 1973.
- Their decline was likely caused by overexploitation and exacerbated by habitat destruction, water quality deterioration, and other factors.
- Trustees estimated that between 1,100 and 3,600 Gulf sturgeon were potentially exposed to oil.
- Gulf sturgeon would likely be very slow to recover from additional challenges such as an oil spill.



Nearshore benthic species (e.g., oysters, shrimp, flounder, amphipods) living adjacent to marsh and beach shorelines were addressed in prior sections because of their role in edge communities and their utility in assessing impacts to shoreline habitat. Shallow unvegetated habitats were evaluated using the Gulf sturgeon as an indicator species. Section 4.5, Benthic Resources, discusses spill-related injuries to habitat and species occurring on the continental shelf from approximately 10 to 200 meters water depth.

The Gulf sturgeon (*Acipenser oxyrinchus desotoi*) is an anadromous fish that migrates from salt water into large coastal rivers to spawn. The U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NOAA Fisheries) designated the Gulf sturgeon to be a threatened species in 1991 under the Endangered Species Act of 1973, as amended (FWS & NOAA 1991). Their decline was likely caused by overexploitation and exacerbated by habitat destruction, water quality deterioration, and other factors (FWS & GSMFC 1995).

The historic range of the Gulf sturgeon extended from the Mississippi River to Tampa Bay (FWS & NOAA 2003a). However, their current range is from Lake Pontchartrain in Louisiana to the Suwannee River in

Florida. During the spring, adult and sub-adult Gulf sturgeon migrate from the northern Gulf of Mexico or nearshore bays into coastal rivers, where sexually mature sturgeon spawn. The major river systems that are known to continue to support reproducing subpopulations include (from west to east): the Pearl, Pascagoula, Escambia, Blackwater, Yellow, Choctawhatchee, Apalachicola, and Suwannee Rivers (FWS & NOAA 2003a). Rather than returning to marine waters after spawning, fish remain in rivers throughout summer. During the cooler months, from October through April, Gulf sturgeon adults move along the coast in nearshore waters less than 10 meters (33 feet) deep (FWS & NOAA 2003a). In 2003, the USFWS and NOAA Fisheries designated 14 geographic areas in the northern Gulf of Mexico, and its rivers and tributaries, as critical habitat for the Gulf sturgeon (Figure 4.6-60).

Gulf sturgeon are bottom feeders, eating primarily small invertebrates in the sediment. The type of invertebrates they ingest vary, but are mostly soft-bodied animals that occur in sandy substrates. Many reports indicate that adult and sub-adult Gulf sturgeon lose a substantial percentage of their body weight while in freshwater (Mason Jr. & Clugston 1993; Wooley & Crateau 1985) and then compensate the loss during winter-feeding in the estuarine and marine environments (Wooley & Crateau 1985).

4.6.7.1 Approach to the Assessment

In the Gulf sturgeon NRDA assessment, the Trustees integrated field and laboratory approaches to determine exposure and injuries of the threatened Gulf sturgeon in shallow unvegetated habitats (FWS 2015). Section 4.6.7.1.1 describes the field-based assessment and Section 4.6.7.1.2 describes the laboratory study.

4.6.7.1.1 Field-Based Assessment

The Trustees monitored wild Gulf sturgeon in the northern Gulf of Mexico to document exposure and assess injuries from the *Deepwater Horizon* oil spill. The assessment targeted the eight river systems listed above. The capture and sampling of adult Gulf sturgeon were completed in fall 2010 and 2011 during outmigration and spring 2011 and 2012 during inmigration. Captured Gulf sturgeon were measured and weighed, and blood samples were collected and analyzed for DNA fragmentation, hematology (e.g., white blood cells, neutrophils, thrombocytes, and red blood cells), cell cycle, DNA repair protein, and other parameters.

To track the movement and residency patterns of Gulf sturgeon in the northern Gulf of Mexico, an ultrasonic acoustic transmitter was surgically implanted into the gastric cavity of adult fish from each of the eight river systems during the fall outmigration in 2010 and 2011. In addition, an extensive network of receivers deployed at the mouths of the eight rivers and bays and throughout the nearshore area of the Gulf coast monitored fish movements. Overall, 270 transmitters were implanted in adult sturgeon.

4.6.7.1.2 Laboratory Toxicity Study

To better understand injuries shown by the analytical results for the field collected blood samples, controlled exposures of shovelnose sturgeon (*Scaphirhynchus platorynchus*)—a surrogate for the Gulf sturgeon and a closely related species—were conducted for comparison. Juvenile shovelnose sturgeon were exposed to *Deepwater Horizon* weathered oil through a high-energy water-accommodated fraction (HEWAF-MTS) at a TPAH50 concentration range of 5-10 µg/L for 7 or 28 days (FWS 2015). Potential effects were investigated at biochemical, cellular, and organ levels. Endpoints included organ

weights (liver and spleen), histology, EROD activity/CYP450, DNA fragmentation, DNA repair, blood cell counts, and gene expression.

4.6.7.2 Pathways for Oil and Response Actions to Affect Gulf Sturgeon

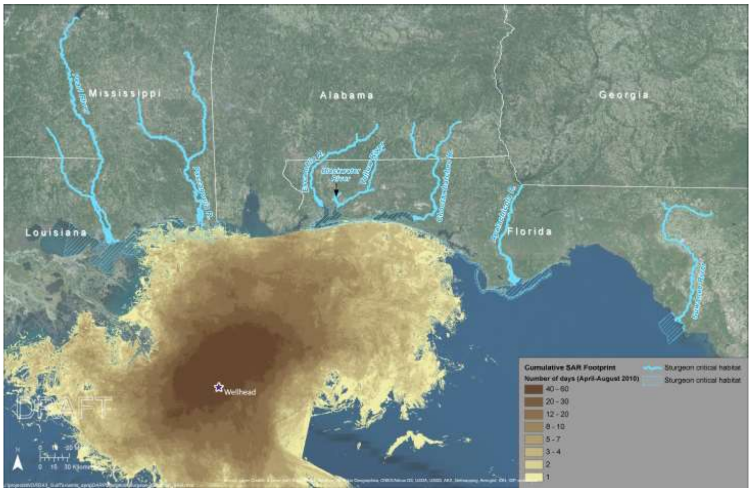
As oil was released into the open waters of the northern Gulf of Mexico, ocean currents and wind transported the oil to nearshore habitats, including beaches and marsh edges. Preliminary oiling occurred during spring and summer 2010, which led to dissolved, entrained, and submerged oil in the nearshore environment (Section 4.2, Natural Resource Exposure) utilized by Gulf sturgeon as overwintering habitat (FWS 2015; FWS & NOAA 2003a). Remote sensing and SCAT field assessments documented oil extending to the nearshore environment in Louisiana, Mississippi, Alabama, and western Florida (Section 4.2, Natural Resource Exposure); and oiling in nearshore environments intersected with Gulf sturgeon critical habitat (Figure 4.6-60). Submerged oil mats were also recorded along the shorelines of Louisiana, Mississippi, Alabama, and Florida (Hayworth et al. 2011; OSAT-2 2011; OSAT-3 2013), with observations of buried oil as late as September 2011 (Hayworth et al. 2011).

The sturgeon's freshwater residency during the summer of 2010 provided temporary refuge from exposure to oil associated with the *Deepwater Horizon* incident. However, upon their outmigration beginning in October 2010, Gulf sturgeon from six of the eight natal river systems were found within the area of the northern Gulf of Mexico impacted by the *Deepwater Horizon* oil spill (FWS 2015). As evident in the telemetry data, Gulf sturgeon migrated into nearshore northern Gulf waters during fall 2010, staying until spring 2011; and migrated again during fall 2011, staying in the northern Gulf of Mexico until spring 2012 (Figure 4.6-61; FWS 2015). The first dates of northern Gulf of Mexico entry by Gulf sturgeon were typically in late October or early November. They remained in the northern Gulf until they returned to rivers between March and May.

To determine exposure in the absence of quantitative, comprehensive substrate analyses or chemical body burdens for benthic macroinvertebrate prey items, the Trustees relied on available oil distribution data from before and during the residency time for Gulf sturgeon in the northern Gulf of Mexico (FWS 2015). Surface oiling data, collected by Synthetic Aperture Radar (SAR), were available during the time preceding (April 23, 2010, to August 11, 2010) the emergence of Gulf sturgeon into the northern Gulf, while the SCAT shoreline oiling data were available for almost 4 years following the spill (May 2010, to April 2014) (Section 4.2, Natural Resource Exposure; FWS (2015)). Shoreline oiling was observed during the Gulf sturgeon's residency in the northern Gulf of Mexico over the fall and winter months. Based on SCAT data, oiling of the shorelines and barrier islands of Louisiana, Mississippi, Alabama, and western Florida was documented from October 2010, to March 2011 (FWS 2015). Sturgeon telemetry data during the same time period showed the close proximity of Gulf sturgeon to the oiled shoreline (Figure 4.6-62; FWS (2015)).

Using the SAR surface oil footprint as an indication of potential exposure, the Trustees calculated the percent of time that tagged individuals were detected within the area of oiling (FWS 2015). For the purposes of this assessment, the three conditions considered to constitute potential oil exposure for each fish were: (1) telemetry readings must be within 1 kilometer of recorded surface oil as represented by SAR; (2) 10 percent or more of telemetry readings were recorded in the oiled area (defined in 1); and (3) at least 24 hours of exposure were recorded. The telemetry and oil distribution data were compared

to sturgeon river populations to estimate what percentage of each Gulf sturgeon river population was present in the area of oiling and were therefore potentially exposed to that oil. Based on this analysis, a substantial fraction of the tagged individuals from six of the eight river populations (Pearl, Pascagoula, Escambia, Blackwater, Yellow, Choctawhatchee) were found to reside in the area of oiling during the fall and winter months (FWS 2015). Extrapolated to Gulf sturgeon river populations, this indicates a large number of Gulf sturgeon potentially exposed to DWH oil.



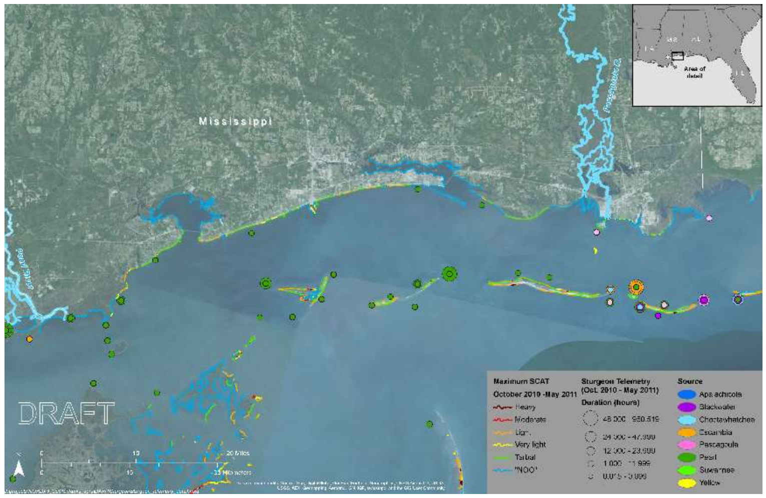
Source: *FWS (2015).*

Figure 4.6-60. Cumulative SAR footprint from April to August 2010. The 14 geographic areas shown in light blue are designated as Gulf sturgeon critical habitat, including rivers, estuaries, and marine waters.

Source: *FWS (2015).*

Figure 4.6-61. Distribution and residence time of Gulf sturgeon from fall, 2010, to spring 2011, by source river.

4.6.7
Shallow Unvegetated Habitats—Gulf Sturgeon Assessment



Source: *FWS (2015).*

Figure 4.6-62. Maximum SCAT from October 2010, to May 2011, overlaid with sturgeon telemetry data from the same time period. Source rivers are shown in light blue.

4.6.7.3 Injury Determination: Effects of Oil on the Resource

Analytical samples taken from Gulf sturgeon in the river populations showed increased incidence of DNA fragmentation and up-regulation of DNA repair proteins between the spring in-migrant and the fall ex-migrants (FWS 2015). Additionally, immunological responses were observed in fish that were potentially exposed to *Deepwater Horizon* oil during this same time period—winter northern Gulf of Mexico residency in late 2010 and early 2011. Subsequent laboratory experiments and additional analyses on field blood samples also provided evidence of genotoxicity and immunosuppression at the molecular, cellular, and organ levels (FWS 2015). Altogether, these findings lead to the conclusion that Gulf sturgeon potentially exposed to *Deepwater Horizon* oil in the northern Gulf displayed both genotoxicity and immunosuppression, which can lead to malignancies, cell death, susceptibility to disease, infections, and a decreased ability to heal (FWS 2015). Since large numbers of fish from most Gulf sturgeon river populations were potentially exposed to *Deepwater Horizon* oil, an important number of these federally protected species was affected (FWS 2015).

4.6.7.4 Injury Quantification

Using the surface oil footprint as an indication of potential exposure, the Trustees estimated that between 1,100 and 3,600 Gulf sturgeon were potentially exposed to *Deepwater Horizon* oil in the nearshore areas of the northern Gulf of Mexico (as defined in Section 4.6.7.2 above; FWS (2015)). Moreover, this estimated range of Gulf sturgeon represented a large proportion of the populations from six of the eight natal river systems. The estimated percent of populations potentially exposed, excluding the Suwannee and Apalachicola from where no fish were found in the area of oiling, ranged from 27 and 100 percent. Overall, more than six out of every ten (63 percent) fish from these six populations (Pearl, Pascagoula, Escambia, Blackwater, Yellow, Choctawhatchee) were potentially exposed to oil from the *Deepwater Horizon* spill (FWS 2015).

4.6.7.5 Recovery

Based on the Trustees' assessment, between 1,100 and 3,600 Gulf sturgeon were estimated to be exposed to *Deepwater Horizon* oil in the nearshore areas of the northern Gulf of Mexico, representing a large proportion of the populations from six of the eight river systems occupied by Gulf sturgeon (FWS 2015). This species' exposure to oil likely resulted in genotoxicity and immunosuppression, as supported by field observations and laboratory studies (FWS 2015).

As discussed above, the Gulf sturgeon is a threatened species under the Endangered Species Act of 1973. Given the listed status and existing threats to Gulf sturgeon populations (FWS & NMFS 2009), this species would likely be very slow to recover from additional stressors, such as an oil spill. However, the scant information on long-term impacts of genotoxicity and immunosuppression on this species' fitness, the influence of non-oil spill related factors impacting these populations, and the relatively long life history of Gulf sturgeon suggest that recovery of Gulf sturgeon without restoration could take several decades or more.

4.6.7.6 Conclusions and Key Aspects of the Injury for Restoration Planning

In the early 1900s, Gulf sturgeon populations were reduced dramatically as they were exploited for their meat and caviar (FWS & GSMFC 1995). The species was further impacted by the construction of dams on rivers, which blocked the fish from reaching their historical spawning sites (FWS & GSMFC 1995). Water pollution and loss of habitat have also had adverse impacts (FWS & GSMFC 1995).

The continued existence of this threatened species depends on maintaining and protecting important riverine and marine habitats. As an anadromous species, the Gulf sturgeon relies on two distinctly different habitats (FWS & NOAA 2003a). During the winter months, the species depends heavily on the food resources and sandy substrates in the sediments of the Gulf of Mexico to feed and grow (FWS & NOAA 2003a). During the spring months, Gulf sturgeon migrate up rivers to reach their spawning grounds (FWS & NOAA 2003a).

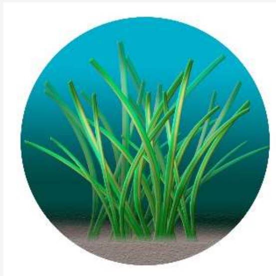
The Trustees considered all aspects of the Gulf sturgeon injury assessment in planning restoration for this endangered species. Key points that informed the Trustees' restoration planning include:

- The Trustees conducted a focused assessment of potential injuries to Gulf sturgeon (*Acipenser oxyrinchus desotoi*), because Gulf sturgeon are listed as a threatened species under the Endangered Species Act and inhabit areas exposed to *Deepwater Horizon* oil.
- Between 1,100 and 3,600 Gulf sturgeon were estimated to be exposed to *Deepwater Horizon* oil in the nearshore areas of the northern Gulf of Mexico in the fall of 2010 (FWS 2015). This represents a large proportion of the populations from six of the eight natal rivers systems. Although a direct kill of Gulf sturgeon from the oil was not observed, the Trustees found evidence of physiological injury. This evidence includes exposure biomarkers for DNA damage and immunosuppression between Gulf sturgeon that were—and were not—exposed to the oil (FWS 2015).

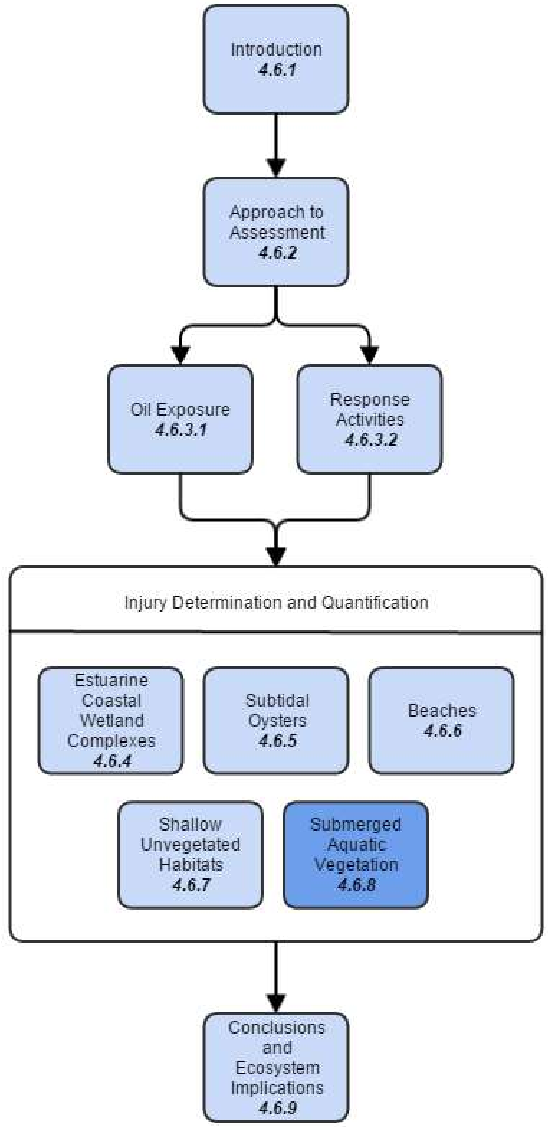
As described in Chapter 5 (Section 5.5.7), the Trustees have identified restoration approaches for this threatened species that emphasize spawning habitat and reproductive success.

4.6.8 Submerged Aquatic Vegetation Assessment

Key Points



- The Trustees conducted a series of field assessment studies to evaluate three broad categories of injuries due to oil exposure, physical response activities, and summer freshwater releases.
- A total of 271 acres (110 hectares) of seagrass were lost in the Chandeleur Islands due to oil.
- The Trustees documented 876 square meters of scars and blowholes in Florida seagrass beds from 16 scars due to physical response activities.
- A total of 50 acres (20 hectares) of SAV was lost along the Lake Cataouatche shoreline in Jean Lafitte National Historical Park and Preserve due to summer river water releases as part of response actions.
- Considerations are provided for restoring SAV in the unique areas impacted.



SAV resources are a vital component of coastal aquatic ecosystems in the northern Gulf of Mexico, which has at least 26 species of SAV growing in fresh, brackish, and saline coastal environments (Cosentino-Manning et al. 2015). SAV that grows in saline environments is called seagrass. SAV is among the most productive primary producers in the biosphere. In the northern Gulf of Mexico coastal ecosystems, SAV provides a wide range of ecological services rivaling or, in some instances, exceeding the functions of tropical rain forests and coral reefs (Barbier et al. 2011; Orth et al. 2006; Rasheed et al. 2006). SAV and its epiphytic communities produce large quantities of organic matter that form the structural habitat and biochemical basis of a diverse food web leading to high secondary production rates of ecologically important and commercially valuable fish, shellfish, and wildlife communities

(Borowitzka et al. 2006; Ogden & Zieman 1977). SAV primary production also maintains good water quality by recycling and temporarily storing nutrients, filtering the water column, dissipating wave and current energy, and stabilizing sediments (Romero et al. 2006; Zieman 1982; Zieman & Zieman 1989).

SAV are rooted vascular plants that are physically and chemically integrated with the sediments they grow in. These plants are fixed in place and are unable to actively avoid contact with submerged oil transported in the water column or deposited on and in the substrate. These characteristics make SAV vulnerable to oiling. The SAV plants in the northern Gulf of Mexico are generally distributed in water depths less than 2 meters. During low tide, SAV in shallow water can form a three dimensional canopy that occupies the entire water column, further increasing potential exposure to both surface and submerged oil. The SAV canopy and epiphytes growing on the leaves baffle water currents and wave turbulence, acting as a filter that traps and promotes deposition of suspended materials within the SAV meadow (Short et al. 2000). The plants' physical structure and associated metabolism are also key components of the biogeochemical cycling of materials between the water column and the substrate. In the substrate beneath the canopy, roots and rhizomes bind and stabilize sediments, effectively retaining and concentrating inorganic particulate material, organic matter, and any other materials susceptible to deposition (Short et al. 2000). These attributes enhance the potential for direct exposure to oil within a SAV meadow by intercepting water flow, increasing deposition, and concentrating organic and inorganic material.

Potential direct impacts of oil and dispersants on SAV range from complete mortality (Jackson et al. 1989; Sandulli et al. 1998; Scarlett et al. 2005; Thorhaug & Marcus 1987) to sublethal stress and chronic impairment of SAV and sediment metabolism and function (Hatcher & Larkum 1982; Peirano et al. 2005; Ralph & Burchett 1998). Secondary impacts can also include biophysical and chemical disturbance to sediments, microbes, microfauna, and microflora (Short et al. 1995), and the impairment and mortality of secondary producers residing in the SAV canopy and sediments (e.g., invertebrates, crustaceans, fishes, and waterfowl) (Carls & Meador 2009).

SAV are also vulnerable to physical disturbances. Simple, linear propeller scars from vessels are one example. More complex injuries arise when vessels, especially large ones powered with twin propellers, create a blowhole—a propeller washed excavation of the SAV and underlying substrate (Meehan 2015). SAV communities are also vulnerable to sustained freshwater inputs and excess inputs of nutrient runoff from coastal areas. In addition, sustained high flows can destabilize root systems of mature freshwater SAV. Potential effects of increased freshwater and nutrients include diminished water quality, eutrophication, and physical loss of unrooted plants. These changes in the abiotic habitat conditions may then result in changes in the diversity and abundance of SAV species and shifts in the extent of nuisance algal blooms (Harlin 1995). Additionally, the loss of SAV and proliferation of dense floating aquatic vegetation can result in significant habitat changes with implications for fish and wildlife (Poirrier et al. 2009).

4.6.8.1 Approach to the Assessment

The SAV assessment included a series of field assessment studies to evaluate three broad categories of injuries.

1. Oil-related injury in the Chandeleur Islands.

2. Physical response injury throughout the region.
3. Freshwater injury in Jean Lafitte National Historic Park and Preserve (JELA).

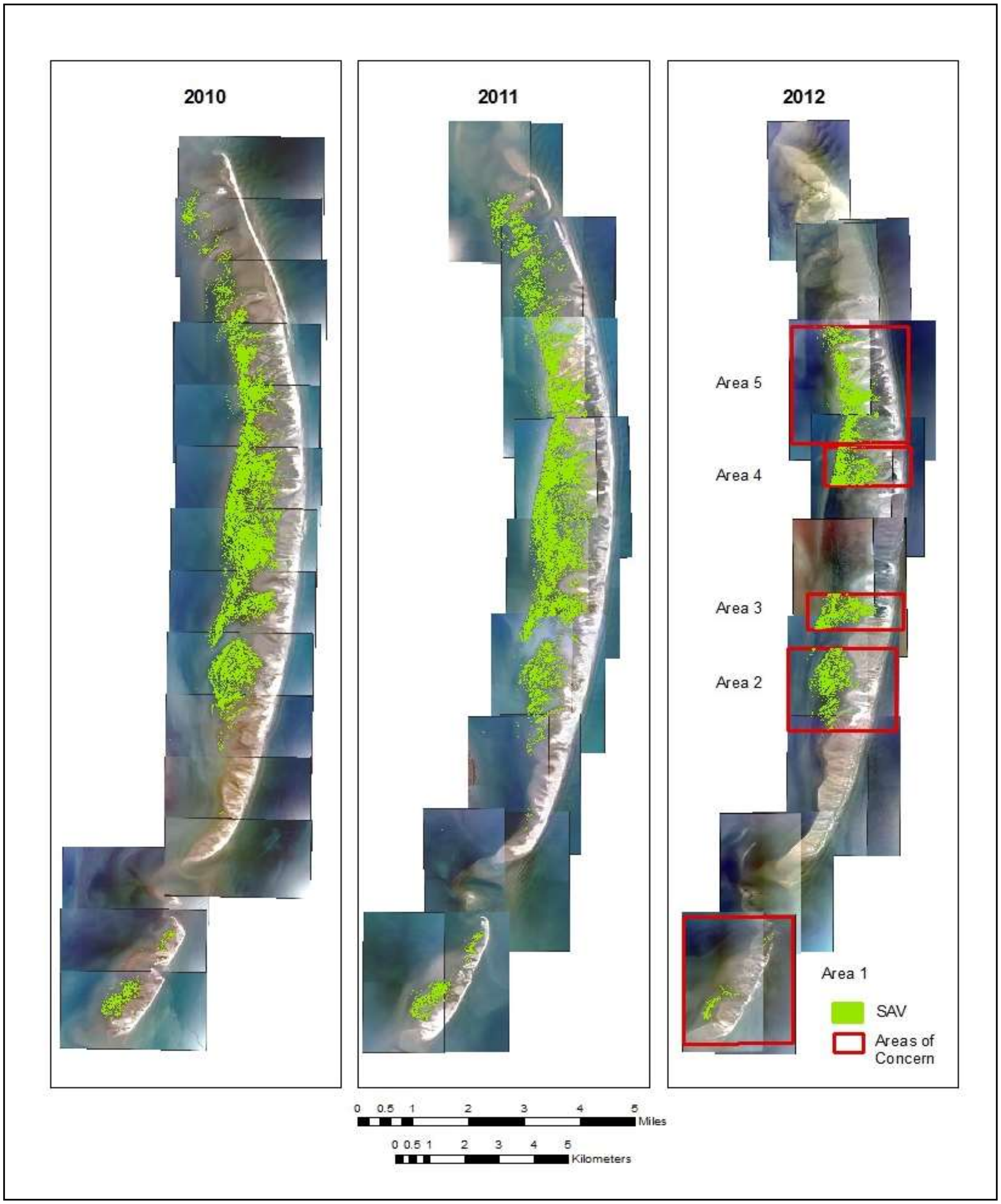
4.6.8.2 Injury Determination

4.6.8.2.1 Oil-Related Injury in Chandeleur Islands

As a result of exposure to oil in the water and sediments, the spatial distribution of seagrasses decreased from 2010 to 2012 along the shallow shelf west of the Chandeleur Islands (Figure 4.6-63) (Cosentino-Manning et al. 2015). In 2011 and 2012, seagrasses in this area were more heterogeneously distributed compared to 2010 and were present in various sized patches among gaps of unvegetated shoreline. This patchy distribution persisted despite relatively homogeneous environmental conditions and water depths suitable for growth. The heterogeneous seagrass distribution pattern was consistent with the variation in oil exposure documented by sediment and tissue samples, shoreline oiling classifications, and oil on water observations (Cosentino-Manning et al. 2015).

Oil impacts on seagrass beds throughout the region were evaluated using a three-tier study design (Cosentino-Manning et al. 2015)**.** Under Tier 1, baseline conditions were characterized before oil reached SAV beds, considering sites from Louisiana to the Florida Keys. Tier 2 characterized initial post-spill conditions through analysis of TPAH50 in sediment, seagrass tissue, and invertebrate tissue. For this effort, updated oil pathway information was used to select five sites threatened by potential exposure to oil. These included Big Lagoon, Florida; Robinson Island in Perdido Bay, Alabama; Horn Island, Mississippi; Petit Bois Island, Mississippi; and Chandeleur Islands, Louisiana. Tier 3 sampling focused on injury assessment and recovery in areas determined to be exposed to oil based on shoreline oiling classifications assessments (SCAT surveys), provisional total petroleum hydrocarbon (TPH) concentrations in sediment, and estimates of oil on surface waters using SAR and aerial imagery. Of the five Tier 2 sites, only the Chandeleur Islands were determined to be exposed to oil. These sites were further assessed during Tier 3 sampling in June 2011, which included TPAH50 analysis in sediment, seagrass tissue, and invertebrate tissue. Sampling also included field and laboratory observations of seagrass species composition, abundance, and health and condition (Cosentino-Manning et al. 2015).

Samples of sediments, seagrass tissue, and invertebrate tissue within affected seagrass beds in the Chandeleur Islands showed TPAH50 concentrations orders of magnitude higher than ambient (baseline) concentrations and forensic PAH and biomarker analyses matched with the MC252 oil from the *Deepwater Horizon* spill. In fact, almost all stranded oil samples and 70 percent of sediment samples matched MC252 oil. Concentrations of sediment TPAH50 were 8 to 12 times higher, on average, than baseline, pre-spill conditions; and SAV tissue TPAH50 concentration were 13 times higher than baseline. Elevated TPAH50 concentrations corresponded with shoreline SCAT data and SAR accumulation estimates of oil on surface water (Cosentino-Manning et al. 2015).



Source: *Cosentino-Manning et al. (2015).*

Figure 4.6-63. SAV distribution derived from fall 2010, 2011, and 2012 imagery. This time series illustrates the reduction in SAV coverage between 2010 and 2012 on the Chandeleur Islands.

In addition to the field sampling described above, seagrass distribution was mapped using an object-based image analysis method. Trustees conducted a quantitative change analysis of seagrass areal coverage using high resolution aerial imagery from fall 2010, fall 2011, and fall 2012 (Cosentino-Manning et al. 2015). The imagery analysis focused on documenting changes in seagrass coverage in five core areas of the Chandeleur Islands following exposure to MC252 oil. The areal coverage of seagrass in fall 2010 was considered baseline for injury assessment. The change analysis identified areas of seagrass loss that likely resulted from MC252 oil exposure and could not be attributed to natural processes or interpretation error. For the change analysis, the areal coverage of SAV (seagrasses) was quantitatively documented for each time interval as gains, losses, or no change in SAV. Areas were designated as "persistent loss" if seagrass was absent (no SAV) from an area for two consecutive mapping intervals (2011 and 2012) following acute exposure to oil and the initial areal mapping in 2010. A "delayed loss" classification was assigned to areas that had seagrass in 2010 and 2011, but lost seagrass in 2012 possibly due to chronic exposure.

4.6.8.2.2 Physical Response Injury

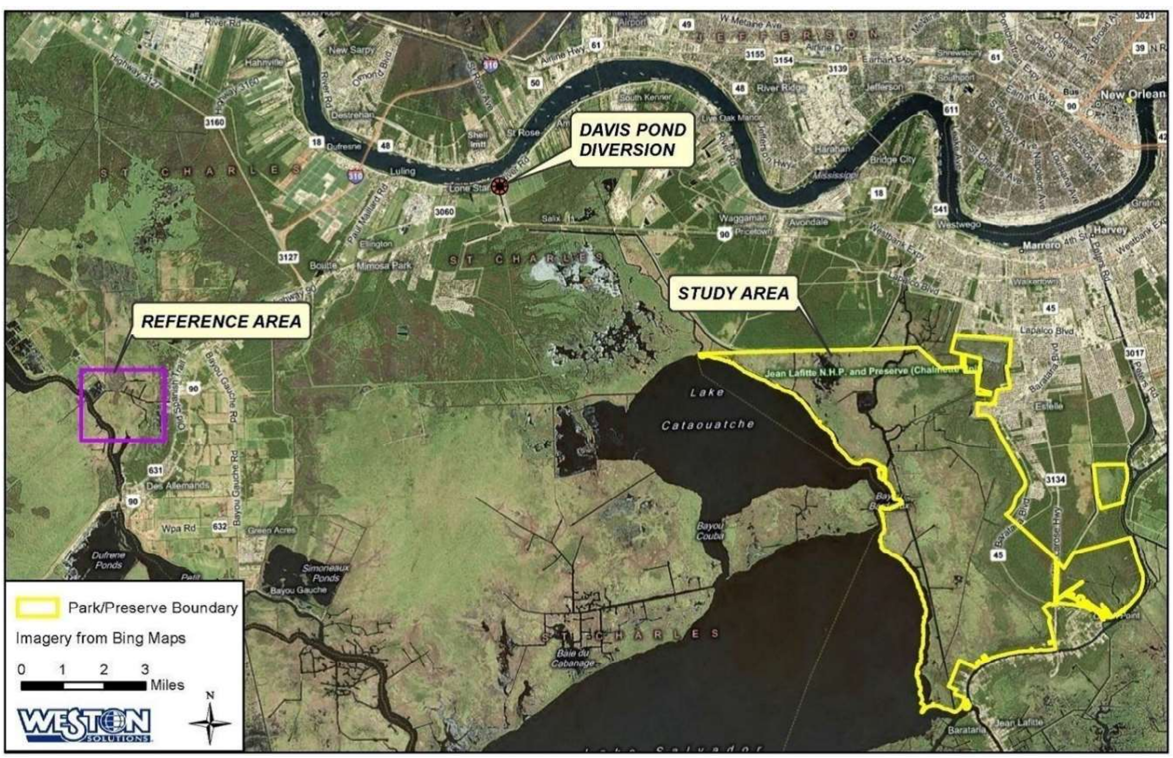
The increased vessel traffic associated with the placement of booms and berms resulted in propeller scars and blowholes in seagrass beds. Boom curtains and anchors used to hold boom in place also scoured seagrass beds. These anchors and curtains were pulled over the seagrass beds with the rising and falling tides and with wind, vessel waves/wakes, or currents (Meehan 2015). To assess impacts of physical response actions on seagrass beds, Trustees used aerial imagery taken in October 2010 from Chandeleur Islands, Louisiana, to Apalachee Bay, Florida. This imagery identified areas potentially damaged by propeller scars, booms, silt curtains, and anchors used during the oil spill response. Field surveys were conducted to verify imagery and collect more detailed information on scars and blowholes in seagrass beds. Based on these assessments, response activities resulted in a total of 73 scars and/or blowholes, as identified by aerial imagery and field surveys (Meehan 2015). Of these, 57 were less than 15 centimeters deep or were dominated by the seagrass species *H. wrightii,* and the Trustees assumed they would recover relatively quickly. The remaining 16 scars were determined to be more significant and would not recovery quickly without intervention (ERMA 2015; Meehan 2015).[6] Thirteen of these scars were located within Gulf Islands National Seashore in Florida (Meehan 2015).

4.6.8.2.3 Freshwater injury in Jean Lafitte National Historical Park and Preserve

In response to the *Deepwater Horizon* oil spill, Mississippi River water flows through the Davis Pond structure to Lake Cataouatche were increased during the summer of 2010 to reduce the potential for oil intrusion into inland marshes, including the Jean Lafitte National Historical Park and Preserve. Field surveys were performed within the Park's boundaries in fall 2010 and spring and fall of 2011 and 2012 (Figure 4.6-64) to assess impacts of these increased river water flows. In addition, field surveys were simultaneously conducted in a reference area (Bayou des Allemands) outside the National Park Service boundaries. Up to 39 stations were sampled in the Barataria estuary in Jean Lafitte National Historical Park and Preserve, and 5 stations were sampled in Bayou des Allemands. Data were collected on water quality parameters, sediment and water nutrient levels, SAV community structure, and floating aquatic species abundance. Details of this study are provided in Weston Solutions Inc. (2015).

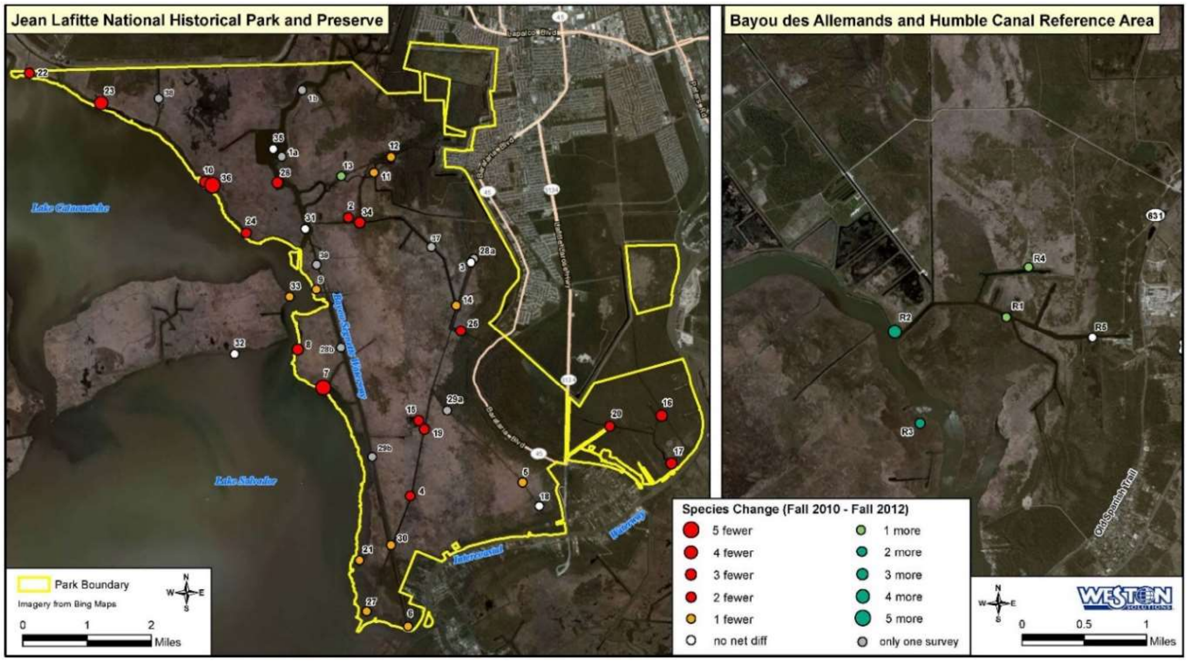
[6] Maps available at: *http://gomex.erma.noaa.gov/erma.html#/x=-86.22752&y=30.13024&z=8&layers=1+15561+5402*

The results of these assessment studies indicate that, between May and August 2010, the sustained increased flows from the Davis Pond structure resulted in reduced salinity into Lake Cataouatche and Jean Lafitte National Historical Park and Preserve (Weston Solutions Inc. 2015). These studies also showed an increase in freshwater flows and turbidity along the Lake Cataouatche shoreline. Focusing on the sampling stations along this shoreline, changes in habitat conditions coincided with changes in SAV community structure within the Park, including reductions in SAV diversity (Weston Solutions Inc. 2015). From fall 2010 to fall 2012, SAV diversity on the lake shoreline decreased from an average 4.6 (± 0.55) species per station to 1.3 (± 0.86) species per station (Figure 4.6-65; Weston Solutions Inc. (2015)). After the river water releases, SAV percent cover also dramatically decreased along the Jean Lafitte National Historical Park and Preserve Lake Cataouatche shoreline from an average 10.34 (± 2.92) percent cover per station to an average 1.76 (± 2.56) percent cover per station (Weston Solutions Inc. 2015). Conservatively, 60 acres (24 hectares) of SAV along the shoreline experienced 83 percent decline in percent cover from baseline, which was calculated using 2006 survey data (Poirrier et al. 2010) and aerial imagery from 2008 (Figure 4.6-66). Earlier research indicated that SAV beds remained stable or increased after normal flow from Davis Pond structure became more regular beginning in 2002 (Poirrier et al. 2010); however, SAV beds were apparently unable to withstand the increased flow rate and turbidity associated with the 2010 releases. Concurrent with the losses of cover and species diversity along Lake Cataouatche, both percent cover and species diversity increased at the reference sites in Bayou des Allemands from fall 2010 to fall 2012 (Weston Solutions Inc. 2015). Results for the reference sites suggest that, absent the river water releases, conditions in the area were otherwise favorable for SAV growth during the assessment period.



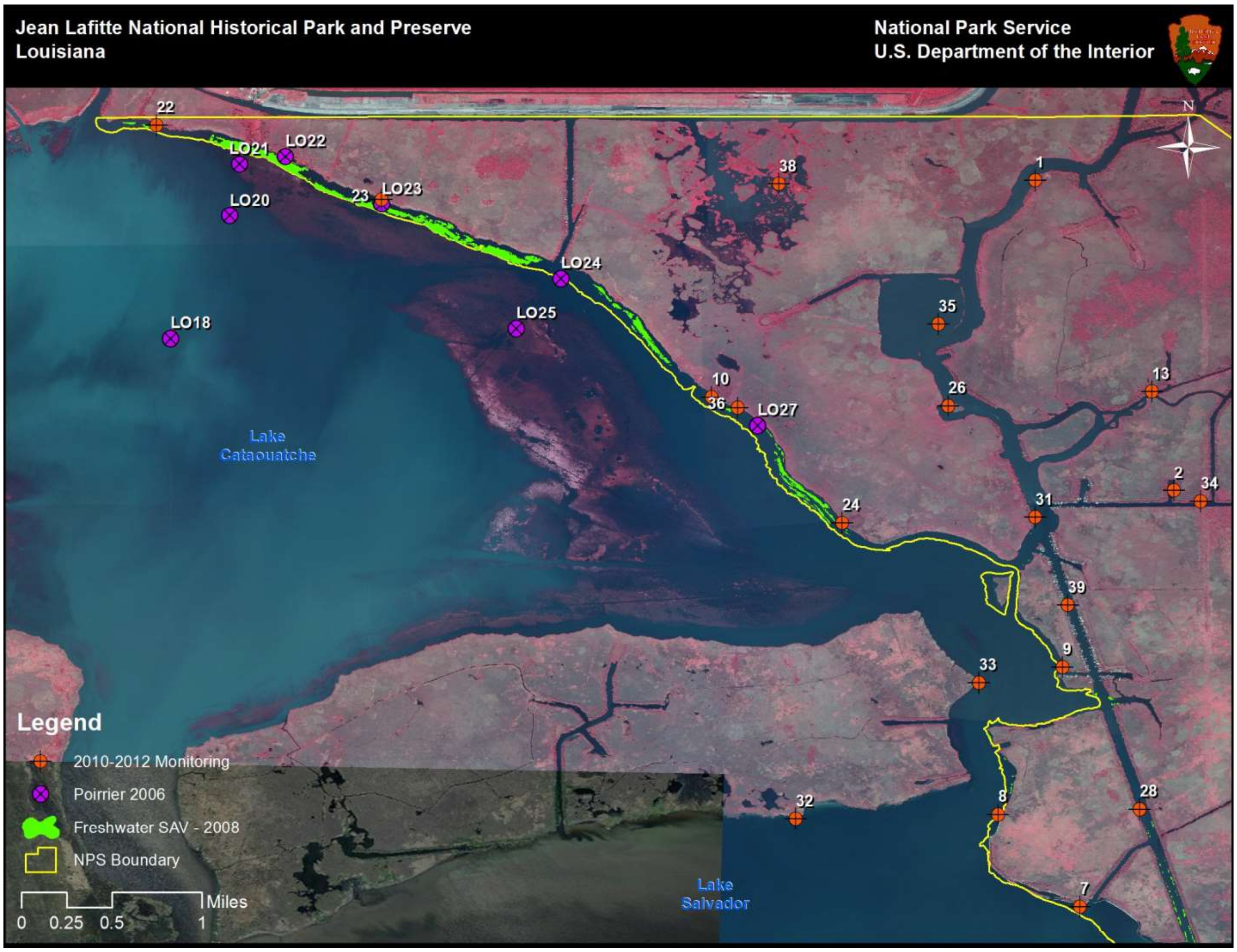
***Source**: Weston Solutions Inc. (2015).*

Figure 4.6-64. Jean Lafitte National Historical Park and Preserve study area and stations.



__Source__: Weston Solutions Inc. (2015).

Figure 4.6-65. Changes in SAV species diversity between fall 2010 and fall 2012. On average, SAV diversity decreased by 1.3 species per station.



Source: DOI.

Figure 4.6-66. Infrared image of baseline distribution of freshwater SAV (indicated in green) along the Lake Cataouatche shoreline within the Jean Lafitte National Historical Park Barataria Preserve.

4.6.8.3 Injury Quantification

4.6.8.3.1 Oil-Related Injury in Chandeleur Islands

The Trustees quantified injury to the seagrasses of Chandeleur Islands by acres lost. A total of 112 acres (45 hectares) of seagrass beds were identified as persistent loss (i.e., loss for two consecutive mapping intervals), and 159 acres (65 hectares) were classified as delayed loss (i.e., areas where seagrass was present in 2010 and 2011 but lost in 2012). These two results add up to 271 acres (110 hectares) of seagrass lost in the Chandeleur Islands (Cosentino-Manning et al. 2015). To determine the length of time it would take for this seagrass to reestablish without intervention or restoration, the Trustees applied information from past natural resource damage assessments of vessel grounding sites where seagrass cover was destroyed. This science-based approach recognizes that ecological services provided by seagrasses are lost or impaired during the recovery period; the approach also recognizes that the time needed to reach full recovery from an injury is contingent upon the type and size of the injury, the species composition, and the prevailing environmental conditions. Trustee recovery calculations were limited to areas of persistent loss exceeding 100 square meters (0.0247 acres). Areas of persistent loss <

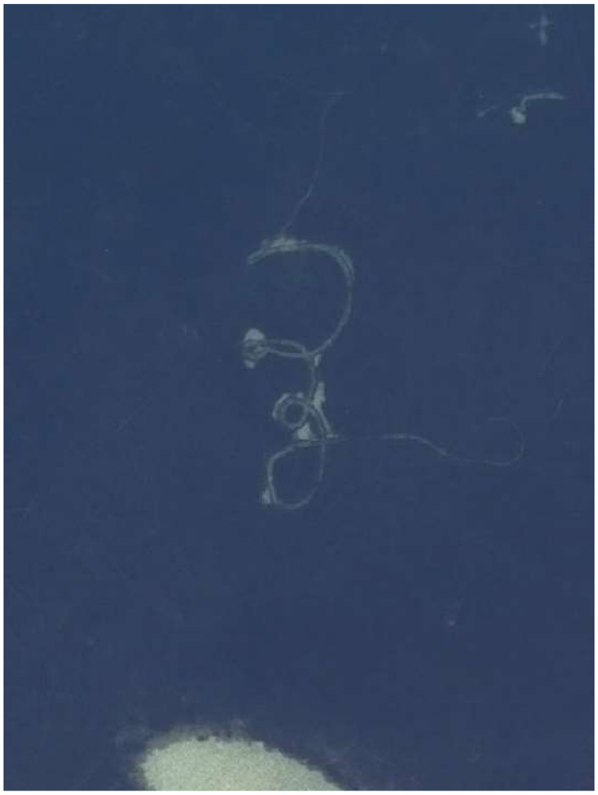
100 square meters were assumed to recover in 1 year. Overall, 51 acres of seagrass had persistent loss greater than 0.0247 acres (100 square meters); of these, 34 acres (14 hectares) were identified as having recovery times predicted to exceed 1 year. Predicted recovery times for these 34 acres (14 hectares) varied as follows:

- Approximately one-third of these persistent loss areas (11 acres or 4.5 hectares) have a predicted recovery time of between 1 and 2 years.
- 37 percent of the persistent loss areas (13 acres or 5.3 hectares) have a predicted recovery time between 2 and 10 years.
- The remaining 10 acres (4 hectares) with a patch size between 1 and 2 acres (between 0.4 and 0.8 hectares) have predicted recovery times ranging from 14 to 26 years (Cosentino-Manning et al. 2015).

Recovery times of equivalent areas of delayed loss are assumed to be comparable to recovery of persistent loss areas.

4.6.8.3.2 Physical Response Injury

The assessment effort described in Section 4.6.8.2.1 (Injury Determination) documented 876 square meters of scars and blowholes in Florida seagrass beds from 16 scars. In this process, 13 response vessel scars totaling 502 square meters were identified within the boundaries of Gulf Islands National Seashore, Florida District, in the vicinity of Pensacola, Florida (Meehan 2015). Representative seagrass scars are shown in Figure 4.6-67.



Source: NOAA.

Figure 4.6-67. Seagrass scars typical of the types caused by vessels in Gulf Islands National Seashore waters.

4.6.8.3.3 Freshwater Injury in Jean Lafitte National Historical Park and Preserve

A total of 50 acres (20 hectares) of SAV was lost along the Lake Cataouatche shoreline in Jean Lafitte National Historical Park and Preserve between March 2010 and November 2012, as described in Section 4.6.8.2.2 (Physical Response Injury). This loss was estimated based on an aerial imagery analysis conducted by the National Park Service in combination with fall and spring field measurements of species diversity and percent cover at established sampling stations from fall 2010 through fall 2012. During the assessment period, little indication of natural recovery was seen at the Lake Cataouatche stations; Poirrier et al. (2009) indicate that natural recovery for freshwater SAV can take 6 years or more under the best conditions; it can possibly take longer where currents or wave energy are unattenuated, limiting the ability of seedlings to become established (EPA 2000).

4.6.8.4 Conclusions and Key Aspects of the Injury for Restoration Planning

The seagrass beds off the Chandeleur Islands are unique and extremely productive. They are the only existing marine seagrass beds in Louisiana, and are the largest, most continuous seagrass beds in the northern Gulf of Mexico (Cosentino-Manning et al. 2015). They are part of the Breton National Wildlife Refuge—the second-oldest refuge in the National Wildlife Refuge System. These islands are prolific environments where hundreds of species of finfish, crustaceans, and wildlife flourish (Cosentino-Manning et al. 2015). The heavily vegetated interiors of this fragmented chain are veritable sanctuaries, where juvenile fish, crabs, and shrimp can find refuge, nursery, and feeding grounds, increasing their

odds of survival in the Gulf (Cosentino-Manning et al. 2015). The islands' location serves as a "fly trap" in that they are the first area of vegetated shallow water habitat that pelagic juvenile fish and invertebrates encounter in the vast Gulf. There, animals are able to escape predation and feed in productive shallows.

The seagrasses off these islands also provide habitat and food for green sea turtles and support the overwintering of waterfowl. In addition, for generations, recreational anglers have enjoyed world-class fishing associated with seagrass productivity in the Chandeleur Islands (Cosentino-Manning et al. 2015).

While the Chandeleur Islands are physically and biologically isolated from the mainland, they are ecologically connected to a much larger oceanic region: the wider Gulf of Mexico, the tropical western Atlantic, and the Caribbean Basin. They are the only seagrass beds in the United States to have many of the species found in these other locations. The islands and surrounding waters are considered pristine as demonstrated by baseline sampling results, and they are isolated from chemical and nutrient contamination, unlike many of the other shallow coastal areas within the Gulf that are adjacent to human populations and urban runoff.

The Chandeleur Islands, act as a defense "barrier" that absorbs the initial impacts of wind and wave fetch and tropical weather systems. The seagrasses have helped create that protective barrier and stability of the Islands for hundreds of years. The existence of seagrass beds in the Chandeleur Islands is made possible by two critical factors: (1) the presence and persistence of emergent land features (the islands) above sea level that baffle wave and current energy, and (2) a source of sediment to maintain suitable water depth (≤ 2 meters) on the leeward platform where the seagrasses occur (Cosentino-Manning et al. 2015). The emergent islands and the platform are a coupled geological unit (barrier island system) slowly migrating west into Chandeleur Sound. The leeward platform is the foundation upon which the islands are perched and maintained above sea level. The seagrasses play an important role in this process functioning as a stabilizing feature on the submerged platform and helping to maintain its elevation as well as the Islands (Cosentino-Manning et al. 2015).

Seagrass beds in Florida coastal waters are an important resource for many recreationally and commercially important aquatic species and many endangered species, including sea turtles and manatees. Response actions to the *Deepwater Horizon* oil spill caused two types of injuries: propeller scars from response vessels and blow holes from vessels attempting to power off a shallow seagrass bed (Meehan 2015). The area and depth of prop scars and blow holes vary, and restoration options depend on the length and width of the damage. Natural recovery relies on natural re-colonization of seagrass species and natural sediment filling (Uhrin et al. 2011).

Freshwater SAV beds such as those found in Jean Lafitte National Historical Park and Preserve provide numerous ecological functions. These include providing food and cover for fish and wildlife, decreasing wave energy, increasing sedimentation, and stabilizing sediments (Poirrier et al. 2010). However, freshwater SAV has requirements for growth and survival. These requirements include correct ranges for salinity, light, total suspended solids, plankton chlorophyll *a*, dissolved inorganic nitrogen, dissolved inorganic phosphorus, water movement, wave tolerance, sediment grain size, and organic matter. Slow current velocities are needed for the development of freshwater SAV seedlings. High wave energy can affect SAV in multiple ways: it can cause erosion, which increases total suspended solids in the water

column (reducing light availability); it can change grain size in the sediment, which can reduce the success of SAV becoming anchored and established (EPA 2000); and it can also uproot plants (Poirrier et al. 2010).

The Trustees considered the totality of the SAV injury in planning restoration. The key aspects of the SAV injury that informed the Trustees' comprehensive restoration planning include:

- SAV in the Chandeleur Islands, Louisiana, were injured as a result of oiling. The spatial distribution of seagrasses decreased from 2010 to 2012 along the shallow shelf west of the Chandeleur Islands. A total of 112 acres (45 hectares) of seagrass beds were identified as persistently loss (defined as loss for two consecutive mapping intervals), and 160 acres (65 hectares) were classified as delayed loss (areas where seagrass was present in 2010 and 2011 but lost in 2012).

- SAV was injured across the northern Gulf of Mexico due to the physical effects of vessels used during response activities. The effects including propeller scars and blow holes from vessels attempting to power off a shallow seagrass bed. The assessment effort documented 876 square meters of scars and blowholes in Florida seagrass beds; and 502 square meters were identified within the boundaries of Gulf Islands National Seashore, Florida District.

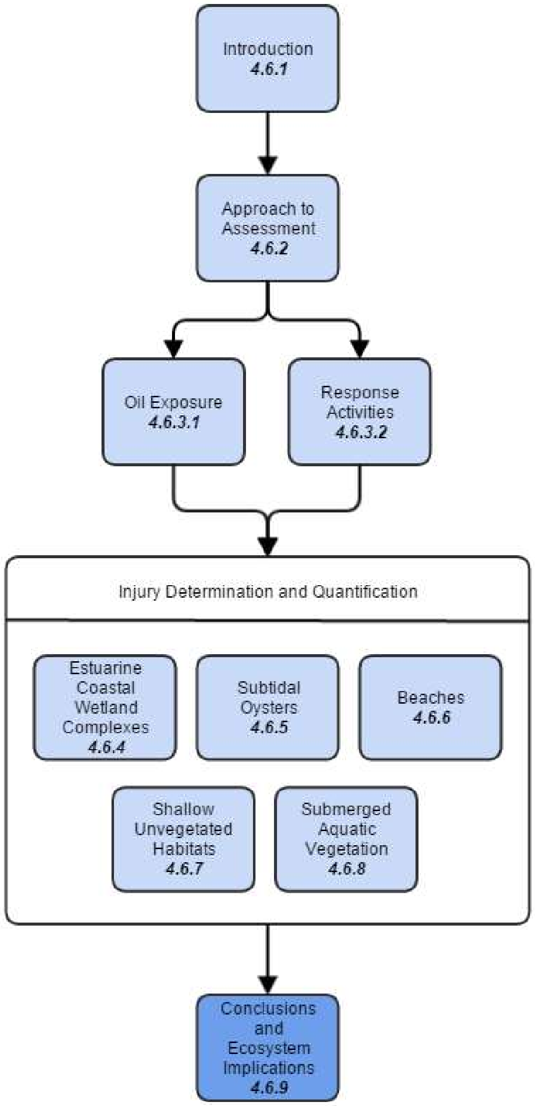
- SAV in the federally managed Jean Lafitte National Historic Park and Preserve, Louisiana, was injured as a result of freshwater releases. Increased amounts of freshwater from the Davis Pond Diversion release reduced salinity, resulting in reductions in SAV species diversity and percent cover. Along the Lake Cataouatche Shoreline in the Park, Trustees documented an 83 percent loss of SAV cover between March 2010 and Nov 2012.

As described in Chapter 5 (5.5.2 and 5.5.3), the Trustees have identified restoration approaches for these injuries, including restoration on federally managed lands. Emergency restoration activities (see Chapter 1, Introduction) also addressed SAV injuries in Florida seagrass beds.

4.6.9 Conclusions and Key Aspects of the Injury for Restoration Planning

Key Points

- Injuries were detected over a range of species, communities, and habitats. The injuries affected a wide variety of ecosystem components over many hundreds of kilometers of the northern Gulf of Mexico coastline.
- Injuries to nearshore resources have cascading impacts throughout the ecosystem. The injuries to nearshore and shoreline resources influence the overall health and productivity of the Gulf of Mexico ecosystem.
- Restoration will utilize a comprehensive, integrated portfolio approach that includes representative resource groupings and supporting habitats, such as coastal wetlands that provide benefits to various species and ecological services.



The *Deepwater Horizon* spill and associated response actions caused a suite of injuries to nearshore marine resources and the services they provide. These injuries occurred at the species, community, and habitat level and affected a wide variety of ecosystem components over an area extending along many hundreds of kilometers of the northern Gulf of Mexico coastline.

The Gulf Coast, from Texas to Florida, contains some of the world's most biologically diverse habitats, including coastal marshes, estuaries, sand beaches, dunes, and barrier islands. These habitats are critical to the survival of wildlife populations and are home to many federally protected threatened and endangered species. As testament to the ecological and public value these habitats represent, many kilometers of this shoreline have been set aside by local, state, and federal agencies to preserve and protect these habitats and the wildlife that depend on them. Table 4.6-23 summarizes the miles and acres of federal lands that were adversely affected by the *Deepwater Horizon* spill.

Table 4.6-23. Federal lands impacted by oil and response activities.

Habitat	Texas		Louisiana		Mississippi		Alabama		Florida	
	km	ha	km	ha	km	ha	km	ha	km	has
Sand Beaches	13	80	26	151	92	544	19	99	129	729
Marsh	-	-	23	-	10	-	2	-	-	-
Submerged Aquatic Vegetation (SAV)	-	-	-	20	-	-	-	-	-	.05

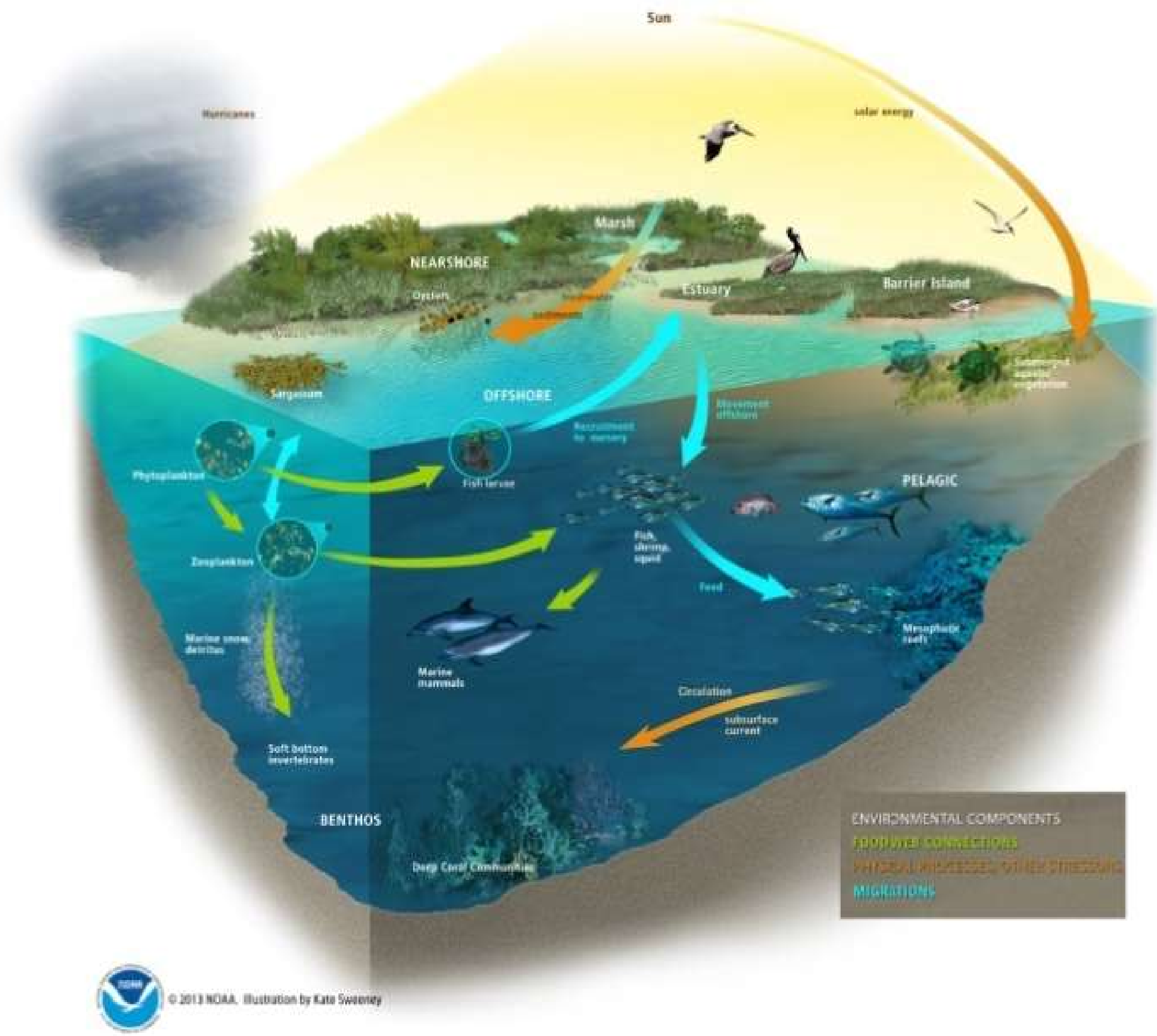
- Habitat was not present or was not measured by the specified metric.

Although the injuries described in this Section occurred in nearshore and shoreline habitats, these habitats and biological resources are interconnected through ecological and physical relationships such as foodweb dynamics, organism movements, nutrient and sediment transport and cycling, and other fundamental ecosystem processes (Figure 4.6-68). Due to these interactions, injuries to nearshore resources can have cascading impacts throughout the ecosystem, and the injuries to nearshore and shoreline resources influence the overall health and productivity of the Gulf of Mexico ecosystem. Further, because the approach to assessing nearshore impacts focused on injury to accessible habitats and species over a limited area and time period, the total injury to the nearshore ecosystem is almost certain to be larger than the sum of the studied components.

The key over-arching elements of the injury assessment findings include:

- Injuries were extensive and pervasive, affecting several hundred kilometers of interconnected coastal habitats. Affected habitats include salt marsh, mangrove, SAV, unvegetated areas, and sand beaches and dunes. The animals that live in these habitats were also injured. These animals include crabs, snails, insects, shrimp, resident fish, oysters, and federally listed threatened species (e.g., Gulf Sturgeon and beach mice).
- The ecological linkages of these habitats and communities and their connectivity to the larger Gulf of Mexico ecosystem can result in cascading impacts, influencing the overall health and productivity of the Gulf of Mexico ecosystem

The Trustees considered all aspects of the injury in restoration planning. The broad nature and extent of injuries to nearshore resources, species, and habitats, in particular, served as an important basis for the Trustees' restoration planning. The Trustees also considered the ecosystem effects that are described below in their restoration planning, and the restoration plan therefore was informed by reasonable scientific inferences based on the information collected relative to specific injuries.



Source: *Kate Sweeney for NOAA.*

Figure 4.6-68. The injured nearshore and shoreline habitats of the northern Gulf of Mexico are connected with the overall health and productivity of the Gulf through fundamental ecosystem relationships and processes.

4.6.9.1 Ecosystem Effects

Given the overall scale of the incident and the lack of practical feasibility to study every species and location exposed to the oil in the nearshore marine environment, the Trustees employed an ecosystem approach to the assessment. They evaluated injuries to a suite of representative habitats and faunal species. The implications of the measured resource injuries with regard to broader coastal ecosystem impacts include:

- **Coastal marsh and mangrove vegetation**. Injury to nearshore wetland vegetation was observed over hundreds of kilometers of coastline in the northern Gulf of Mexico, with more severe and broader injuries documented along more heavily oiled shorelines. In particular, herbaceous salt marsh vegetation exposed to trace or greater vertical oiling of plant stems displayed reductions in live plant cover and above ground biomass, particularly in the marsh edge zone closest to the shoreline.

  The implications of these measured injuries to vegetation extend far beyond the loss of the vegetation itself, as marsh vegetation contributes to the overall health of the Gulf of Mexico. Marsh plants produce biomass through photosynthesis and release nutrients through decomposition, thereby forming the basis of terrestrial and aquatic food webs (Figure 4.6-1). Marsh habitat provides invaluable spawning, nursery, and feeding grounds for the many commercial fish and shellfish species that depend upon the physical protection of the estuary to complete their lifecycles. In particular, the marsh edge, where the most acute injuries occurred as a result of the spill, serves as a critical transition between the emergent marsh vegetation and open water: it serves as the gateway for the movement of organisms and nutrients between intertidal and subtidal estuarine environments (Levin et al. 2001). Injuries to marsh vegetation therefore initiate a cascade of trophic-level impacts to bacteria, invertebrates, plankton, and higher-level organisms. Some of these impacts were not directly measured by the assessment, but can be inferred.

  Marsh plants also play an important role in shoreline stabilization, holding and stabilizing soil and sediment, and helping to retain and accumulate soil in the marsh (Figure 4.6-39). The marsh serves a role in coastal flood protection by attenuating storm and wave energy. Marsh habitat helps to protect water quality by capturing suspended sediment and removing excess nutrients and pollutants from upland environments (Bricker et al. 1999; Fisher & Acreman 2004). A loss of marsh vegetation therefore has adverse implications for all of these marsh functions and processes.

- **Marsh Fauna**. The studies conducted by the Trustees showed injury to all marsh fauna species that were studied. Examples included: a reduction in periwinkle abundance and recruitment; reductions in growth (associated with reduced survival) of shrimp, juvenile flounder, and red drum; reduced amphipod survival; reduced reproductive success of *Fundulus* spp.; reduced fiddler crab abundance (as measured by burrow density); and decreased cover of nearshore oysters. In addition, non-NRDA studies conducted by university researchers demonstrated that small organisms that live in marsh sediments known as meiofauna were injured in heavily oiled areas. Meiofaunal community composition and the density of meiofauna (e.g., copepods and

worms) were also adversely affected (Brunner et al. 2013). Injuries to marsh birds are discussed in Section 4.7.

The significance of these injuries extends far beyond the impact to the individual species studied. Rather, the injuries are indicative of adverse effects to the broader ecosystem. For instance, meiofauna provide ecological functions as herbivores, detritivores, and scavengers, and further support the aquatic food web. Shrimp and fish are important prey organisms to higher trophic levels and also play an important role in exporting nutrients from nearshore habitats to offshore areas. Some additional specific examples of these broader ecosystem implications include:

- *Fundulus spp.* plays a key role as a connector of energy between the marsh and the open Gulf waters. Found predominantly in shallow nearshore waters, they are among the largest of the Gulf forage fish. Additionally, they are preyed upon by wildlife, birds, and many sport fish, including flounder, speckled trout, and red snapper (Ross 2001). Therefore, a loss of this species can have negative implications for energy transfer dynamics between the nearshore and open water systems.

Fiddler crabs are important prey items and play a functional role in modifying marsh vegetation, sediments, organic material, nutrient cycling, microbial communities, and meiofauna. Therefore, a reduction in fiddler crab abundance (as indicated by burrow density) would have adverse implications for all of these physical processes and dependent communities. Through complex foodweb interactions, these nearshore species are also inextricably linked to higher trophic levels in the Gulf of Mexico, including top-level predators such as birds (Section 4.7) and dolphins (Section 4.9). These relationships are conceptually shown in Figure 4.6-2. Accelerated erosion of marsh edge habitat will also have cascading effects for the diverse species that rely on this habitat.

- **Subtidal Oysters**. An estimated 2.8 to 5.1 billion subtidal oysters (adult equivalent) were killed over an area of 479 square kilometers of oyster habitat in Louisiana. When combined with losses to nearshore oysters over hundreds of kilometers of oiled shoreline, the reductions in the spawning stock of oysters in the northern Gulf of Mexico will affect reproduction and recruitment over multiple generations. Trustees estimate total losses of oysters from death and reproductive impairment over 7 years to be 4 to 8.3 billion adult equivalents. Oyster reefs and beds serve as feeding and foraging habitat for other aquatic organisms such as shellfish, crabs, and finfish. Oysters also contribute to water quality and clarity through their filtering action. Therefore, a loss of oysters will have cascading adverse effects to all of these supported organisms and functions.

- **Sand beach habitat**. Sand beaches across the northern Gulf of Mexico were widely oiled as a result of the spill. Response activities disturbed habitats extensively and repeatedly at sand beaches and dunes across the northern Gulf, causing additional injuries. These beaches and dunes are ecologically and recreationally important shoreline habitats that provide breeding, nesting, wintering, and foraging for nearshore biota. Furthermore, they are inextricably intertwined with other coastal habitats. For example, beach mice live their entire lives scurrying

about the beach and dunes. These mice are dependent upon seeds of specialized dune vegetation for food, leaves and stems for shelter from predators, and roots to stabilize the walls of their underground burrows. Additionally, many Gulf bird species rely on sand beaches, dunes, and marshes for their existence. The birds nest on the beaches and dunes, and they feed on crustaceans and fish in the nearby marshes (Caffey et al. 2000). It is the combined presence and connectivity of these habitat types in close proximity that makes the shoreline so ideal. Consequently, impacts to sand beaches and dunes can have effects beyond the injury to the habitat itself.

- **Shallow unvegetated habitat**. The continued existence of the threatened Gulf sturgeon depends on maintaining and protecting important riverine and marine habitats. Large numbers of fish from most Gulf sturgeon river populations were likely exposed to *Deepwater Horizon* oil, and were likely injured.
- **Submerged aquatic vegetation.** SAV was adversely affected by oiling and by response activities, including river water releases and response vessel propellers. SAV habitats provide food and shelter for birds, fish, shellfish, invertebrates, and other aquatic species, and are highly productive. The Chandeleur Islands SAV, for example, is a critical link in the lifecycle of many species of fish, turtles, and birds. The islands' location in effect serves as a "fly trap," as they are the first area of vegetated shallow water habitat that pelagic juvenile fish and invertebrates encounter in the vast Gulf. There, animals are able to escape predation and feed in productive shallows before moving on to their adult habitats. Therefore, loss of this habitat has much broader implications for many Gulf species that rely upon them for food and shelter.

In summary, injuries to nearshore marine habitats and resources occurred across all trophic levels and biological scales of organization. Coastal resource injuries were documented across all trophic levels, from primary producers (plants) to top level predators (e.g., fish, birds, marine mammals); and these injuries affected a variety of ecological functions that link this coastal environment with the broader northern Gulf of Mexico ecosystem (Figure 4.6-2).

4.6.9.2 Restoration Considerations

As described in Chapter 5 (Section 5.5.2), the Trustees have identified a comprehensive, integrated portfolio approach to restoration. This restoration portfolio includes species groupings, such as oysters and fish, as well as supporting habitats, such as coastal wetlands, that provide a diversity of ecological services, and benefits to a large variety of species.

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