4. Injury to Natural Resources

4.1 Approach to the Injury Assessment

What Is in This Section?

- **Introduction (Section 4.1.1):** When in the NRDA process was the injury assessment conducted, how is the injury assessment presented in Chapter 4, and how do the assessment results relate to restoration planning?
- **Regulatory Framework for the Trustees' Injury Assessment (Section 4.1.2):** What is the regulatory basis for and how did that basis frame the Trustees' injury assessment?
- **The Trustees' Ecosystem Approach to Injury Assessment (Section 4.1.3):** Why did the Trustees choose an ecosystem approach to the injury assessment and which scales of biological organization did the Trustees study?
- **Injury Assessment Timeline and Stages (Section 4.1.4):** What were the stages of the Trustees' injury assessment process?
- **Injury Assessment Methods (Section 4.1.5):** What methods did the Trustees use to conduct the injury assessment and why?
- **Trustees' Data Management Process and Systems (Section 4.1.6):** What data management systems and processes did the Trustees use?
- **Road Map to the Trustees' Injury Assessment (Section 4.1.7):** What are the major sections of the injury assessment presented in this chapter?
- **References (Section 4.1.8)**

Executive Summary

The Trustees conducted the injury assessment presented in this chapter under the authority of and in accordance with Oil Pollution Act (OPA) regulations. The injury assessment establishes the nature, degree, and extent of injuries from the *Deepwater Horizon* incident to both natural resources and the services they provide. Injury assessment results are used to inform restoration planning so that restoration can address the nature, degree, and extent of the injuries.

Under OPA, injury assessment involves determining whether resources or their services were injured, and then quantifying the degree and extent of those injuries and service losses:

- **Injury determination.** To determine injury, the Trustees evaluated whether a pathway could be established from the discharge to the exposed resource, whether the resource had been exposed to oil, and the injury caused by that exposure. The Trustees evaluated not only the extent of injuries to natural resources, but also to the services those resources provide. They also evaluated injuries resulting from response activities, including fishery and beach closures, beach excavation, removal of oil from marshes and beaches, boat activities, and placement of protective boom (see Chapter 2, Incident Overview, for further discussion of response activities).

- **Injury quantification**. To quantify the degree and extent of the injuries, the Trustees compared the injured resources or services to baseline conditions—i.e., the condition of the natural resources or services that would have existed had the incident not occurred. The Trustees did not quantify all injuries they determined. Rather, they focused injury quantification where it could best aid restoration planning. The Trustees' approaches to quantifying injuries varied by resource type and scientific study. Generally, the Trustees did not quantify effects in terms of population size or status (due to infeasibility), and they placed limited reliance on collected or observed counts of animals killed by the incident (because so many animals killed were not observable).

Based on the vast scale of the incident and potentially affected resources, the Trustees employed an ecosystem approach to the assessment. This involved evaluating injuries to a suite of representative habitats, communities, and species, rather than to all potentially affected individual species and habitats. The Trustees also evaluated injuries to representative ecological processes and linkages.[1] The Trustees conducted their assessment at multiple scales of biological organization, including the cellular, individual, species, community, and habitat levels.

The Trustees' injury assessment started as soon as news of the spill was received and continued with a multi-phased iterative approach, in which planning and design decisions were informed by the data collected and evaluated. The Trustees used a variety of assessment procedures, including field and laboratory studies, and model- and literature-based approaches. They used scientific inference to make informed conclusions about injuries not directly studied.

Field data collection by the Trustees involved roughly 20,000 trips, which generated over 100,000 samples of water, tissue, oil, and sediment and over 1 million field data forms and related electronic files. Testing of samples generated millions of additional records. The Trustees developed rigorous protocols and systems to manage sample collection, handling, and data storage. To store data, the Trustees developed a "data warehouse," referred to as the Data Integration, Visualization, and Reporting system (DIVER), which is publicly accessible at *https://dwhdiver.orr.noaa.gov/*.

Sections 4.2 to 4.10 of this chapter present the Trustees' injury determination and quantification methods, results, and findings for seven resource categories: water column, benthic resources, nearshore marine ecosystems, birds, sea turtles, marine mammals, and lost recreational use. The length and detail level in these sections vary because of the differing scopes of the assessment efforts and the different number of individual resources within each resource category. Section 4.11 summarizes the Trustees' injury assessment key findings and conclusions, which were based on the results of the resource-specific assessments, as well as the Trustees' understanding of the inherent connectivity across these habitats and biota within the northern Gulf of Mexico ecosystem.

[1] "Ecological linkages" refers to the interactions between organisms—from microbes to plants to animals—and their chemical, biological, and physical environment. See Chapter 3, Ecosystem Setting, for further information.

The Trustees used the results of the injury assessment to develop their restoration plan, described in Chapter 5. Though some studies are ongoing, the Trustees do not expect that further study will change selection of the preferred alternative presented in this Final PDARP/PEIS.

4.1.1 Introduction

As described in Chapter 1, the Trustees performed the Natural Resources Damage Assessment (NRDA) in accordance with OPA regulations (33 USC 2701 *et seq.*) at 15 CFR Part 990. Per OPA regulations, the *Deepwater Horizon* NRDA involves three main phases (15 CFR §§ 990.12, 990.40–990.66):

- **Preassessment,** in which the Trustees evaluate the potential for injuries to natural resources resulting from the *Deepwater Horizon* incident.
- **Restoration planning,** in which the Trustees evaluate and quantify injuries to natural resources to determine the need for, type of, and extent of restoration.
- **Restoration implementation,** in which the Trustees ensure that restoration is implemented.

The Trustees' injury assessment, presented in this chapter, was conducted across both the preassessment and restoration planning phases. Chapter 4 presents the approach to and results of this injury assessment, as follows (see Section 4.1.7 for a more detailed road map):

- **Section 4.1 (Approach to the Injury Assessment)** provides an overview of the regulatory framework for and approaches the Trustees used to determine and quantify injuries to natural resources.
- **Section 4.2 (Natural Resources Exposure)** describes the nature and extent to which natural resources were exposed to contamination from the *Deepwater Horizon* incident.
- **Section 4.3 (Toxicity)** describes the approach to and results of characterizing the toxic effects of *Deepwater Horizon* oil.
- **Sections 4.4 through 4.10** describe the methods, results, and conclusions specific to each of the resources assessed: water column (Section 4.4), benthic resources (Section 4.5), the nearshore marine ecosystem (Section 4.6), birds (Section 4.7), sea turtles (Section 4.8), marine mammals (Section 4.9), and lost recreational use (Section 4.10).
- **Section 4.11 (Summary of Injury Effects and Quantification)** presents the Trustees' key findings and conclusions resulting from the injury assessment.

The Trustees' injury assessment establishes the nature, degree, and extent of injuries from the *Deepwater Horizon* incident to both natural resources and the services they provide. As described in Chapter 5 (Restoring Natural Resources), the Trustees have used the assessment results presented in Chapter 4 to formulate restoration approaches targeted to restoring the full range of resources and ecosystem services injured from this incident.

4.1.2 Regulatory Framework for the Trustees' Injury Assessment

Injury is defined in the OPA regulations as:

> an observable or measurable adverse change in a natural resource or impairment of a natural resource service. Injury may occur directly or indirectly to a natural resource and/or service. Injury incorporates the terms "destruction," "loss," and "loss of use" as provided in OPA (15 CFR § 990.30).

OPA regulations identify several potential **types of injuries**, including (but not limited to):

> adverse changes in: survival, growth, and reproduction; health, physiology and biological condition; behavior; community composition; ecological processes and functions; physical and chemical habitat quality or structure; and public services [15 CFR § 990.51(c)].

As described in Sections 4.2 to 4.11, *all of the above types of injury occurred as a result of the Deepwater Horizon incident*.

Under OPA, injury assessment involves two elements, described below:

- **Injury determination,** in which the Trustees evaluate whether the *Deepwater Horizon* incident injured natural resources or impaired their services (15 CFR § 990.51).
- **Injury quantification,** in which the Trustees quantify the degree and the spatial and temporal extent of those injuries and service losses (15 CFR § 990.52).

4.1.2.1 Injury Determination

4.1.2.1.1 The Components of an Injury Determination

Figure 4.1-1 illustrates the three components of an injury determination.

- **Pathway evaluation.** The Trustees evaluated whether a pathway could be established from the discharge to the exposed natural resource [15 CFR § 990.51(b)]. As defined in the OPA regulations, pathways may include (but are not limited to) "the sequence of events by which the discharged oil was transported from the incident and either came into direct physical contact with a natural resource, or caused an indirect injury" [15 CFR § 990.51(d)].
- **Exposure assessment.** As part of their injury determination, the Trustees evaluated whether the injured natural resource had been exposed to the discharged oil (directly or indirectly) [15 CFR § 990.51(b), (d)]. Section 4.2 summarizes the widespread exposure of natural resources in the northern Gulf of Mexico to oil and provides an overview of the pathways by which discharged oil was transported in the environment, resulting in those exposures.
- **Injury evaluation.** The Trustees evaluated injuries caused by exposure to oil, as well as injuries that resulted from actions taken to respond to the *Deepwater Horizon* incident. For these "response" injuries, the Trustees must determine whether the injury or impairment of a natural resource service occurred as a result of the incident [15 CFR § 990.51(e)]. Section 4.3 describes

the toxic effects of the oil that the northern Gulf of Mexico resources were exposed to, and Sections 4.4 to 4.10 describe resource-specific pathways, exposure, and injury.

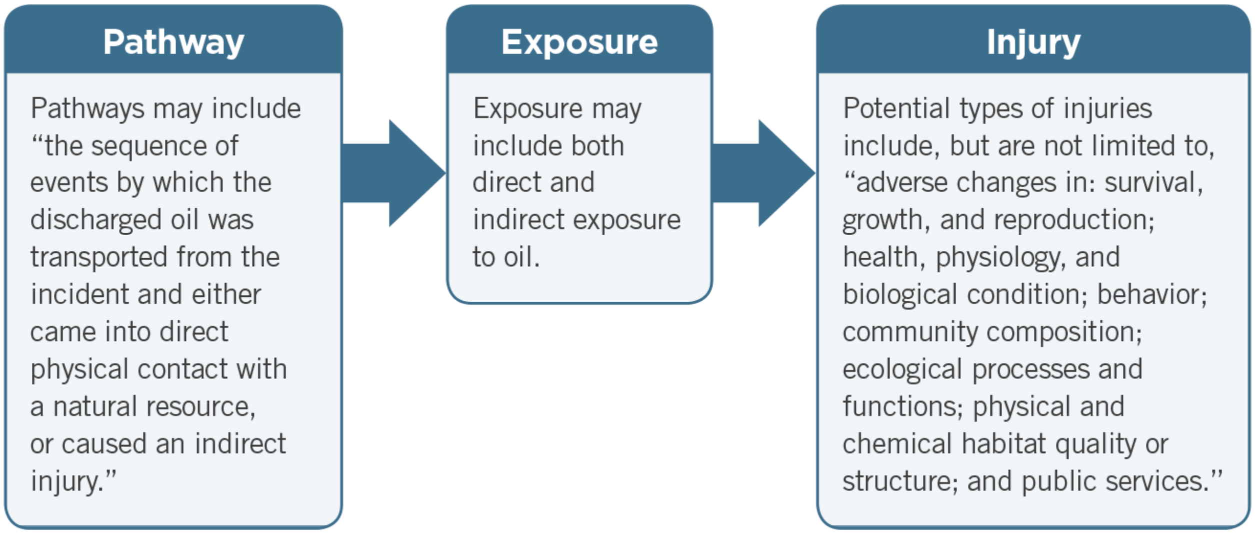


Figure 4.1-1. Pathway evaluation, exposure assessment, and evaluation of whether injuries occurred are the fundamental elements of injury determination (15 CFR § 990.51), and OPA regulations (15 CFR § 990.51) provide specific definitions for "pathway" and "injury," as shown in this figure. Resource-specific methods of evaluating pathway, exposure, and injury are described in Sections 4.2 to 4.10 of this chapter.

OPA regulations indicate that the "Trustees must determine if injuries to natural resources and/or services have resulted from the incident" [15 CFR § 990.51(a)]. The Trustees used a variety of standard scientific approaches, appropriate to the nature of the resource and the injury in question, to make this determination.

In some instances, the relationship between the incident and injuries did not require explicit evaluation (e.g., impacts directly resulting from response actions, such as physical disturbance and removal of beach sands). In other cases, the Trustees determined that the incident caused injuries by **establishing pathway and exposure**. For example, Section 4.4 demonstrates a clear relationship between the exposure of marsh vegetation to *Deepwater Horizon* oil and a series of consequential injuries, including death of marsh plants.

In Sections 4.2 to 4.10, the Trustees present information and conclusions related to:

- **Exposure** of the resource to *Deepwater Horizon* oil.
- **Injury determination**, in which the adverse effects of that exposure are evaluated and described.
- **Injury quantification**, in which injuries are quantified, where feasible, in terms of their degree, and spatial and temporal extent.

For other resources, the Trustees determined injuries based on **field studies** that compared locations exposed to oil with unoiled reference areas. For example, the Trustees documented a series of injuries to bottlenose dolphins in oiled areas that were not observed in unoiled reference locations. For observational injury determinations, the Trustees considered several common evaluative factors, including:

- The presence of known or likely exposure pathways, either direct or indirect.
- The spatial pattern of observed injury and relationship to the oil exposure, either direct or indirect, including comparisons with unoiled (or less oiled) reference areas.
- The timing of observed adverse changes in relation to the timing of the oil exposure.
- Statistical relationships in the data.
- Whether the cause-effect relationship is consistent with known processes in ecology, biology, and/or toxicology.
- Evaluation of possible alternative causes, where appropriate.

4.1.2.2 Injury Quantification

Injury quantification involves quantifying the degree and spatial/temporal extent of the injury relative to **baseline conditions**—i.e., the condition of the natural resource or services that would have existed had the incident not occurred (15 CFR § 990.30). As described in Sections 4.4 to 4.10, the Trustees use various techniques to evaluate baseline conditions: comparison to historical data, evaluation of time trends, comparison with reference areas or conditions, calculation of incremental losses through counts of collected injured organisms (e.g., number of dead birds), and quantitative models based on oil exposure.

Following OPA regulations (15 CFR § 990.52), the Trustees quantified injuries in terms of the degree and spatial and temporal extent of the resource injury or the amount of services lost as a result of the incident. To estimate the temporal extent of injury, the Trustees considered several factors, including:

- Time trend data, where available.
- Life history data for injured organisms.
- Relevant scientific information from prior oil spills.
- Natural recovery times.
- Important physical/chemical processes.

The Trustees also used numerical models in calculating the amount of injury caused by oil exposure. For example, the Trustees' quantification of injuries to water column resources (such as fish and invertebrates) relied in part on the use of numerical models to calculate the total fish mortality based on their exposure to toxic concentrations of *Deepwater Horizon* oil. The Trustees quantified certain injuries by reference to other relevant indicators, rather than relying on resource-specific injury data.

4.1.3 The Trustees' Ecosystem Approach to Injury Assessment

4.1.3.1 The Basis for the Trustees' Ecosystem Approach

As discussed in other chapters of this document, the scale of the *Deepwater Horizon* spill was unprecedented in terms of geographic extent, duration, and the complex array of ecosystem zones and habitats affected:

- **Geographic extent.** As detailed in Section 4.2, 3.19 million barrels of oil were released into the Gulf of Mexico over nearly 3 months. This resulted in a surface slick that cumulatively covered over 43,300 square miles of the ocean surface, an area roughly equivalent to the size of Virginia, and oiled more than 1,300 miles of shoreline habitats (see Section 4.2).

- **Response effort.** The spill necessitated a similarly unprecedented extensive and diverse response effort, including application of nearly 2 million gallons of dispersants, burning of surface oils, application of more than 9 million feet (more than 1,700 miles) of absorbent boom in offshore and nearshore environments, releases of fresh water in Louisiana, and oil removal from shorelines at different levels of intensity and destructiveness (Figure 4.1-2) (see Chapter 2). Over 600 million pounds of oily waste material were collected and transported to disposal facilities. The volume of this waste could have filled approximately 80 football fields 3 feet deep (EPA 2011).

***Source**: U.S. Coast Guard photo by Petty Officer 3rd Class Stephen Lehmann (top left), Mabile and Allen (2010) (top right), (Zengel & Michel 2013) (lower left), NOAA/Scott Zengel photo (lower right).*

Figure 4.1-2. Response activities. Top left: Aircraft applying dispersant. Top right: In-situ oil burns conducted on June 18, 2010. Lower left: Example of boom stranded in the marsh, Bay Jimmy, LA, June 2010. Lower right: Workers manually raking oiled marsh vegetation in Barataria Bay, LA, October 2010.

- **Ecological scope.** The ecological scope of this incident was also unprecedented, with oiling occurring in the deep ocean a mile below the surface and in offshore "blue-water" habitats, *Sargassum* habitats, coral habitats, and nearshore and shoreline habitats (see Section 4.2).

In designing their injury assessment approach, the Trustees considered the practical realities resulting from this enormous scope:

- The spatial extent, type, degree, and duration of oiling were vast, both geographically and in terms of the complex set of habitats affected, from the deep sea to the coastline.
- The geographic scale, extent, and diversity of response activities, which had also impacted Gulf resources, was huge.
- The number of species, resources, and services potentially injured was vast.
- Evaluation of all potentially injured natural resources in all potentially oiled locations was cost-prohibitive and scientifically impractical.

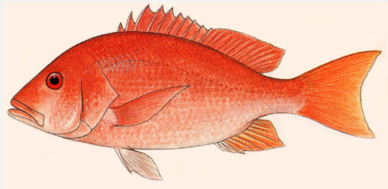
For these reasons, the Trustees determined that it was not feasible to study every species or habitat potentially affected by the incident in all locations exposed to oil or response activities. Instead, they employed an ecosystem approach to the assessment by evaluating injuries to a suite of representative habitats, communities, and species, as well as select human services, ecological processes, and ecological linkages. The Trustees used the information collected to develop scientifically informed conclusions not only about injury to the resources, processes, and locations studied, but also, by scientific inference (Section 4.1.5.3), about injury to resources, ecological processes, and locations that they could not directly assess.

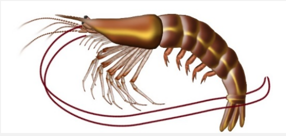
The oil discharged into the environment is a complex mixture containing thousands of individual chemical compounds (Section 4.2). Once in the environment, those chemical compounds, in turn, may change as they are subject to natural processes such as mixing with air and water, microbial degradation, and exposure to sunlight. As described in Sections 4.2 and 4.3, the Trustees' injury assessment considered, to the extent feasible, this suite of chemical and physical environmental stressors, including the effects of oil and response actions individually and collectively.

The Trustees Used Information from the Representative Resources, Processes, and Locations They Studied to Reach Broader Ecosystem Conclusions

Due to the unprecedented scale of the *Deepwater Horizon* incident, the Trustees evaluated injuries to *representative* habitats, communities, and species, and to *representative* ecological processes and linkages. They applied scientific inference to the results to develop broader conclusions about injury to northern Gulf of Mexico resources they could not directly study during the assessment. For example:

- In the water column resources assessment (Section 4.4), the Trustees used representative species such as red snapper, sea trout, and mahi-mahi to evaluate injuries to the large variety of fish species in the Gulf.





- In the nearshore marine ecosystem assessment (Section 4.6), the Trustees used species such as brown shrimp, red drum, and oysters to represent the many different fauna that rely on the edges of coastal salt marshes.
- To assess injury to coastal marshes, which support several important ecosystem processes (see Chapter 3), the Trustees considered one of these processes (the role of healthy marsh habitat in stabilizing the marsh and slowing coastal erosion rates) as representative of other ecological processes that marshes support.

4.1.3.2 Biological Scales of the Trustees' Injury Assessment

Injury assessments can be conducted at different scales of biological organization:

- **Organisms** can be evaluated at scales ranging from cellular processes that underlie physiological fitness, to the health or survival of individual organisms, to the status of sub-populations or populations of species.
- **Species** can be evaluated individually or in terms of the sometimes complex multi-species communities on which they rely. Communities of organisms are supported by habitats and ecological landscapes.

OPA regulations do not specify what scale(s) or organization to use in an injury assessment [see 15 CFR § 990.51(c)]. Rather, the OPA regulations leave the consideration and selection of injuries to include in the assessment to the discretion and expertise of the Trustees. As an ecosystem-level assessment, the Trustees' injury assessment evaluated injuries across a range of components and functions of Gulf of Mexico ecosystems. Thus, the Trustees' injury assessment was conducted at multiple scales of organization, including the cellular, individual, species, community, and habitat levels. In addition, the Trustees' assessment considered organism life history requirements and reproductive biology by evaluating injuries to embryonic and juvenile organisms and adult organisms. Using this approach, the

Trustees can interpret conclusions they derive from the injury assessment broadly over different scales of biological organization.

4.1.4 Injury Assessment Timeline and Stages

Figure 4.1-3 shows a generalized timeline for the Trustees' injury assessment process, from 2010 to 2015. This process involved several stages, described below. Throughout, the Trustees pursued an *iterative* injury assessment process. In other words, on a continual basis, they have used the data and results from work already performed to inform design and planning of subsequent assessment efforts.

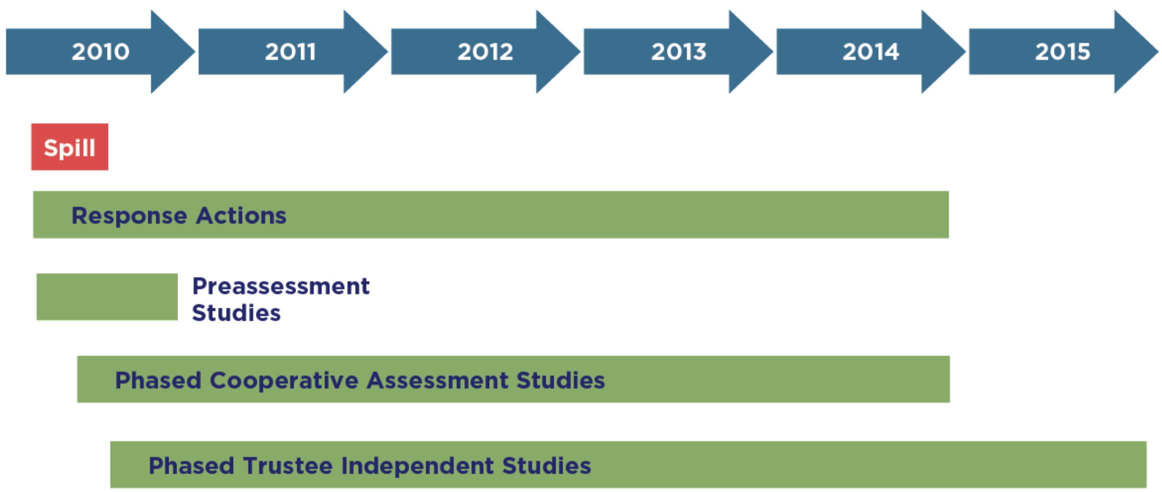


Figure 4.1-3. Generalized timeline for the Trustees' iterative, phased assessment process.

4.1.4.1 Immediate Data Collection

The Trustees' injury assessment started as soon as news of the spill was received. Working together with oil spill response efforts, the Trustees mobilized teams of scientists to rapidly evaluate the potential for injury, taking into account modeled and observed oil trajectories along the ocean surface, their experience at other oil spills, and their fundamental understanding of the natural resources of the northern Gulf of Mexico.

4.1.4.2 Data from Response Efforts

Immediately following the spill, response efforts were initiated by numerous government agencies, overseen by the U.S. Coast Guard (USCG 2011). BP participated actively throughout the response, working collaboratively with the U.S. Coast Guard and other response agencies as required by OPA.

Although not formally part of the injury assessment, response efforts collected considerable amounts of data to inform and support cleanup decisions. These data include environmental samples, extensive shoreline observations to identify oiled locations, collections of live oiled organisms (with subsequent efforts at rehabilitation), and identification and collection of dead animals. The Trustees judged that certain response data, though collected for another purpose, had value for the injury assessment; they relied on response data, where appropriate, in their injury assessment.

4.1.4.3 Preassessment Studies

Shortly following the spill, the Trustees initiated a series of preassessment studies designed to provide initial information to inform Trustee decisions about potential injury assessment studies.[2] The Trustees relied on these results in the NRDA and interpreted them in conjunction with subsequent data collection efforts.

4.1.4.4 Cooperative Assessment Studies

Following the preassessment phase, the Trustees and BP initiated a series of cooperative assessment studies, pursuant to a Memorandum of Understanding between the Trustees and BP,[3] designed to broadly evaluate the potential for injuries to a variety of natural resources over a wide geographic area. The cooperative nature of these studies was generally limited to the collection of certain data and not the analysis of data or interpretation of the results. In many instances, cooperative studies were performed in a phased manner, often over several years.

4.1.4.5 Independent Assessment Studies

In addition to cooperative assessment studies, the Trustees conducted a series of independent assessment studies that focused on discrete issues of concern to the Trustees. Independent assessment studies included human use studies, toxicity testing studies, and other resource-specific studies and evaluations. The Trustees also analyzed and evaluated data independently from BP.

4.1.5 Injury Assessment Methods

4.1.5.1 Assessment Procedures

OPA regulations identify different assessment procedures for use in an injury assessment, including field studies, laboratory studies, model-based approaches, and literature-based approaches (15 CFR § 990.27). The Trustees used combinations of all four approaches. As described in Sections 4.1.5.1.1 and 4.1.5.1.2, they used:

- **Field studies** to measure environmental exposure to oil, evaluate biological responses, and quantify lost human uses.
- **Laboratory studies** to evaluate the toxicological responses of organisms to *Deepwater Horizon* oil under controlled conditions.
- **Scientific literature** to supplement their injury assessment.
- **Model-based approaches** to quantify exposure and injuries to resources where direct measurement was infeasible given the scope of the incident. For example, numerical modeling was employed to quantify injuries to nearshore resources based on exposure to toxic concentrations of oil, and to quantify injuries to marsh fauna such as flounder and shrimp.

[2] See work plans available at *http://www.doi.gov/restoration/Gulf-Coast-Oil-Spill-Work-Plans.cfm* and *http://www.gulfspillrestoration.noaa.gov/oil-spill/gulf-spill-data/*.
[3] See work plans available at *http://www.la-dwh.com* and *http://www.gulfspillrestoration.noaa.gov/oil-spill/gulf-spill-data/*.

4.1.5.1.1 Use of Field, Laboratory, and Literature Information

The Trustees used field studies when, in their judgment, such studies would yield valuable and usable observational data to inform the injury assessment. They used laboratory studies to evaluate the toxicological responses of different organisms to oil and dispersant under controlled conditions. They also relied on published literature to support the injury assessment, including previously published studies about how oil affected natural resources, as well as academic research conducted outside the NRDA but related to the oil spill and/or to the Gulf's natural resources. The Trustees also considered independent data collected by BP.

4.1.5.1.2 Use of Numerical Models

When field studies were judged infeasible or impractical (e.g., when sampling was impractical or sufficient samples could not be collected to support robust conclusions), the Trustees used alternative assessment approaches, such as numerical modeling, to determine and quantify injuries based on known data or environmental processes.

Widely utilized in environmental science fields, numerical models simulate or calculate quantitative relationships between environmental variables based on understood and hypothesized conditions. The Trustees used numerical models as part of their assessment because, as discussed earlier, they could not measure oil concentrations everywhere in the northern Gulf of Mexico or directly study the impact of the incident on the vast number of potentially affected species and habitats. For example, as described in Section 4.4 (Water Column), because the Trustees could not study the response of every potentially affected fish species (and life stage) to oil exposure, they developed numerical models to quantify injuries to fish based on:

- Toxicological dose-response relationships derived from controlled laboratory studies.
- The estimated abundance of fish from available survey data.
- Modeled concentrations of oil in water developed from understood physical-chemical relationships.

When developing such models, the Trustees used empirical data from the Gulf (when available), well-understood environmental processes, and standard approaches in the field of environmental modeling.

4.1.5.2 Use of Scientific Inference

As noted earlier, the scale of the *Deepwater Horizon* incident precluded studying all individual components of the affected ecosystem, in all locations affected, over the full time period of potential effects. Instead, the Trustees' injury assessment focused on representative habitats, communities, species, processes, and linkages. To assess injuries not directly studied, The Trustees applied scientific inference to the study results to make informed conclusions about resources and locations that could not practically be assessed.

Scientific inference is the process of using data, observations, and knowledge to make reasonable conclusions about things that may not have been directly observed. For example, the Trustees may use observations and data they obtained showing that oil in sufficient amounts can smother wetland plants to infer that similar plants they did not study, when similarly oiled, would also be smothered. Similarly, existing knowledge can support reasonable scientific inferences. For example, if prior published studies have shown that certain species of organisms depend on marsh plants, scientists can reasonably infer that loss of those marsh plants in the Gulf would harm those dependent organisms. Section 4.1.3 provides more details on the types of scientific inference used by Trustees.

4.1.5.3 Trustees' Approach to Addressing Uncertainty

In scientific studies, scientists use uncertainty as a measure of how well they know something. For example, scientists may have a very good idea of the approximate number of dolphins living in Barataria Bay, Louisiana, but will use the term "uncertainty" to describe the degree to which they are not certain about the precise number of dolphins.

Scientific Uncertainty

In science, use of the term "uncertainty" does not mean that scientists don't know something. Rather, scientists use uncertainty to describe how precise their understanding is.

Scientific studies and findings typically involve some degree of uncertainty due to factors such as:

- Natural environmental variability.
- Variability related to sampling and measurement.
- Limited ability to collect data.
- Basic unknowns about the systems being studied.

In the case of the *Deepwater Horizon* incident, several factors introduce scientific uncertainties. For example, the ecology of the deep sea is not yet well understood, knowledge about many resources of the open ocean is limited, and scientists' ability to collect samples in and study these environments is limited. Many of the Gulf resources affected by the incident are highly mobile and difficult to observe, posing practical limitations on sampling and study. The incident's spatial scale was sufficiently large that scientists cannot survey or sample all areas that may have been injured.

These uncertainties do not mean that the Trustees have not been able to reach a series of scientifically robust and accurate conclusions. Rather, the scientific uncertainty discussed in this injury assessment provides transparency into the precision of the Trustees' numerical findings.

The Trustees employed a number of commonly used approaches to address uncertainty, including use of alternative model scenarios, presentation of injury calculations in reasonable numerical ranges to reflect uncertainty, statistical analysis of uncertainties through sensitivity and randomization analyses, and calculation of confidence intervals and error rates (NRC 2004).

In evaluating scientific uncertainties, the Trustees weighed information against two different possible errors:

- **Type I error:** a conclusion that injury had occurred when, in fact, there was no injury.
- **Type II error:** a conclusion that injury had not occurred, when in reality an injury had occurred.

In the case of the *Deepwater Horizon* incident, a Type I error would lead to overestimating restoration needs, whereas a Type II error would lead to underestimation, resulting in insufficient restoration.

Scientists often use statistical analysis (see Section 4.1.5.4) to describe the level of uncertainty. In the environmental sciences, use of statistical analysis frequently minimizes Type I errors, particularly in variable natural environments.

The Trustees considered both Type I and II errors during injury assessment and restoration planning.

The purpose of this Final PDARP/PEIS is to determine the need for and decide on restoration. Not all uncertainty must be fully resolved to meet these objectives. Even with more time and resources for scientific study, substantial uncertainty would remain. While further study could somewhat decrease uncertainty, the Trustees do not expect that the degree of increased certainty would change their selection of the preferred alternative presented in Chapter 5 of this Final PDARP/PEIS.

The extensive scientific data and conclusions set out in this Final PDARP/PEIS can address most of the questions the Trustees face regarding the nature and extent of injuries and the need for required restoration actions. However, to make all the determinations required to fulfill their trust responsibilities, the Trustees must exercise informed judgment in light of expert opinion to address remaining uncertainties and unresolvable data gaps. The result, reflected in this document, is a series of critical decisions based on a combination of the best available scientific information, agency expertise, and extensive experience gained from other cases.

4.1.5.4 Statistical Analysis

The Trustees and their principal investigators selected a variety of statistical approaches for the injury assessment based on the type of data collected, the nature of the study and study design, and the specific questions to be addressed in analyzing the data. Examples include:

- **Regression analysis.** This involves analyzing the numerical relationship between a dependent ("response") variable and one or more independent ("driver") variables. With regression analysis, the Trustees can calculate how changes in a driver variable will result in changes to a response variable (e.g., how increases in oil concentrations are related to increases in an organism's mortality rates).

- **Tests comparing mean (i.e., average) values of two or more groups.** These include t-tests (to compare means of two groups) and analysis of variance (ANOVA, to compare means of multiple groups). The Trustees used these methods to determine whether the average condition of a variable differed between groups (e.g., to compare average vegetation health at oiled and unoiled sites).

- **Tests comparing attributes of individual organisms against reference values.** The Trustees used these types of tests to determine if the value of a parameter associated with a particular organism, or locality, was typical or atypical relative to a known standard value (or range of values) for that parameter. For example, physiological data collected from an individual organism (such as a hematocrit level measured in a sample of bird blood) may be compared with a reference interval developed from hematocrit levels measured in unaffected birds to determine whether the condition of the sampled organism is impaired.

- **Geostatistical modeling techniques, such as kriging.** These are used to evaluate spatial patterns in data. A kriging model uses information about attributes of sampled locations (e.g., oil concentrations in shoreline sediments) to describe how those attributes are similar or different across a landscape. A kriging model may be used to infer conditions in unsampled locations based on the spatial patterns found among sampled locations.

Trustees' Approach to Determining Statistical Significance

For this injury assessment, the Trustees did not adopt a single universal threshold value for statistical significance. Rather, the investigating scientists interpreted the data as they judged appropriate, considering factors such as the resource investigated, type of data collected, statistical power of a test to detect a difference, and the associated possibility of making a Type II error (i.e., concluding there is no effect when there actually is one—see Section 4.1.5.3, above). Two techniques used by investigators to determine significance are the use of p-values and confidence intervals.

P-values and the null hypothesis. An output of some statistical tests, the "p-value" refers to the probability that patterns in sample data occurred through random chance alone, rather than as a result of an actual effect being evaluated. Calculations to determine p-values are based on clearly defined statistical models, such as the widely used "null hypothesis." A typical null hypothesis might be that the mean value of a parameter of interest in each of two populations is identical. Because of variability in a study, the difference in the mean values determined from *samples* collected from two populations is likely to be non-zero even if the difference in *actual* population means is in fact zero. The probability of finding a particular degree of difference between sample means, if the null hypothesis is true, is defined as the p-value. Briefly, if the differences in sample means are "large enough," the results are interpreted as evidence *against* the null hypothesis. Small p-values are interpreted as evidence that the null hypothesis is unlikely to be true (i.e., there is a small probability that the null hypothesis is true).

Confidence intervals. In the injury assessment, the Trustees used another type of statistic: the "confidence interval." A confidence interval is derived from sample data and may be used to qualify the mean value of sample data as a reflection of the true, but unknown, population mean. For example, a 95 percent confidence interval around a sample mean denotes that there is a 95 percent probability that the true population mean lies within that interval.

4.1.5.5 Overarching Injury Quantification Factors

The Trustees' approaches to quantifying injuries varied by resource type and according to the specifics of the scientific study. Details are provided in subsequent sections of this chapter. Two principles, described below, impacted quantification design. The Trustees did not conduct population-level assessments, and they placed limited reliance on counts of animals killed by the incident.

As a general matter, the Trustees did not quantify effects in terms of changes in population size or status. There are a number of reasons why the Trustees concluded that seeking to quantify population-level changes would be of little scientific value for the assessment.

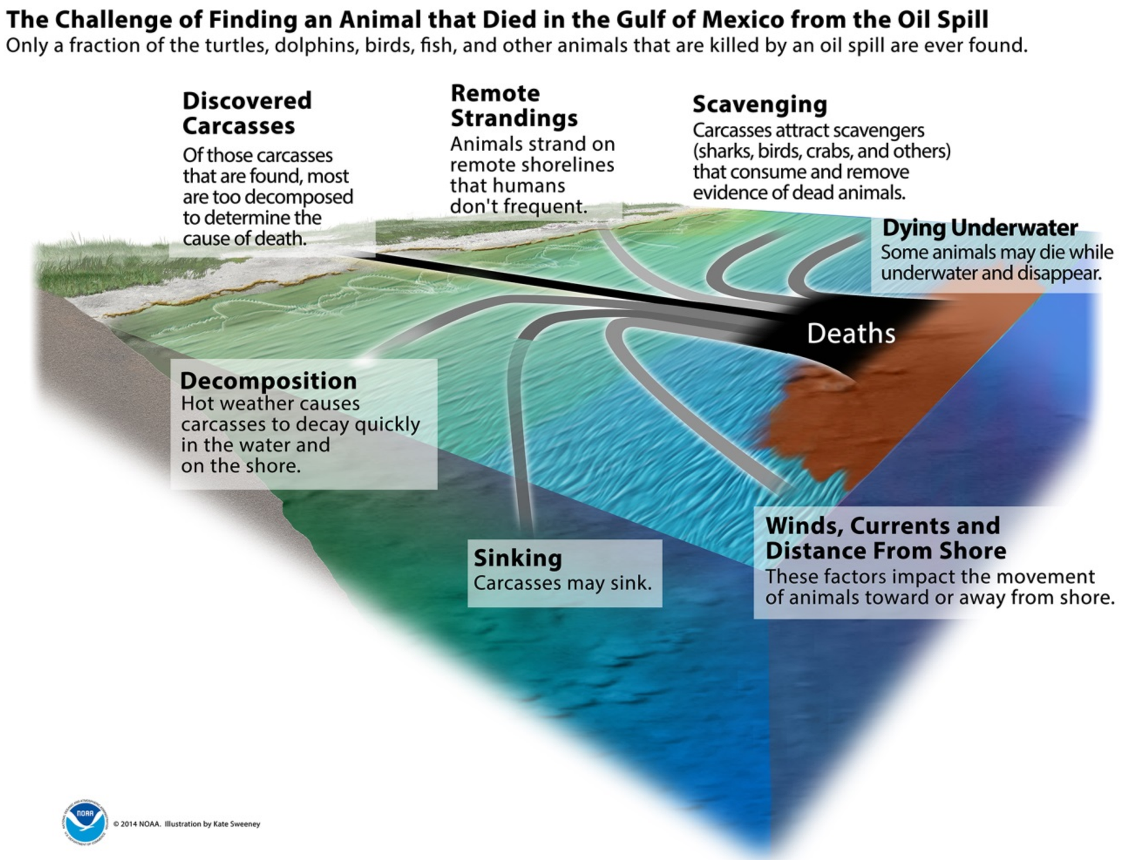
> "[P]opulation-level effects are inherently difficult to assess because of high variability, migrations and multiple factors affecting the populations."
>
> Boesch (2014)

The Gulf of Mexico covers a very large area that includes a U.S. shoreline length exceeding that from Florida through Maine, as well as a vast ocean area and volume. Organisms move freely within this system and between the Gulf, the Caribbean Sea, and the Atlantic Ocean. Given the natural changes and variability, assessing population-level impacts within such a huge area where organisms move freely is impractical. The sheer number of samples that would be required for a population-level assessment renders such approaches unrealistic and potentially misleading. As noted by Boesch (2014), analyses of such broad regional trends are incapable of quantifying impacts on resources within the geographic areas exposed to substantial oiling.

In their injury quantification for some resources, Trustees relied on collection and counts of animals killed by the spill. However, due to difficulties associated with observing and collecting carcasses of killed animals over such a vast geographic and temporal scale, such counts drastically underestimate injury. The Trustees therefore developed methods and models that accounted for these challenges to develop a representative quantification of actual loss.

As illustrated in Figure 4.1-4, carcass collection studies are greatly limited by a number of different factors, including:

- Animals that die often are consumed by predators or sink before being observed.
- Animals that die offshore may not be observed unless they are pushed by winds/waves to shoreline areas where observation likelihoods may be greater.
- The Gulf's warm weather causes rapid decomposition of carcasses, rendering them impossible to observe.
- Carcasses that do make it to shorelines may end up in locations such as marshes that are remote or otherwise difficult to sample. For example, Louisiana's intertidal marshes are extremely difficult to survey.
- Small organisms, such as juvenile fish and crustaceans, are virtually impossible to observe, even when studies are designed to survey them.



Source: *Kate Sweeney for NOAA.*

Figure 4.1-4. Quantifying injuries based on observed dead animal carcasses is extremely difficult and leads to drastic underestimates of injury.

4.1.6 Trustees' Data Management Process and Systems

4.1.6.1 Types of Data Collected

Over the course of the injury assessment, the Trustees collected information to document the quantity and location of oil in the environment, as well as the ways in which the oil affected natural resources:

- **Field data.** Data collected in the field included photographs; global positioning system (GPS) data on locations and movements; videos; instrument data; physical samples of water, air, tissue, and sediment; direct observations (e.g., plant stem length, type of marsh vegetation); carcasses of thousands of dead and dying birds; telemetry information from fish, turtles, and marine

The injury assessment relied on data collected by Trustees as part of NRDA studies, as well as data and information collected by other researchers and agencies. Each NRDA study was conducted under a work plan, which is available for public review. These work plans can be found at the NOAA Gulf Spill Restoration site: *http://www.gulfspillrestoration.noaa.gov/oil-spill/gulf-spill-data/*.

mammals; and remote-sensing information. Field data collection involved roughly 20,000 trips and generated over 100,000 samples of water, tissue, oil, and sediment, and over 1 million field data forms and related electronic files.

- **Chemical testing data.** Chemical testing performed on samples resulted in over 4 million laboratory result records, as well as about 1 million records of the biological and physical composition of the samples.
- **Other data.** This work also generated over 1 million additional instrument files, photographs, telemetry records, and observation records.
- **Lost human use data.** The Trustees also collected substantial amounts of data for their evaluation of lost human use.

4.1.6.2 Data Management

The Trustees developed a robust set of data management protocols and systems to manage how field samples were collected and how the resulting samples were handled. These protocols and systems allowed Trustees to track collections through several distinct stages, including the data intake process, quality assurance/quality control (QA/QC) verifications (e.g., double transcription when turning written field sheets into electronic files), analysis at a laboratory, validation of analytical data by a third party, and final addition of the data to the appropriate databases.

To store data generated during the NDRA process, the Trustees developed a set of databases. Together, these databases constitute the Trustees' overall data repository, or "data warehouse," referred to as the **Data Integration, Visualization, and Reporting system (DIVER)**. DIVER is publicly accessible at *https://dwhdiver.orr.noaa.gov/*.

Data managers organize all the data elements (i.e., each sample, data point, analysis, or photograph) so that data users can systematically track and analyze the information. All elements are linked to the original information collection activity to better characterize data origin and to trace how each data element passed from one official handling stage to the next (known as "chain of custody"). In addition to field data, DIVER house information on laboratory experiments and other analyses conducted by NRDA investigators.

4.1.6.3 The NRDA Data Management Process

Figure 4.1-5 shows the steps in the Trustees' data management process:

- **Work plan.** The first step generally is to develop a work plan to collect data that will answer specific questions about the incident. *Deepwater Horizon* NRDA work plans can be found at *http://www.gulfspillrestoration.noaa.gov/oil-spill/gulf-spill-data/*. Then, the Trustees generally work with the data management team to plan field events, train the sample collectors on proper data management protocols, and coordinate with sample intake teams on where and when samples will be handed over.
- **Sample collection.** Sample collection procedures are governed by scientific protocols. Upon collection, each sample is given a unique sample identification (ID) that follows the sample

throughout its life cycle. Sample documentation includes the field sheets that scientists use to write down sample locations, time, environmental conditions, etc.; the chain-of-custody forms; the original photographs; GPS files; mailing labels routing the sample to the proper laboratory; and any other information associated with that particular collection.

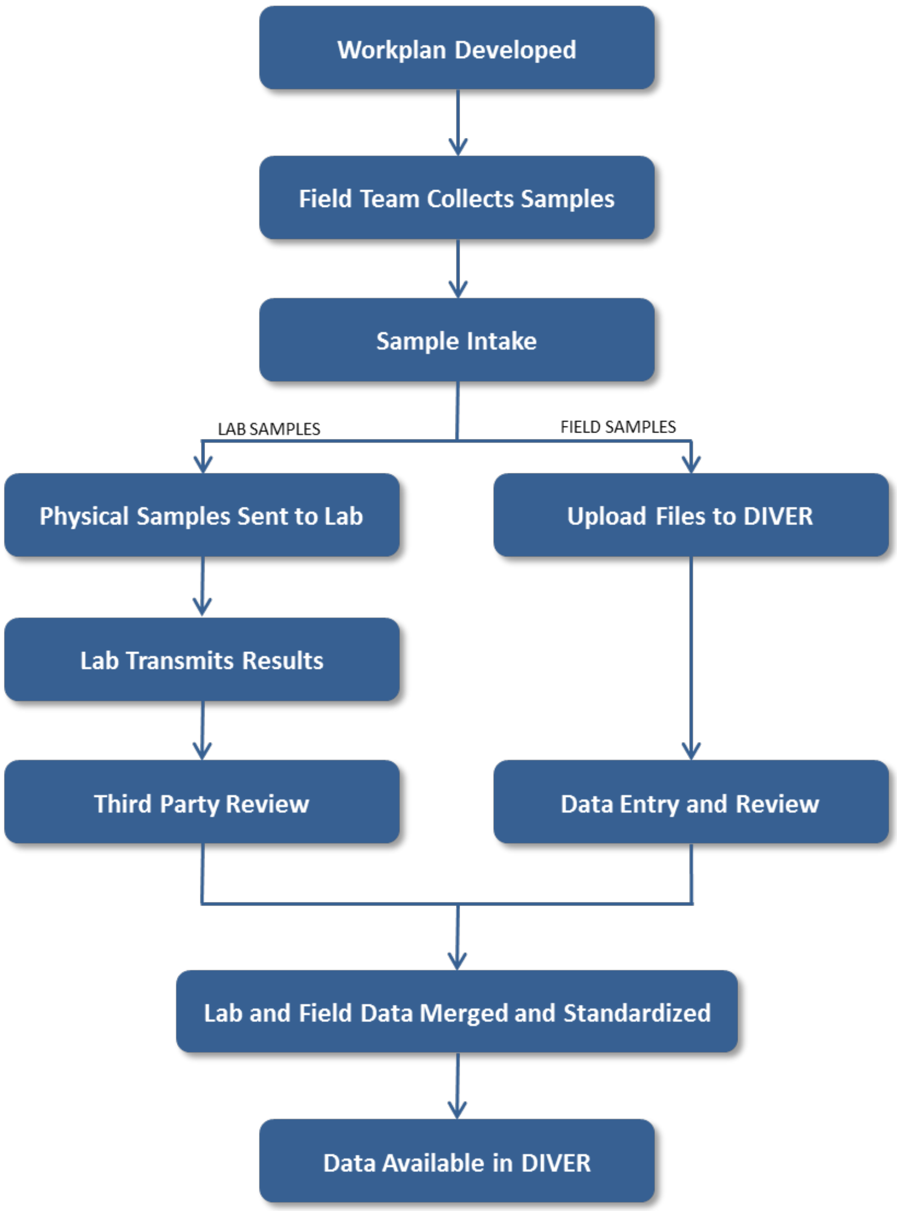


Figure 4.1-5. The Trustees' data management process for the *Deepwater Horizon* NRDA.

- **Sample intake.** Following collection, field researchers submit samples and associated documentation to one of several sample intake centers in the Gulf region. Sample intake teams are responsible for uploading the complete bundle of information—both electronic files and scanned paper forms—to DIVER. The sample intake and data management teams also provided

QA/QC checks to ensure that all documentation is complete and has been correctly transcribed and standardized (e.g., consistent measurement units and species names).

- **Laboratory samples.** The intake teams mail the samples to the appropriate laboratories for analysis. Laboratory researchers process samples and instruments according to procedures specified in the Analytical Quality Assurance Plan (AQAP) or work plan. All samples analyzed for contaminant chemistry (e.g., water, sediment, fish tissue, marine mammal blood samples, source oil) generally are sent to a third party to validate the laboratory results. The validators ensure that analytical equipment was calibrated correctly and that results meet performance standards. The *Deepwater Horizon* AQAP outlines all validation protocols and performance standards.

- **Field samples.** After intake, field samples are uploaded to DIVER for data entry and review.

- **Merge and standardization of laboratory and field data.** After the analytical results have been validated, they are integrated with the corresponding field data and made available within DIVER.

4.1.6.4 Data Management Systems

The Trustees' data management system comprised several components, including:

- **DIVER.** As described earlier, NOAA and the data management team created DIVER to serve as a warehouse and portal for all data related to the *Deepwater Horizon* NRDA effort. DIVER is designed to address the unique data demands associated with the *Deepwater Horizon* incident. DIVER allows the user to access not only analytical chemistry results, but also original field data, work plan-specific observation data, photographs taken during sampling trips and other field research, instrument data, and information on the status of samples as they proceed through laboratory analysis. Figure 4.1-6 depicts this data integration graphically.

- **Environmental Response Management Application (ERMA®).** NOAA's ERMA® is an online mapping tool that integrates static and real-time data in a centralized, easy-to-use map to provide environmental responders and decision-makers with faster visualization of the situation and improve communication and coordination for environmental response, planning, and restoration (see *http://gomex.erma.noaa.gov/erma.html*). During major response activities, ERMA was a main way the Trustees shared data publicly.

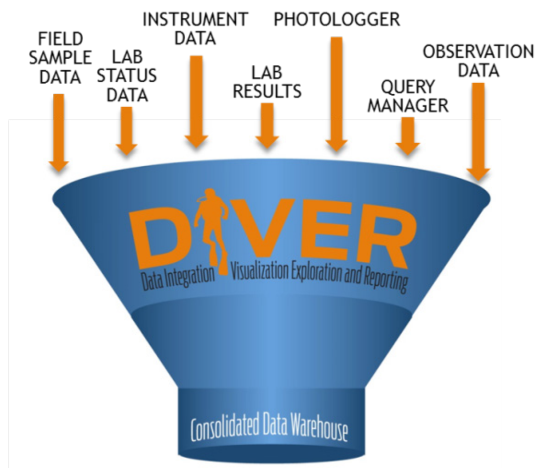


Figure 4.1-6. DIVER data integration.

- **Data management protocols.** Specific data management protocols were developed for distinct purposes. For example:
    - The field study of recreational uses included onsite surveys, ground counts, and oblique aerial imagery counts to count, for example, the number of individuals on a segment of beach. Sampling plans and data management protocols were developed to manage how the field and aerial data were collected, handled, and processed.
    - Toxicity tests required a tracking system to monitor and manage all phases of toxicity testing, from test development through to receipt of laboratory testing and analytical data, validation of chemistry data, QC, and analysis.

4.1.6.5 Data Tracking and Integrity

While samples were being analyzed, the Trustees' data management team tracked the status of the sample (i.e., whether it was in the queue, under analysis, archived, delivered to validation, validated, or shared publicly). As case data were integrated into the system repositories, the data management team conducted quality reviews to promote consistent data suitable for application in this Final PDARP/PEIS. These quality reviews included basic standardization (e.g., correcting misspelled species names, converting to consistent units), reviews to ensure record completeness and accuracy, and coordination with work plan principal investigators to make any necessary updates.

4.1.7 Road Map to the Trustees' Injury Assessment

Figure 4.1-7 provides an overall road map to the Trustees' injury assessment. As the figure shows, the Trustees' injury assessment, described in subsequent sections of this chapter, is organized into four elements: exposure assessment, toxicity assessment, resource-specific injury assessment, and summary and synthesis of findings.

- **Section 4.2 (Natural Resource Exposure).** The Trustees' exposure assessment provides a summary and synthesis of all the evidence gathered and analyzed about where and how natural resources were exposed to *Deepwater Horizon* oil, as well as other contaminants (e.g., dispersants) and stressors (e.g., drilling muds). The exposure assessment chronicles how the oil and other contaminants moved and changed as they were transported through water, sediment, air, and biota, and ultimately exposed many habitats, plants, and animals in the deep sea up to and across a broad expanse of the ocean surface and along the coastlines of the northern Gulf of Mexico.

- **Section 4.3 (Toxicity).** The Trustees' toxicity assessment involved detailed evaluation of the toxic effects of *Deepwater Horizon* oil and dispersants (Figure 4.1-8) to determine what concentrations cause toxic effects. The Trustees used this information together with the exposure assessment results when they evaluated injuries to natural resources in the northern Gulf of Mexico (Sections 4.4 to 4.10).

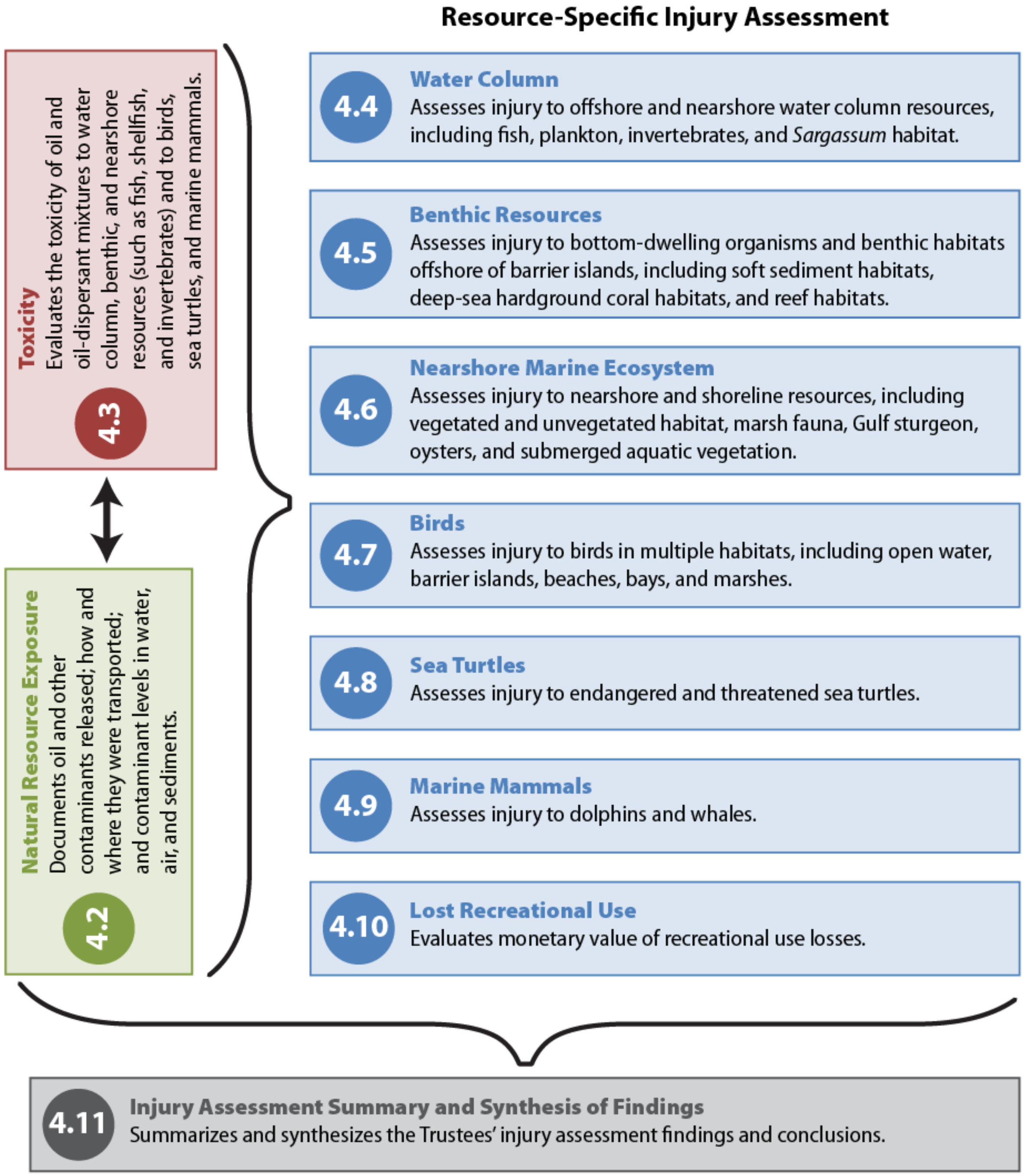


Figure 4.1-7. "Road map" to the Trustees' injury assessment presented in Chapter 4.

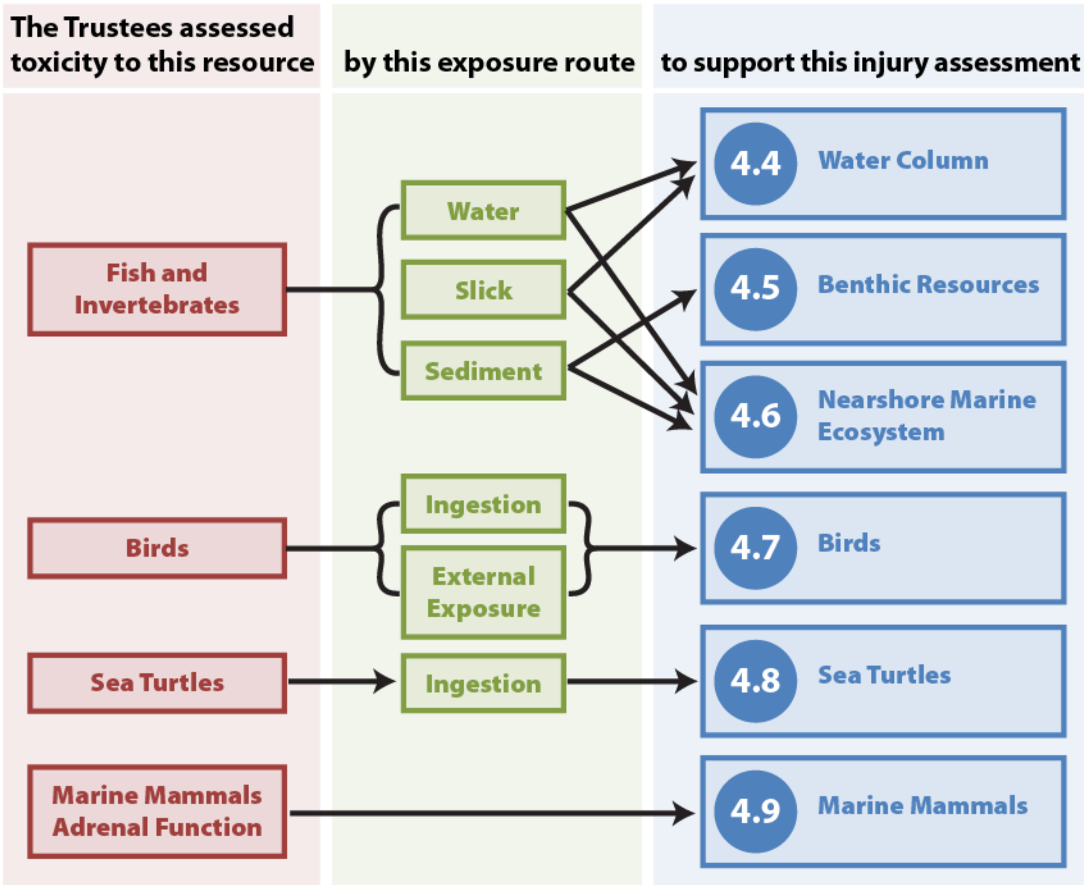


Figure 4.1-8. Use of toxicity information in the Trustees' injury assessment.

- **Resource-specific injury determination and quantification (Sections 4.4 to 4.10).** Approaches and results for each of seven resource categories are presented in separate sections:
    - **Section 4.4 (Water Column)** contains information on injuries to fish and invertebrates in the open water column and in nearshore waters. This resource analysis relies heavily on understanding the relationship between exposure to oil, the toxicity of the oil, and the application of numerical models.
    - **Section 4.5 (Benthic Resources)** describes a variety of soft-bottom (mud) and hard-bottom (rock and coral) habitats. Despite the challenges of working in the deep sea, this section relies largely on empirical field studies to document harm to sea bottom habitats and organisms.
    - **Section 4.6 (Nearshore Marine Ecosystem)** addresses a wide variety of habitats and representative species, including vegetation shorelines (e.g., marsh and mangroves), sand beaches, and submerged aquatic vegetation, as well as a series of representative resources that serve primarily as indicators of marsh habitat quality. This section combines extensive field data, toxicological data, published literature, and numerical models.

- **Section 4.7 (Birds)** describes injuries to a wide variety of bird species, including pelagic sea birds, colonial nesting birds, waterbirds, and marsh birds. The assessment relies on observations of numerous dead birds, numerical extrapolations of those dead birds to estimate a range of bird mortality where possible, and toxicity tests in which birds were exposed to oil under various conditions.
- **Sections 4.8 (Sea Turtles) and 4.9 (Marine Mammals)** describe the Trustees' assessment of injury to these highly charismatic organisms, which are protected by the Endangered Species Act and the Marine Mammal Protection Act. The sea turtle assessment relied on extensive observations of oiled turtles data from NRDA field studies, stranded carcasses collected by the NMFS SEFSC Sea Turtle Stranding and Salvage Network, historical data on sea turtle populations, and the published literature to develop opinions regarding sea turtle injuries, as supplemented by veterinary assessments of captured turtles and a laboratory study of surrogate freshwater turtles. The marine mammal assessment synthesized data from NRDA field studies, stranded carcasses collected by the Southeast Marine Mammal Stranding Network, historical data on marine mammal populations, NRDA toxicity testing studies, and the published literature.
- **Section 4.10 (Lost Recreational Use)** focuses on losses of human recreational use services attributable to the incident. Losses are expressed in terms of lost recreational trip opportunities and monetary damages.

- **Summary of Injury Effects and Quantification (Section 4.11).** The final section of Chapter 4 presents the Trustees' key findings and conclusions resulting from the injury assessment.

4.1.8 References

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4.2 Natural Resource Exposure

What Is in This Section?

- **Executive Summary**
- **Introduction (Section 4.2.1):** How does this section inform the Trustees' injury assessment, and what are key facts about the scope and scale of natural resource exposure to *Deepwater Horizon* (DWH) oil and other contaminants?
- **Contaminants Released During the Spill (Section 4.2.2):** What are oil and dispersants, and how were they measured in the environment?
- **Exposure in the Deep Sea and Sea Floor (Section 4.2.3):** To what extent were natural resources in the deep sea exposed to DWH oil and other contaminants?
- **Exposure Within the Rising Plume (Section 4.2.4):** To what extent were natural resources in the water column between the sea floor and ocean surface exposed to DWH oil and dispersant?
- **Exposure at the Sea Surface (Section 4.2.5):** To what extent were natural resources at the sea surface exposed to DWH oil and dispersant?
- **Exposure in the Nearshore (Section 4.2.6):** To what extent were natural resources in nearshore and shoreline habitats exposed to DWH oil and dispersant?
- **Conclusions (Section 4.2.7):** What are the Trustees' conclusions about the nature, spatial extent, and temporal extent of natural resource exposure to DWH oil and other contaminants?
- **References (Section 4.2.8)**

Executive Summary

- The DWH disaster released approximately 134 million gallons (3.19 million barrels) of oil and 1.84 million gallons of dispersant into the environment.
- Every day for 87 days, BP's Macondo well released an average of more than 1.5 million gallons of fresh oil into the ocean, essentially creating a new oil spill every day for nearly 3 months.
- Combining direct observations, remote sensing data, field sampling data, and other lines of evidence, the Trustees documented that oil:
  - Was transported within deep-sea water currents hundreds of miles away from the failed well.
  - Rose to the sea surface and created 43,300 square miles (112,115 square kilometers) of detectable oil slicks—an area about the size of the state of Virginia.

   - Sank onto the sea floor over an area at least 400 square miles (1,030 square kilometers).
   - Was spread by wind and currents and washed onto more than 1,300 miles (2,100 kilometers) of shoreline.

- Natural resources were exposed to oil, dispersant, or both across a broad range of habitats, including the deep sea; over 5,000 vertical feet (1,500 meters) of water column; the sea surface; and nearshore habitats, such as beach, marsh, mangrove, and submerged aquatic vegetation (SAV).
- A wide variety of biota, including fish, shellfish, sea turtles, marine mammals, and birds, were exposed to oil and/or dispersant throughout the northern Gulf of Mexico. Natural resources were exposed through various pathways, including direct exposure and contact with contaminated water, air, vegetation, and sediments.
- Despite natural weathering processes over the past 5 years, oil persists in some habitats where it continues to expose resources in the northern Gulf of Mexico.

4.2.1 Introduction

This section provides an overview of the nature of the oil and other contaminants that were released into the environment. It then describes the pathways by which the oil and other contaminants moved through the ocean to the sea floor, upward toward the oceanic surface waters, into the atmosphere, and onto the shorelines. In doing so, this section answers the following questions:

- How did the oil and other contaminants move from the wellhead throughout the northern Gulf of Mexico (how was it transported)?
- Where exactly did the oil go (what was its fate)?
- How did the oil change (weather) during its transport throughout the northern Gulf of Mexico?
- What were the levels of oil-derived contamination in water, air, and sediments throughout the northern Gulf of Mexico?

How Did the Trustees Confirm Exposure?

The Trustees examined many lines of evidence, including:

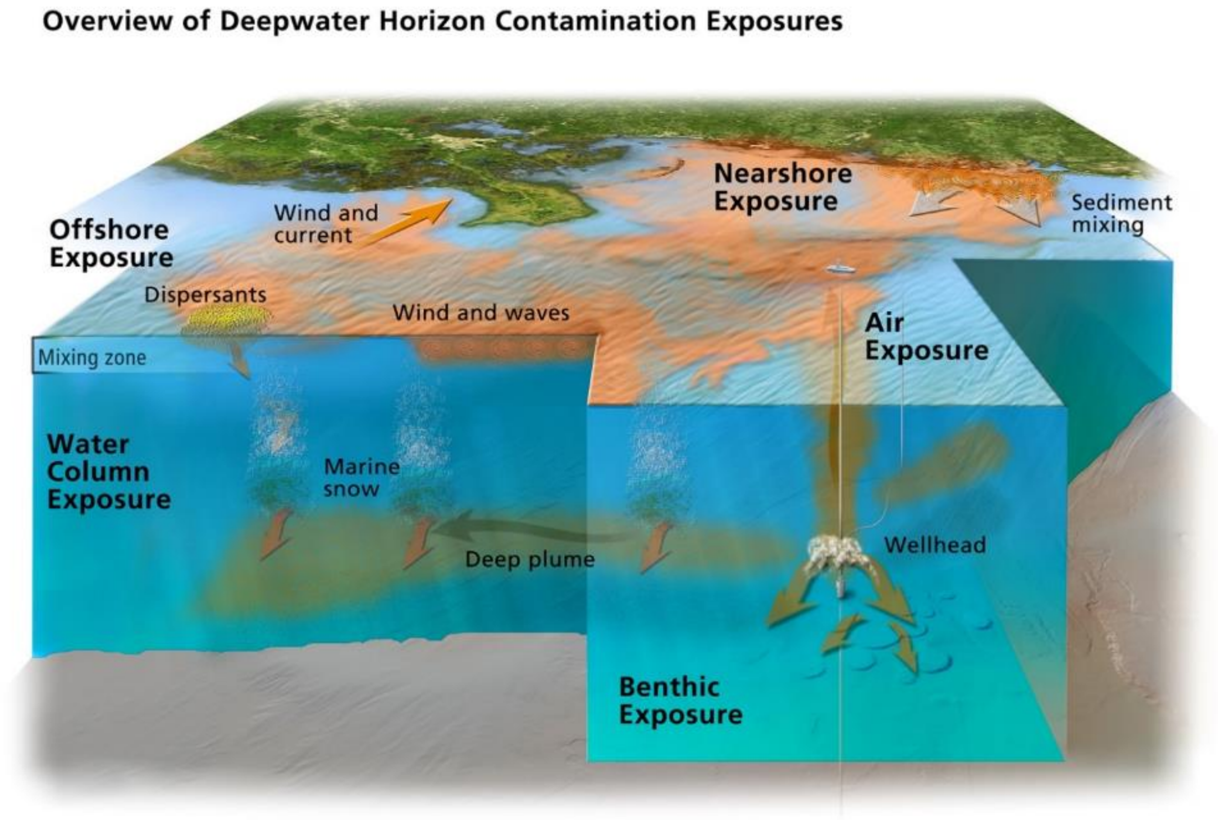
- Photographs and other direct observations from airplanes, helicopters, boats, and shorelines.
- Remote sensing data from both satellite- and airplane-mounted sensors.
- Fluorescence and other data collected from remotely operated vehicles.
- Data from thousands of samples of water, sediment, soil, and other media, confirming both the presence of oil and the specific "fingerprint" of DWH oil.
- Data from birds, dolphins, turtles, and other biota that were captured or had perished during the spill.

The answers to these questions form the foundation for determining the levels of exposure of natural resources to the oil and other contaminants released from BP's Macondo well, and ultimately to assessing the magnitude of the injuries to the natural resources of the northern Gulf of Mexico caused by this spill.

The April 2010 Macondo oil well blowout released 3.19 million barrels of oil (134 million gallons) into the northern Gulf of Mexico—the worst marine oil spill in U.S. history (Boesch 2014). The Macondo blowout occurred at a depth of about 1,500 meters (5,000 feet), some 66 kilometers (41 miles) offshore from the southeastern tip of Louisiana. The volume of oil spilled into the environment, the long duration of the release (oil flowed from the wellhead for 87 days), the depth from which oil was released into the ocean, and the large volume of dispersants applied (both at depth and on the surface) introduced unique challenges for the assessment of environmental harm (Rice 2014).

Once released from the failed Macondo well, DWH oil rose through the water column to the sea surface, creating massive oil slicks that moved throughout the northern Gulf of Mexico. These slicks affected natural resources in the water column, at the sea surface, in nearshore habitats, and along shorelines from Texas to the Florida Panhandle. For 87 days, BP's Macondo well released an average of nearly 38,000 barrels (1.5 million gallons) of fresh oil each day into the ocean. This is essentially equivalent to a substantial oil spill occurring every day for nearly 3 months, or the equivalent of the 1989 *Exxon Valdez* oil spill reoccurring in the same location every week for 12 weeks. The scope of the DWH incident was unprecedented in terms of the quantity of oil released, the release duration, the vertical and lateral extent of oil in the ocean, the spatial extent of oil spread on the sea surface, and the spatial extent of shoreline oiling. In turn, the scale of natural resources exposed to the spilled oil and other contaminants was also unprecedented. Natural resources were exposed repeatedly to DWH oil across a broad diversity of habitats.

Once released, DWH oil moved widely throughout the northern Gulf of Mexico (Figure 4.2-1).



Source: *Kate Sweeney for NOAA.*

Figure 4.2-1. Oil that discharged from the wellhead transported via multiple pathways. Some oil (and most of the natural gas) remained in the deep sea, forming deep plumes. Some oil that remained in the deep sea eventually accumulated on the sea floor. Some oil rose through the water column and reached the sea surface, forming large oil slicks. Those slicks were then transported around the northern Gulf of Mexico, with much of the oil entering nearshore habitats.

Oil slicks (e.g., Figure 4.2-2) cumulatively covered over 112,115 square kilometers (43,300 square miles) of the ocean surface (ERMA 2015), and oil contamination was documented on over 2,100 kilometers (1,300 miles) of shorelines. The Trustees estimate that DWH oil covered at least 1,030 square kilometers (400 square miles) of deep-sea habitat, with sea floor impacts from DWH oil extending beyond this zone.

More than 400 flights sprayed chemical dispersants on surface oil slicks (Houma 2010), and more than 400 fires were set on the sea surface to burn slicks (Mabile & Allen 2010).

The spatial extent of oil exposure and response activities was immense (Figure 4.2-3). Oil, dispersants, and drilling mud introduced in response to the oil spill traveled through the deep sea—some of which was deposited on the sea floor. The estimated 7.7 billion standard cubic feet (scf) of natural gas (based on an approximate gas to oil ratio of 2,400:1) (based on an approximate gas to oil ratio of 2,400:1; Zick 2013a; Zick 2013b) released along with oil from the well remained in the deep sea and was likely consumed by microbes. The microbes, as well as oily particulate matter and burn residues near the sea surface, sank to the sea floor in a so-called "dirty blizzard" of oily marine snow particles that was deposited on the sea floor as brown flocculent material ("floc").

The remainder of Section 4.2 is divided into six sections. First, Section 4.2.2 is an overview of the contaminants that were released into the environment. These contaminants included an estimated 134 million gallons of liquid DWH oil and 7.7 billion scf of natural gas that were discharged from the well. In addition, approximately 1.84 million gallons of chemical dispersants were intentionally introduced to the environment in an attempt to reduce the amount of oil that reached the ocean surface and sensitive shorelines (OSAT-1 2010). Finally, synthetic-based drilling mud—a dense fluid containing numerous synthetic chemicals—was also discharged into the deep sea after unsuccessful efforts to use the heavy mud to staunch the oil flow.

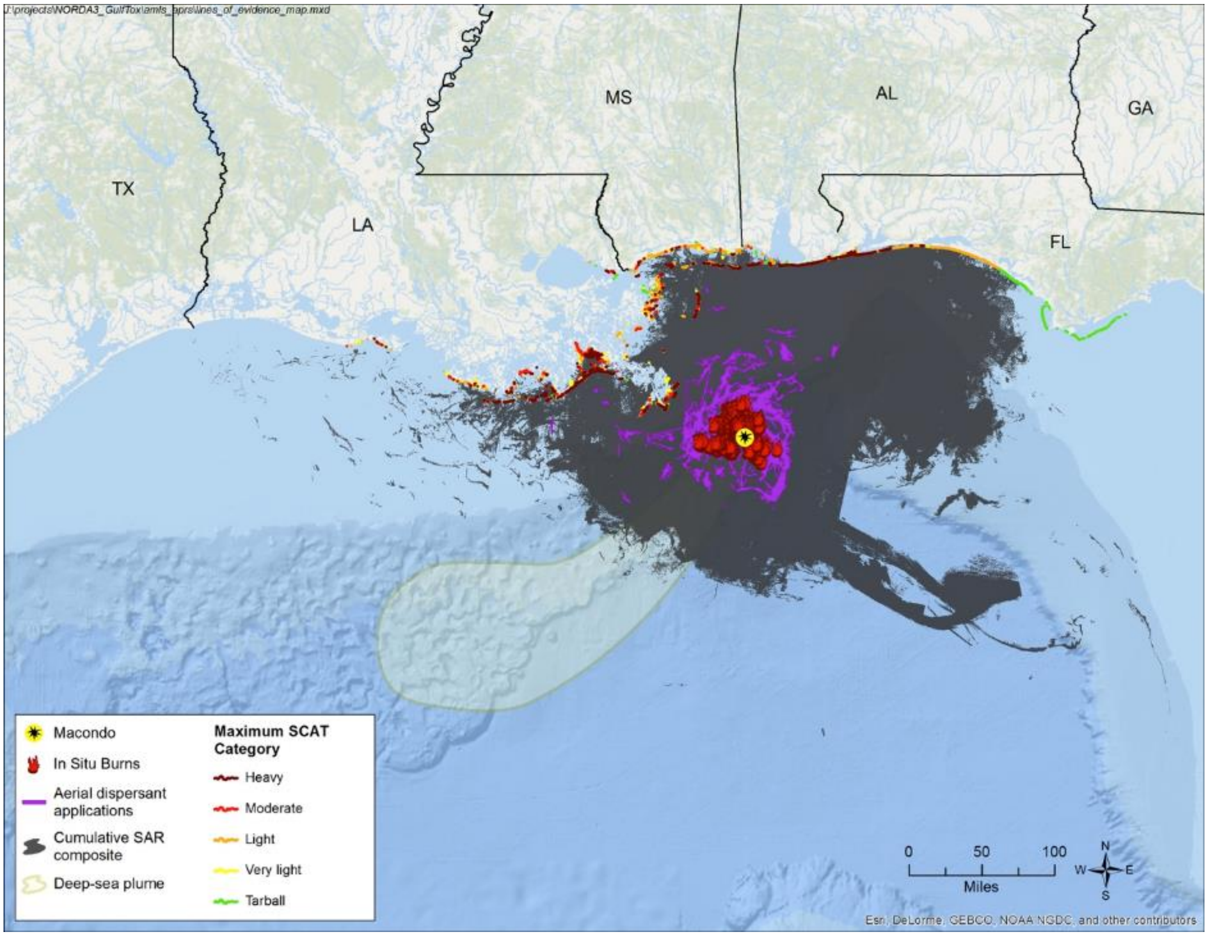
Source: *NOAA.*

Figure 4.2-2. Oil slicks on the Gulf of Mexico during the DWH oil spill, April 29, 2010.

The next four sections describe the exposure of natural resources to these contaminants across a broad diversity of habitats, including:

- Deep-sea water and the sea floor, including the continental slope and shelf (Section 4.2.3).
- The water column between the well and the sea surface (Section 4.2.4).
- The sea surface itself, including the shallow subsurface and near-surface air zone (Section 4.2.5).
- Coastal zones of the northern Gulf of Mexico in Texas, Louisiana, Mississippi, Alabama, and Florida, including diverse nearshore habitats such as beach, marsh, mangrove, and SAV (Section 4.2.6).

Finally, Section 4.2.7 presents the Trustees' conclusions about the nature, spatial extent, and temporal extent of natural resource exposure to DWH oil and other contaminants.



Source: *Abt Associates.*

Figure 4.2-3. The cumulative DWH oil footprint covered a large swath of the northern Gulf of Mexico. Most surface slicks traveled toward shore, intersecting at least 2,100 kilometers (1,300 miles) of shoreline; some slicks followed currents to the southeast. A deep-sea plume migrated more than 400 kilometers (250 miles) southwest of the well (Payne & Driskell 2015a). In response to the surface slicks, more than 400 flights quantitatively sprayed dispersant, and more than 400 fires were set to burn off surface oil. These data are all discussed in greater detail throughout this section.

4.2.2 Contaminants Released During the Spill

Key Points

- The Trustees determined the chemical compositions of the spilled oil and dispersants, as well as the synthetic-based drilling mud that was used unsuccessfully to try to plug the well and was subsequently deposited on the sea floor.
- DWH oil is composed of thousands of different chemicals, many of which are known to be toxic to biota.

- The chemical composition of the DWH oil changed (i.e., weathered) after it was released into the environment: lighter compounds were dissolved into water or evaporated into the air and heavier compounds were changed and concentrated.
- DWH oil has a specific chemical signature or "fingerprint" that, together with other lines of evidence, allowed the Trustees to determine which oil-derived contaminants found in the environment originated from the Macondo well.
- DWH oil, dispersants, and drilling mud were spread throughout the environment. For example, oil and synthetic-based drilling mud were deposited on the sea floor. Additionally, oil and dispersant injected at the wellhead were entrained both in deep-sea plumes and in plumes that rose through the water column, formed surface slicks, and were transported throughout the northern Gulf of Mexico.

The DWH disaster introduced numerous contaminants into the environment. The most obvious of these was the 3.19 million barrels (134 million gallons) of liquid oil. Additionally, an estimated 7.7 billion scf of natural gas were also discharged into the deep sea (Zick 2013a, 2013b), an estimated 1.84 million gallons of chemical dispersants were used in response to the spill (OSAT-1 2010), and an unknown volume (up to 30,000 barrels) of synthetic-based drilling mud was released during the blowout and response. Each of these contaminants introduced chemicals of known and unknown toxicity into the northern Gulf of Mexico. Finally, natural weathering processes (e.g., photooxidation) and intentional burning of the floating oil at sea formed additional contaminants, also of known and unknown toxicity.

In this section, the Trustees provide an overview of the primary contaminants released into the environment during the spill, including a summary of these contaminants' chemical characteristics that aided the Trustees in establishing that natural resources were exposed to them.

4.2.2.1 Oil and Gas

4.2.2.1.1 Composition

Crude oil contains thousands of organic (carbon-containing) compounds, most of which contain mixtures of carbon and hydrogen only (i.e., hydrocarbons). Hydrocarbons in crude oil range from light, volatile chemicals like those in gasoline to heavy, recalcitrant chemicals like those found in tar or asphalt.

Some of the more toxic compounds in crude oil are *aromatic* chemicals—a subset of organic compounds that share a common chemical structure, namely at least one benzene ring. These include mono-aromatic volatile organic compounds such as benzene, toluene, ethylbenzene, and xylenes (BTEX). These volatile aromatic hydrocarbons readily evaporate and are often responsible for the odors from petroleum. Another group of aromatic compounds is less volatile; these compounds are called polycyclic aromatic hydrocarbons (PAHs) because they contain two or more benzene rings (see the text box below).

The Trustees conducted detailed chemical analyses of "fresh" DWH oil samples collected directly from the riser pipe on the ocean floor; these analyses are summarized in technical appendices (Stout 2015a) and are consistent with other researchers' analyses (e.g., BP 2014a; Reddy et al. 2012). Understanding the composition of the "fresh" DWH oil revealed what types and amounts of specific chemicals were introduced into the environment during the spill.

When it is released into the environment, "fresh" oil immediately begins to change its composition through natural processes collectively referred to as *weathering*. Some compounds in oil are susceptible to weathering and others are not. For example, some susceptible compounds may quickly dissolve into the water, evaporate into the atmosphere, or be degraded by bacteria (i.e., biodegrade). Less susceptible compounds, which include some PAHs, do not readily dissolve, evaporate, or biodegrade and therefore become concentrated in the remaining weathered oil residue. The Trustees conducted many analyses on weathered DWH oil to understand which chemicals were weathered and which were not as the oil traveled through the environment (Section 4.2.2.1.2).

Oil on the ocean surface will mix with sea water as it weathers, creating an *emulsion* of oil and water. During the DWH incident, highly viscous and sticky water-in-oil emulsions on the sea surface often appeared reddish brown or orange (see Figure 4.2-2), in contrast to the typical black color of crude oil.

What Are PAHs?

PAHs—polycyclic aromatic hydrocarbons—are hydrocarbon compounds that contain two or more benzene (or aromatic) rings. Many compounds are considered PAHs, including naphthalene (two fused benzene rings), phenanthrene (three rings), chrysene (four rings), and benzo(a)pyrene (five rings). Low molecular weight PAHs (LPAHs), such as naphthalene, typically are more volatile and more soluble than high molecular weight PAHs (HPAHs), such as chrysene, which are less prone to weathering and generally more persistent.

Because of the known toxic effects of PAHs, scientists often present oil concentrations in terms of the concentrations of PAHs. In this Final PDARP/PEIS, the Trustees present PAH concentrations as the sum of 50 individually measured PAH chemicals. This sum is referred to as total PAH50 or TPAH50. Additional details are presented in Section 4.3 (Toxicity) and in Forth et al. (2015).

Different crude oils have different chemical compositions that are governed primarily by the geologic conditions under which they were formed, migrated, and accumulated. These conditions can result in oil from a given location or geologic formation having a unique chemical composition, including specific compounds that help experts distinguish one crude oil from another. This process of distinguishing one oil from another is called *chemical fingerprinting,* which is akin to how human fingerprints can uniquely identify an individual. Chemical fingerprinting was an important tool in determining exposure of the region's resources in that it could be used to recognize DWH oil. Chemical fingerprinting analyses were often used in conjunction with other lines of evidence to help further establish the presence of DWH oil in or on the region's resources. Combining multiple lines of evidence to determine whether oil in the environment originated from the DWH incident is referred to as *environmental forensics*. The Trustees employed chemical fingerprinting and environmental forensics (see text box below) on thousands of samples to establish the presence of DWH oil in resources throughout the northern Gulf of Mexico (e.g., Douglas et al. 2015; Emsbo-Mattingly 2015; Payne & Driskell 2015d; Stout 2015d). The forensic methods employed helped distinguish spilled DWH oil from other oils and other sources of PAHs; they also helped assess the extent to which DWH oil weathered in the environment, both shortly after it was spilled and years later.

In addition to the release of liquid oil, more than 7.7 billion scf of natural gas were also expelled from the Macondo well (Zick 2013a, 2013b). This natural gas contained more than 80 percent methane, with decreasing amounts of ethane, propane, butane, and pentane (Reddy et al. 2012). Much of the expelled gas was dissolved and biodegraded in the deep sea, but the expelled gas played an important role in the deep-sea exposure to the oil, as explained in Section 4.2.3.

Environmental Forensics: Recognizing DWH Oil in the Environment

Following the spill, the Trustees employed environmental forensics to establish the presence of DWH oil in resources throughout the northern Gulf of Mexico. A key component of this process was chemical fingerprinting of the oil, which could often confirm the presence of DWH oil in or on the region's resources. However, sometimes chemical fingerprinting was inconclusive because the DWH oil had become severely weathered or had become mixed with other "background" chemicals already in the environment. In these cases, experts examined chemical fingerprints in light of other lines of scientific evidence (e.g., spatial or temporal trends) and systematically interpreted data to determine if DWH oil residues were present—a process collectively called environmental forensics.

Using environmental forensics, Trustee scientists classified thousands of field samples into the following five generalized classes:

A Consistent with DWH oil
B Consistent with DWH oil with indication of weathering and mixing
C Possibly contains DWH oil based on additional lines of evidence
D Indeterminate (typically no or too little oil present to fingerprint)
E Inconsistent with DWH oil (a different oil is present)

Water samples warranted a somewhat different approach, because these often contained only the water-soluble chemicals that had dissolved from the DWH oil or dispersants. Nonetheless, environmental forensics helped classify thousands of water samples, and those analyses were supported by multiple lines of scientific evidence and systematic data interpretation.

4.2.2.1.2 Fate, Transport, and Weathering of Oil and Gas

As noted above, as soon as the DWH oil entered the deep sea, it began to change due to the process of weathering. The Trustees collected and analyzed thousands of samples that collectively revealed the progression in weathering and how it changed the composition of the expelled gas and oil as they moved through the environment. This provided an understanding of what happened to the oil and gas compounds after they were released from the well (fate) and where they went (transport), and which resources were exposed to these compounds. Fate and transport processes manifested differently for oil and gas:

- After oil was released into the ocean, weathering began to change its chemistry. Many particularly soluble compounds dissolved into the ocean's water, in the deep sea near the well, during the oil's ascent to the surface and after reaching the surface. At the surface, many of the volatile compounds evaporated. As such, the floating DWH oil was enriched in heavier compounds (insoluble and non-volatile). In addition, the floating oil was altered by chemical reactions caused by exposure of the oil to the sun (i.e., photooxidation) (Section 4.2.5.1).

- Most of the natural gas released from the Macondo well, on the other hand, remained within the deep sea (e.g., Reddy et al. 2012) and was likely consumed by bacteria that proliferated in response to the gas release (Kessler et al. 2011; Valentine et al. 2010). The gas-consuming bacteria proliferated, consumed the gas, and then died or were consumed by protozoa or small zooplankton. Mucus produced by bacteria, as well as some of the bacterial mass itself, agglomerated with brown-colored oil droplets and settled through the water column, giving rise to the term "dirty blizzard" (Schrope 2013). This *marine oil snow* formed a widespread brown floc layer on the sea floor that was observed by remotely operated vehicles (ROVs) and sea floor chemistry (Section 4.2.3.3.1).

Although the expelled gas remained in the deep sea and was consumed, the liquid oil released from the Macondo well followed one of three likely pathways (see Figure 4.2-1):

- Direct deposition on the sea floor, often in association with dense synthetic-based drilling mud, and within about 1.6 kilometers (1 mile) of the wellhead (Section 4.2.3.1).

- Entrainment in deep ocean currents in the form of small oil droplets that were not buoyant enough to rise through the water column to the sea surface and not dense enough to settle to the sea floor (Section 4.2.3.2). As noted above, these oil droplets within the deep-sea plume that formed were often carried to the sea floor as marine oil snow or, if not, were carried within the plume up to 400 kilometers (250 miles) from the wellhead before dispersing or becoming undetectable (Section 4.2.3.2).

- Entrainment in a rising buoyant plume of oil droplets that reached the ocean surface and formed oil slicks (Section 4.2.4). The surfaced oils likely followed one of four fate and transport pathways:

    - Some surface oil was re-entrained into the water column by wave action or by the application of chemical dispersants. This re-entrained oil may have been dispersed through the upper 20 meters (65 feet) of the water column or may have resurfaced later (e.g., if winds decreased; Section 4.2.5.4).

    - Approximately 250,000 barrels of DWH surface oil were collected at the sea surface and intentionally burned in 411 separate in situ burn events (Mabile & Allen 2010). The byproducts of burning included both soot particles that entered the atmosphere (some likely settled back to the sea and sank through the water column) and unburned oil residue that, because of its increased density, sunk and settled on the sea floor (Section 4.2.3.4).

- Within the surface waters, oil droplets attached to particulate matter (e.g., oil-degrading bacteria that proliferated in the near-surface waters in response to the oil's presence, phytoplankton, fecal pellets produced by zooplankton, etc.) and sank through the entire water column to the sea floor as another source of marine oil snow. This created a surface-derived contribution to the "dirty blizzard" of oily particulates falling to the sea floor, especially in areas where the deep-sea plume(s) of entrained oil particles did not reach (e.g., along the continental shelf; Section 4.2.3.3.2).

- Much of the surface oil remained as surface oil slicks that were transported across a huge swath of the northern Gulf of Mexico, cumulatively covering 112,115 square kilometers (43,300 square miles)—about the size of the state of Virginia (Section 4.2.5.2). Some surface slicks eventually stranded on northern Gulf of Mexico shorelines, including beach, marsh, mangrove, and other habitats (Section 4.2.6.2). Some oil that reached shorelines formed submerged oil mats (SOMs) in the subtidal areas, which were sometimes re-entrained and deposited on shorelines during storms (Section 4.2.6.2.1).

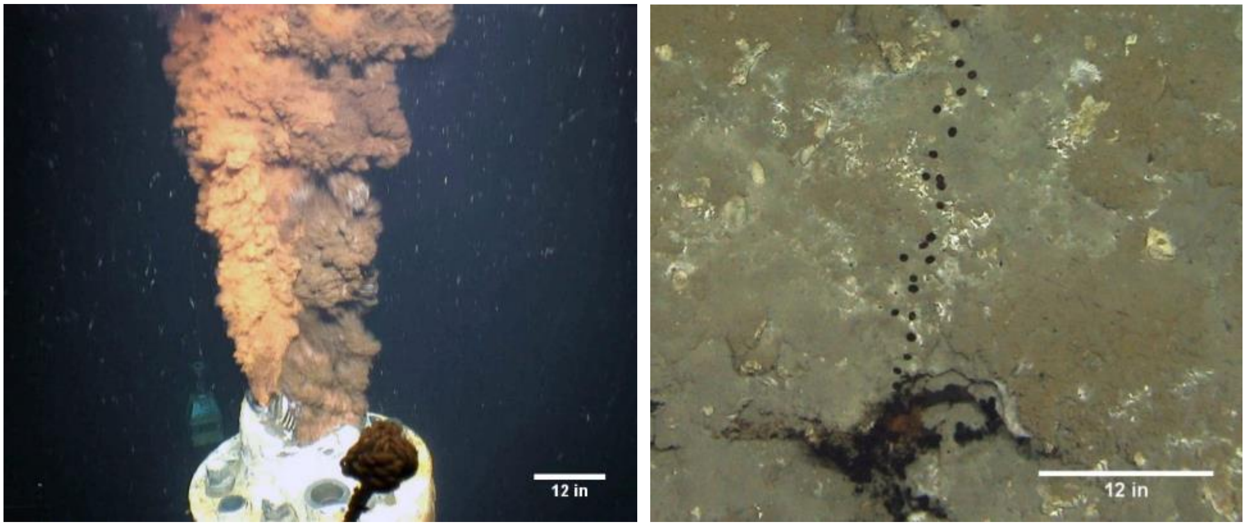
Despite the variable fates of the expelled oil (sunken, entrained, or surfaced) and the varying changes in its composition due to weathering and/or mixing with "background" chemicals in these environments, the DWH oil was still identifiable in various media using chemical fingerprinting and other lines of evidence. This helped the Trustees map the spatial extent of resources exposed to the weathered DWH oil following the spill and then monitor the oil's persistence over time.

4.2.2.2 Comparison to Natural Seepage Rate

The Gulf of Mexico is known for its prolific oil production and for the many natural petroleum seeps that occur in areas of petroleum accumulation. Oil from these sources contributes to the region's "background" chemicals, but the magnitude and effects of these other oil sources on the region's resources are very different from the acute effects of the DWH oil spill. For example, the DWH oil spill was not at all similar to natural seepage in terms of the rate of oil entering the environment. The total amount of natural oil seepage per year, from thousands of natural seeps over the entire 600,000 square miles (1.6 million square kilometers) of the Gulf of Mexico, is estimated to be between 220,000 and 550,000 barrels (9 million to 23 million gallons)(MacDonald 2012). This volume of oil slowly enters the deep sea from thousands of locations over a huge area annually. In contrast, the DWH spill released about six to 15 times the volume of oil from a single location in just 87 days. Exposure of resources to the quantity and extent of oil from natural oil seeps is simply not comparable to what was catastrophically released from the failed Macondo well in 2010 (Figure 4.2-5).

4.2.2.3 Dispersants

Dispersants are chemical mixtures that reduce the surface tension between oil and water, leading to the formation of oil droplets that more readily disperse in the water column (NRC 2005). Generally, dispersants contain surfactants (similar to dishwashing detergent) and solvents that together promote the formation of small oil droplets when added to oil and water.



Source: Oceaneering International (left); I.R. MacDonald, Florida State University (right).

Figure 4.2-4. Oil and gas released from the Macondo well after the riser pipe was cut (left) compared to an active natural seep in Green Canyon (right). Scales are approximate.

Dispersants are sometimes used in oil spill response as a means to break oil slicks into small droplets that can then become entrained in the water column. This process reduces the amount of floating oil available to reach shorelines, but increases the amount of small oil droplets to which underwater biota may be exposed.

Between April 22 and July 19, 2010, approximately 1.84 million gallons of two different dispersants were used during the DWH incident (BP 2014b; USCG 2011). The two dispersants were Corexit 9500A and Corexit 9527. In total, boats and planes applied about 1.07 million gallons of these two dispersants to surface waters (Figure 4.2-4). An additional 770,000 gallons of Corexit 9500A (only) were injected in the deep sea directly at the wellhead (USCG 2011).

Source: Houma (2010).

Figure 4.2-5. Aerial application of dispersant.

The Trustees and other researchers conducted chemical analyses of the two dispersants to determine the likelihood that chemicals within the dispersants would persist in the environment after being applied to the oil. This research focused on two solvents, di(propyleneglycol)-n-butyl ether and 2-butoxyethanol, and a surfactant, dioctyl sodium sulfosuccinate (DOSS), found in the dispersants (Stout 2015i).

Researchers found that DOSS, in particular, persists in the environment. More than 2 months after dispersants were last injected at the wellhead, DOSS was detectable in the deep-sea plume up to 300 kilometers (185 miles) away from the well (Kujawinski et al. 2011). DOSS was also detected on deep-sea corals 6 months after the spill, and traces were still found on northern Gulf of Mexico beaches up to 3 years after the spill (White et al. 2014). These results are consistent with the Natural Resource Damage Assessment (NRDA) sampling data, which include detectable concentrations of DOSS and other dispersant-derived chemicals in the deep-sea plumes, floating oil slicks, and oil stranded on shorelines (Payne & Driskell 2015a, 2015b; Stout 2015g, 2015h).

The surface application of dispersants increased exposure of near-surface biota to oil that re-entered the water column (Sections 4.2.5.3 and 4.2.5.4). The subsea application of dispersants at the wellhead helped keep some oil in the deep sea where it was entrained within the deep-sea plume (Section 4.2.3.2).

Although DWH incident response activities may not be the only potential source of DOSS in the nearshore environment, data suggest that DOSS and other chemicals from the DWH dispersant applications conducted offshore likely persisted in the environment and were transported to shorelines (White et al. 2014). Thus, the overall fate of dispersant-derived chemicals was similar to the fate of the DWH oil: dispersant chemicals applied at the wellhead either deposited on the sea floor or became entrained within deep-sea plumes, and dispersant chemicals applied at the sea surface were transported throughout the northern Gulf of Mexico with surface oil slicks.

4.2.2.4 Synthetic-Based Drilling Mud

Conventional synthetic-based drilling mud was used in the original drilling of the Macondo well prior to the DWH incident. In addition, BP used a similar “kill” mud in response to the spill. Specifically, between May 26 and May 29, 2010, nearly 30,000 barrels of kill mud, along with various bridging materials (e.g., golf balls, cubes, and miscellaneous objects), were pumped into Macondo in a failed effort to plug the well.

Both these muds, herein collectively referred to as synthetic-based drilling mud, contained synthetic chemicals (olefins) along with barium sulfate, the latter of which comprised up to 60 percent of the mud by weight (Stout 2015f). These high levels of barium sulfate make the muds dense (heavy). The drilling mud also contained traces of PAHs and petroleum-based chemicals, such as ethylene glycol.

Synthetic chemicals are designed to resist breaking down when under high temperature and pressure while drilling a well; therefore, they are similarly resistant to breaking down on the sea floor. During the blowout and failed well plugging attempt, an unknown volume of synthetic-based drilling mud was discharged from the well. As indicated by the detection of synthetic chemicals in deep-sea sediments, this mud was determined to have spread over the sea floor within 4 square kilometers (2.5 square miles) of the wellhead, sometimes up to at least 10 centimeters thick, and smothered the benthic habitat. The synthetic-based mud was still found in this area 4 years after the spill (Stout 2015f). Section 4.5, Benthic Resources, presents additional information on the significance of these persistent synthetic-based muds on the sea floor.

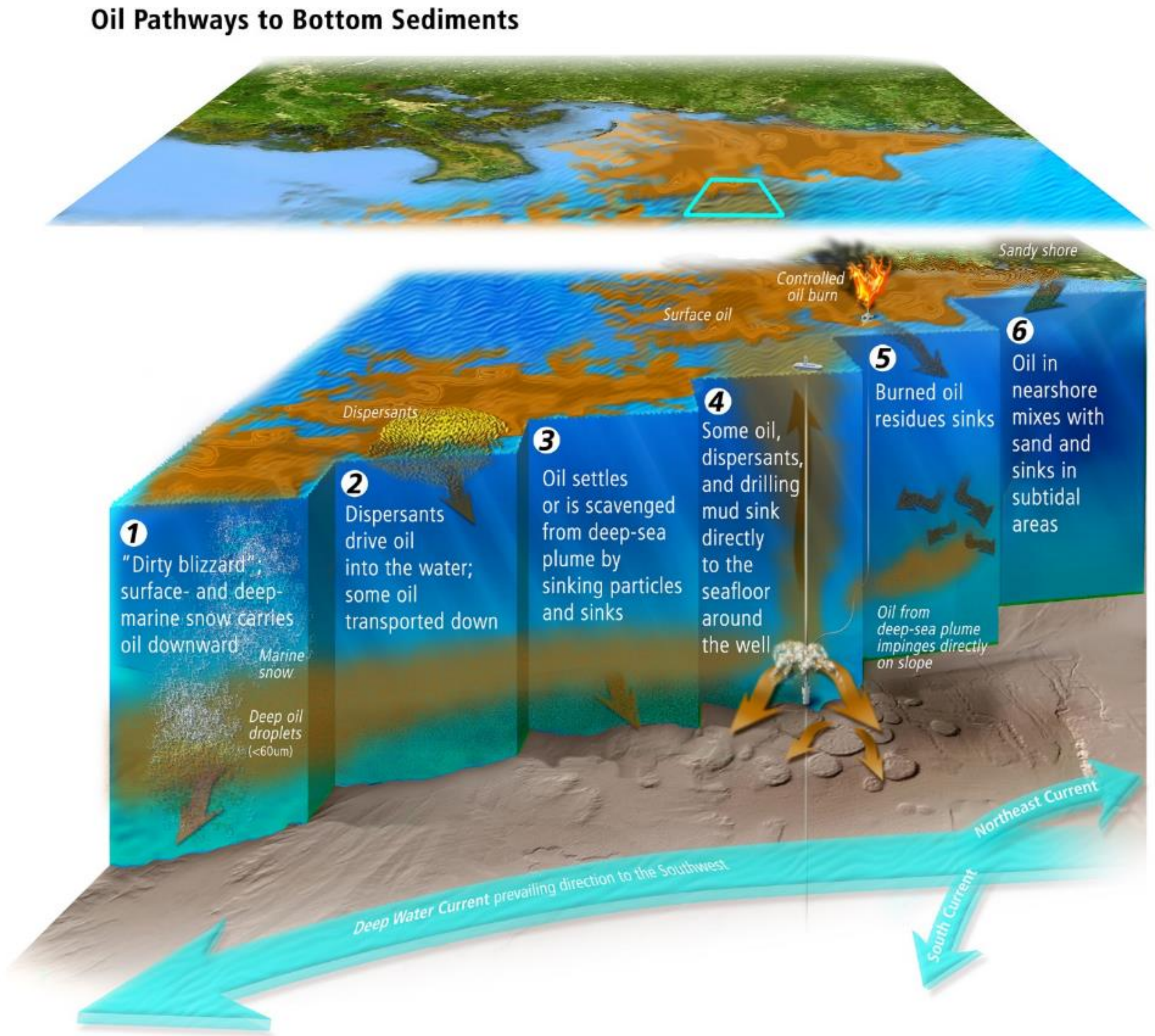
4.2.3 Exposure in the Deep Sea and Sea Floor

Key Points

- Oil and dispersant-derived chemicals from subsea injection remained in the deep sea and were transported laterally within a deep-sea plume that extended more than 250 miles (400 kilometers) southwest of the well and persisted for at least 5 months after the spill ended. Some evidence indicates vestiges of the plume persisted for nearly 1 year after the spill ended.
- Oil was deposited on the sea floor by various mechanisms, including:
  - Direct fallout around the well (which also deposited synthetic-based drilling mud).
  - Deposition from or impingement of the migrating deep-sea plume.
  - Marine snow-facilitated downward transport of surface oil and oil within the deep-sea plume to the sea floor.
  - Sinking of in situ burn residues.
- Exposure to oil from natural oil seeps is restricted to seep areas, where exposure is highly localized and generally distinguishable from DWH oil exposures.
- According to empirical chemical data for sea floor sediments and floc collected in 2010 and 2011, the "footprint" of oil on the sea floor that is clearly derived from DWH oil covers at least 400 to 700 square miles (1,030 to 1,810 square kilometers) of the deep-sea floor, but other evidence indicates impacts occurred over an even larger area of the deep sea and continental shelf.
- Empirical chemical data for sediments collected in 2014, 4 years after the spill, show the sea floor still contains DWH oil (including PAHs) but over a smaller "footprint" of 180 to 220 square miles (466 to 570 square kilometers). These data also show that DWH oil concentrations in sediments have generally decreased since 2010 and 2011.
- Red crabs, coral, and other biota living on the sea floor were exposed to DWH oil that settled on the sea floor.

During and for months following the DWH incident, the deep sea and sea floor resources of the northern Gulf of Mexico were exposed to oil, deep-sea injected dispersants, and synthetic-based drilling mud. These substances had either 1) remained and moved within the deep sea or 2) moved to the sea surface and then back through the water column to the sea floor. In this section, the Trustees describe exposures experienced by resources in the deep-sea water column and the sea floor. (Shallower water column exposures are discussed elsewhere in Sections 4.2.4 and 4.2.5.) Deep-sea and sea floor findings are based on subsea ROV video, photographic observations, various empirical (physical and chemical) measurements, and modeling for all subsurface regions of the northern Gulf of Mexico.

Figure 4.2-6 depicts the processes by which resources in deep-sea water and benthic sediments were exposed to DWH oil and other contaminants. The following sections describe each process in greater detail.



Source: Kate Sweeney for NOAA.

Figure 4.2-6. Depiction of processes by which DWH-related contaminants exposed resources within the deep-sea pelagic water and sea floor.

4.2.3.1 Exposure and Chemistry on the Sea Floor Near the Well

Direct fallout from the original blowout and subsequent attempts to plug the well led to the direct deposition of crude oil onto the sea floor proximal to the Macondo well (see panel 4 in Figure 4.2-6). Much of the oil deposited near the well was "sediment entrained," meaning the oil was deposited in conjunction with dense synthetic-based drilling mud that facilitated its rapid sinking. Oil and varying amounts of synthetic-based drilling mud were found to have accumulated on the sea floor up to 10 centimeters thick (based on the deepest sediment cores taken) and to cover most sea floor sediments collected within a 6.5-square-kilometer (2.5-square-mile) area around the well (OSAT-1 2010; Stout 2015b, 2015f).

Most oil present in these sediments was only minimally weathered, likely owing to its short tenure within the water column and the high concentrations (and likely rate) at which it was deposited. Sediments near the well contained oil-derived TPAH50 at concentrations up to 410 micrograms per gram (µg/g).[1]

Biota living in this direct "fallout" zone were exposed to oil (and synthetic-based mud). Specifically, DWH oil was found in red crabs collected from this area, which are apex members of the deep-sea benthic food web. For example, crabs in this area had PAHs in their body tissue (hepatopancreas) at concentrations up to 3,700 nanograms per gram (ng/g)[2] (Douglas & Liu 2015; also Section 4.5, Benthic Resources).

4.2.3.2 Exposure and Chemistry Within the Deep-Sea Plume

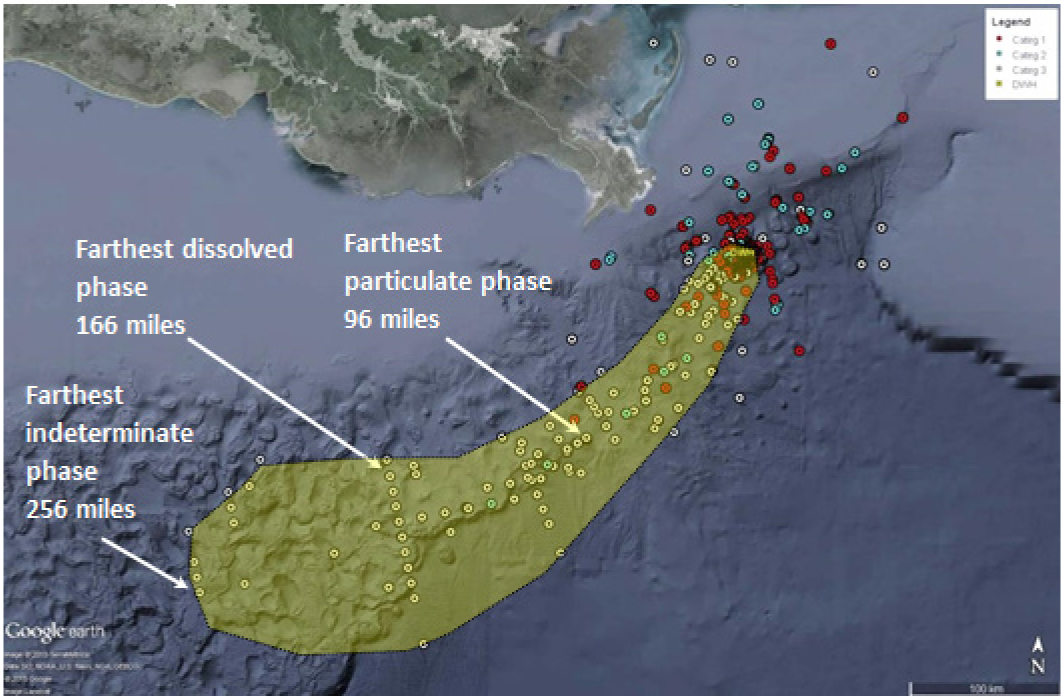
As indicated above, the powerful turbulent discharge of gas and oil from the Macondo wellhead and riser tube (Figure 4.2-5) continued non-stop for 87 days. Nearly all of the methane and other gases expelled and many other soluble hydrocarbons quickly dissolved into the deep-sea water (Reddy et al. 2012). In part aided by the injection of 770,000 gallons of chemical dispersant, the deep water discharge instantly produced very small oil droplets that had insufficient buoyancy to ascend to the surface (Li et al. 2015). Thus, the dissolved gases, any other dissolved chemicals from the oil and dispersant, and small neutrally buoyant oil droplets remained in the deep sea.

Owing to slow moving but constant deep-sea currents, the dissolved gases, other dissolved chemicals, and small oil droplets were transported laterally within a deep-sea plume of neutrally buoyant water, sometimes called the "intrusion" layer. This layer is found approximately 3,200 to 4,200 feet (1,000 to 1,300 meters) beneath the surface (Camilli et al. 2010; A. Diercks et al. 2010; Hazen et al. 2010; Ryerson et al. 2012). A proliferation of bacteria degraded gases and other dissolved chemicals within the plume—a process that also decreased dissolved oxygen levels within the deep-sea plume (Hazen et al. 2010; Joye et al. 2011a; Joye et al. 2011b; Kessler et al. 2011; Valentine et al. 2010). Oil within the plume, having a large surface area to volume ratio, was subjected to intense weathering that caused most low molecular weight and some intermediate molecular weight aromatic compounds to dissolve into the plume water. The residual oil droplets within the plume, consequently, were enriched in the high molecular weight aromatic compounds that did not dissolve.

Despite dilution and subsequent biodegradation of dissolved chemicals, the deep-sea plume of oil and dispersants could be tracked in multiple directions, but mostly toward the southwest (e.g., Spier et al. 2013). Efforts to "track" the plume throughout 2010 showed that it persisted for more than 400 kilometers (250 miles) from the well along the continental slope toward the southwest (Figure 4.2-7). The deep-sea plume persisted during the active spill and could still be detected 5 months after the spill

[1] All concentration values for sediments or other solids are presented on a dry weight basis.
[2] All concentration values for tissues are presented on a wet weight basis.

in December 2010 (Payne & Driskell 2015d). However, investigations (March 2011) could no longer detect the plume 3 months later.



Source: *Payne and Driskell (2015d); yellow plume added for clarity.*

Figure 4.2-7. Spatial extent of deep-sea plume determined through forensic analysis of water samples collected in 2010. Colors depict different forensic match categories reported by Payne and Driskell (2015d), but all samples plotted were impacted by DWH incident.

Concentrations of oil-derived chemicals within the plume were highest nearer the well (where both dissolved and particulate oil was present) and generally decreased with increasing distance from the well. Particulate oil was still present within the plume more than 96 miles (155 kilometers) from the well toward the southwest (less in other directions), and dissolved oil was detected up to 166 miles (267 kilometers) from the well (Figure 4.2-7) (Payne & Driskell 2015d). As noted above, other measured indicators of the deep-sea plume indicated that the plume extended almost 256 miles (412 kilometers) southwest of the well (Figure 4.2-7) (Payne & Driskell 2015e). Those indicators included the presence of dispersant-derived chemicals, fluorescence, and decreased dissolved oxygen concentrations.

Within the deep-sea plume, water sampling data demonstrate that deep water column and deep benthic organisms were exposed to the dissolved and particulate oil and dispersant-derived chemicals. Concentrations of dissolved BTEX exceeding 100 µg/L (micrograms per liter, or parts per billion [ppb]) were measured in numerous deep-sea plume water samples. NRDA water samples showed highly variable TPAH50 concentrations, with the highest being 67.8 µg/L (ppb) (Payne & Driskell 2015e). Other researchers documented TPAH50 concentrations in the deep plume as high as 189 µg/L (ppb) (A.R. Diercks et al. 2010).

Notably, hydrocarbons in sea floor surface sediments were found at somewhat higher concentrations along the continental slope north of the Macondo well (see Figure 4.2-9 and discussion below). The

Trustees recognized this as evidence that the small oil droplets within the deep-sea plume (at 3,200 to 4,200 feet beneath the surface) that moved along the continental slope directly impinged upon the continental slope and created a "bathtub-ring" of oil in this area (Stout et al. 2015; Valentine et al. 2014). Thus, in addition to oil droplets being agglomerated with marine snow and carried to the sea floor (Section 4.2.3.3.1), a second mechanism by which oil impacted the deep-sea sediments resulted from the direct intersection of the deep-sea plume with the deep-sea floor.

4.2.3.3 Exposure to Marine Oil Snow Deposited on Sea Floor

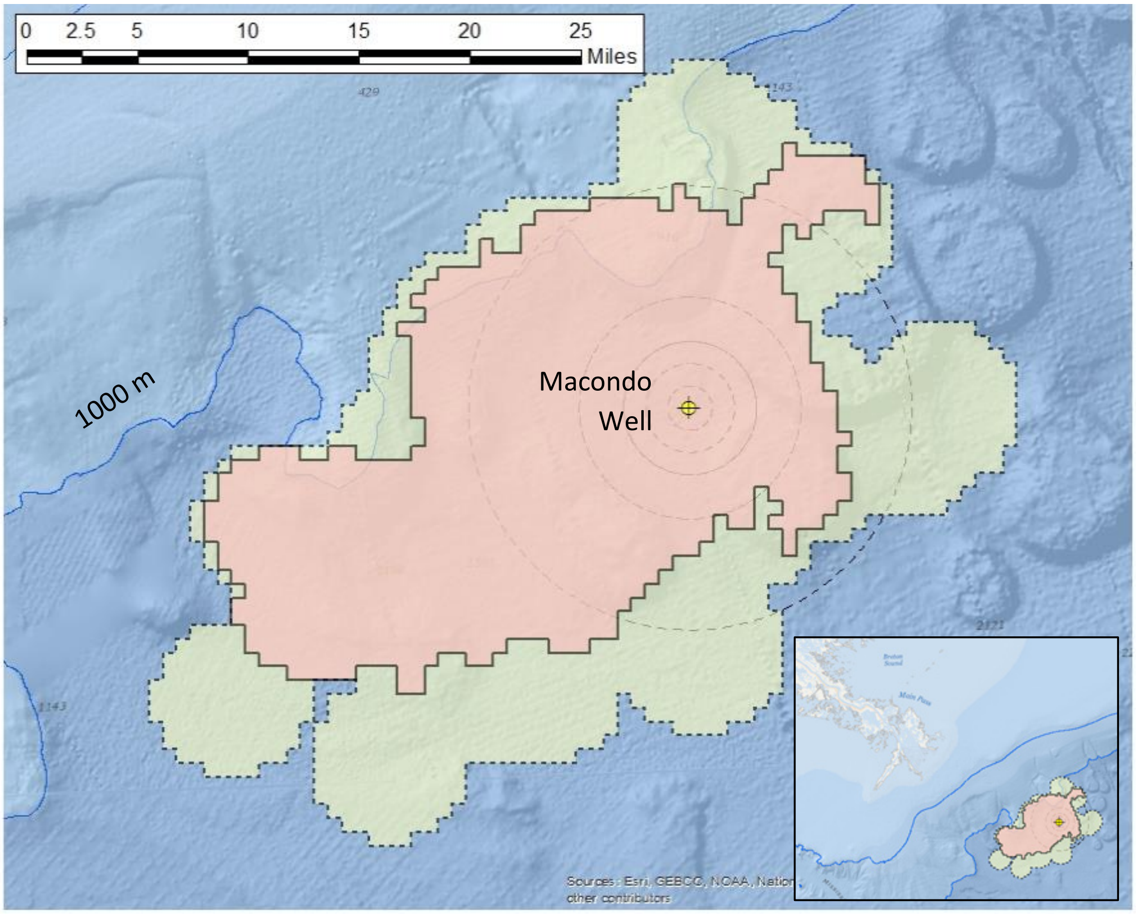
Studies conducted during the response effort following the spill revealed the presence of layers of brown flocculent material, or "floc," on the deep-sea floor in areas beyond the 2.5-square-mile direct fallout area near the well (OSAT-1 2010). The floc was absent in previous studies of sea floor sediments (Joye et al. 2014). The floc was determined to largely consist of bacterial biomass that settled to the sea floor following the massive die-off of bacteria bloom stimulated by oil and gas in both the sea surface and deep-sea plume (Joye et al. 2014; Passow 2014; Passow et al. 2012). The mucus-rich microbial biomass formed aggregates with dispersed oil particles in both the shallow and deep water. These aggregates—referred to as microbially mediated "marine oil snow" (Kinner et al. 2014)—sank to the sea floor during the so-called "dirty blizzard" (Schrope 2013) (see panels 1 and 3 in Figure 4.2-6).

4.2.3.3.1 Marine Oil Snow in the Deep Sea: Depths Greater Than 1,000 Meters

Chemical evidence for the marine oil snow deposition is indicated by the elevated concentrations of hydrocarbons (e.g., hopane; Valentine et al. 2014) and by forensic analysis of hundreds of sediment cores from the deep sea (Stout 2015b). These sediment-based studies show that the impacted areas are largely restricted to the sea floor surface in locations more than 1,000 meters deep and centered around the failed Macondo well, but also skewed toward the southwest (Stout 2015b; Stout et al. 2015) (Figure 4.2-8).

The "footprint" of the deep-sea sediments containing DWH oil covered between approximately 1,030 to more than 1,810 square kilometers (400 to 700 square miles; Figure 4.2-8) . This "footprint" is based upon the ability to chemically fingerprint the DWH oil in surface sediments. As such, this area (400 to 700 square miles) is smaller than the 1,200 square miles reported by Valentine et al. (2014), which was less conservative and based only on the presence of elevated "excess" hydrocarbon (hopane) above background concentrations. Thus, exposures to lower concentrations of hydrocarbons derived from DWH oil likely occurred on the sea floor outside the 1,030- to 1,810-square-kilometer "footprints" containing "fingerprintable" DWH oil (Figure 4.2-8).

The footprints of sea floor sediment recognized to have been impacted by oil or oily floc (Figure 4.2-8) are centered around the Macondo well and distributed in a manner consistent with oil spreading out away from the well. The footprint shape and concentrations are unrelated to the locations of natural seeps in the region; in fact, in some active seep locations, oily floc from the DWH incident could be seen to "blanket" the sea floor sediments already impacted with seeped oil (Stout 2015b). In addition, through a combination chemical fingerprinting and vertical and lateral concentration trends in sediments, the oily floc on the sea floor surface throughout the footprint could be confidently attributed to DWH oil, and not natural seeps (Stout 2015b). The depth (greater than 1,000 meters) and shape of the "footprint" (Figure 4.2-8) indicated that the marine snow that deposited the oily layer was derived

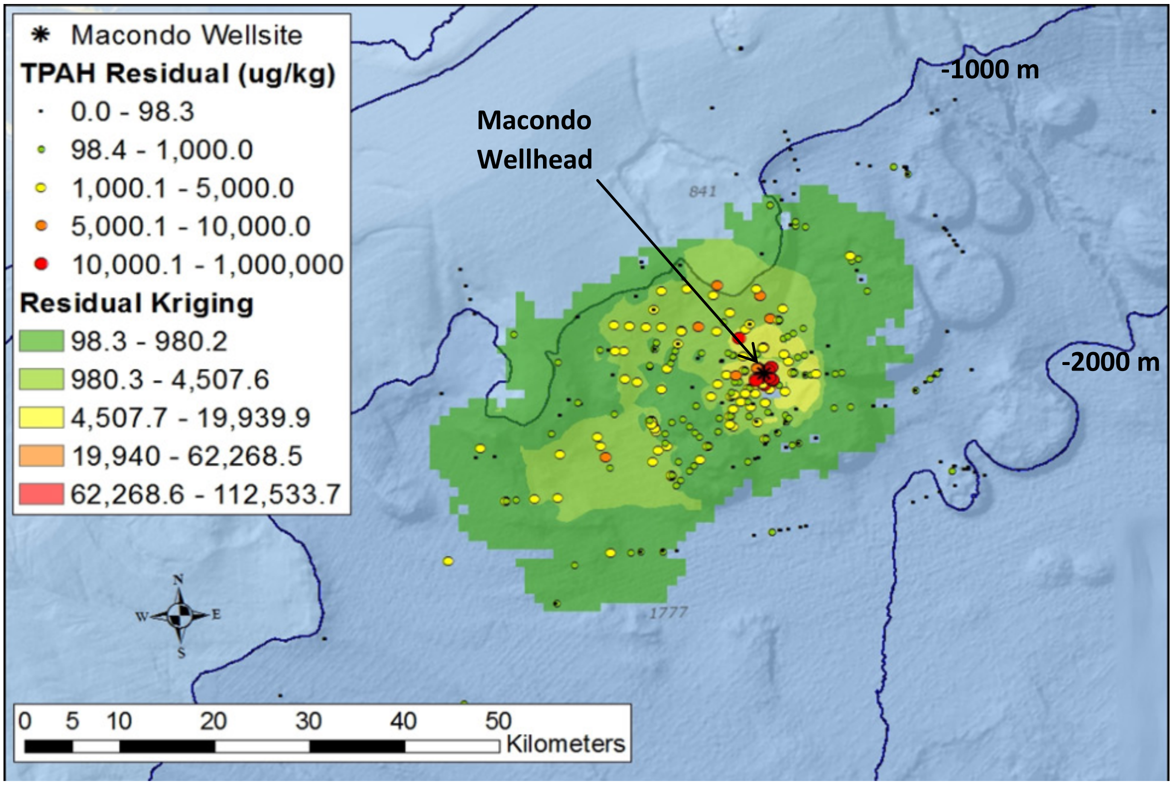


Source: *Stout et al. (2015).*

Figure 4.2-8. Map showing minimal (pink; 1,030 square kilometers [400 square miles]) and maximal (green; 1,810 square kilometers [700 square miles]) "footprints" of Macondo oil recognized through forensic analysis of deep-sea surface sediments. Inset shows location relative to the Mississippi River Delta.

predominantly from the deep-sea plume (Stout 2015a; Valentine et al. 2014), which had only existed at depths below 1,000 meters and spread preferentially toward the southwest (Section 4.2.3.2).

The oil within sea floor floc was enriched in high molecular weight PAHs, owing to the significant dissolution and biodegradation the oil experienced during its transport as small droplets within the deep-sea plume prior to deposition. Concentrations of total PAHs (defined as TPAH50) attributable to DWH oil were highest near the well (Figure 4.2-9), reaching a maximum of 410,000 micrograms per kilogram (µg/kg) (ppb) due to direct fallout of oil (Figure 4.2-9). Concentrations of TPAH50 generally decreased with increasing distance from the well, a pattern consistent with a single, localized source (i.e., the Macondo well), not seeps. Sea floor deposition patterns exhibited some "patchiness," likely owing to sea floor topography and redistribution of oily floc by bottom currents, which may have caused preferential accumulation in localized bathymetric lows. Residual TPAH50 concentrations were somewhat elevated along the continental slope north of the well due to the "bathtub ring" effect noted above, wherein the oil from the plume directly impinged on the sea floor in this area (Figure 4.2-9).



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

Figure 4.2-9. Map showing the concentration of TPAH50 attributable to DWH oil in deep-sea surface sediment (0–1 centimeter). Concentrations were highest near the well and extend both north and west along the continental slope and southwest in the direction of the deep-sea plume. PAH attributable to natural seeps are excluded following forensic analysis. "TPAH Residual" refers to contamination levels, indicating DWH oil above background or seep concentrations.

Exposure of benthic ecosystems below 1,000 meters to oily floc containing DWH oil was also confirmed by visible, biological, or chemical evidence at numerous deep water coral communities (Fisher et al. 2014; Hsing et al. 2013; White et al. 2012) and in soft-bottom benthic infauna (Montagna et al. 2013). Indeed, the oily floc found coating corals from a deep water community near seeps was chemically consistent with the oily floc found widespread on the sea floor farther removed from seeps (Stout 2015b) and provides direct evidence of exposure to DWH oil. Red crabs and other benthic macrofauna (e.g., sea cucumbers) up to 14 kilometers from the well were also found to contain Macondo-derived hydrocarbons (Douglas & Liu 2015).

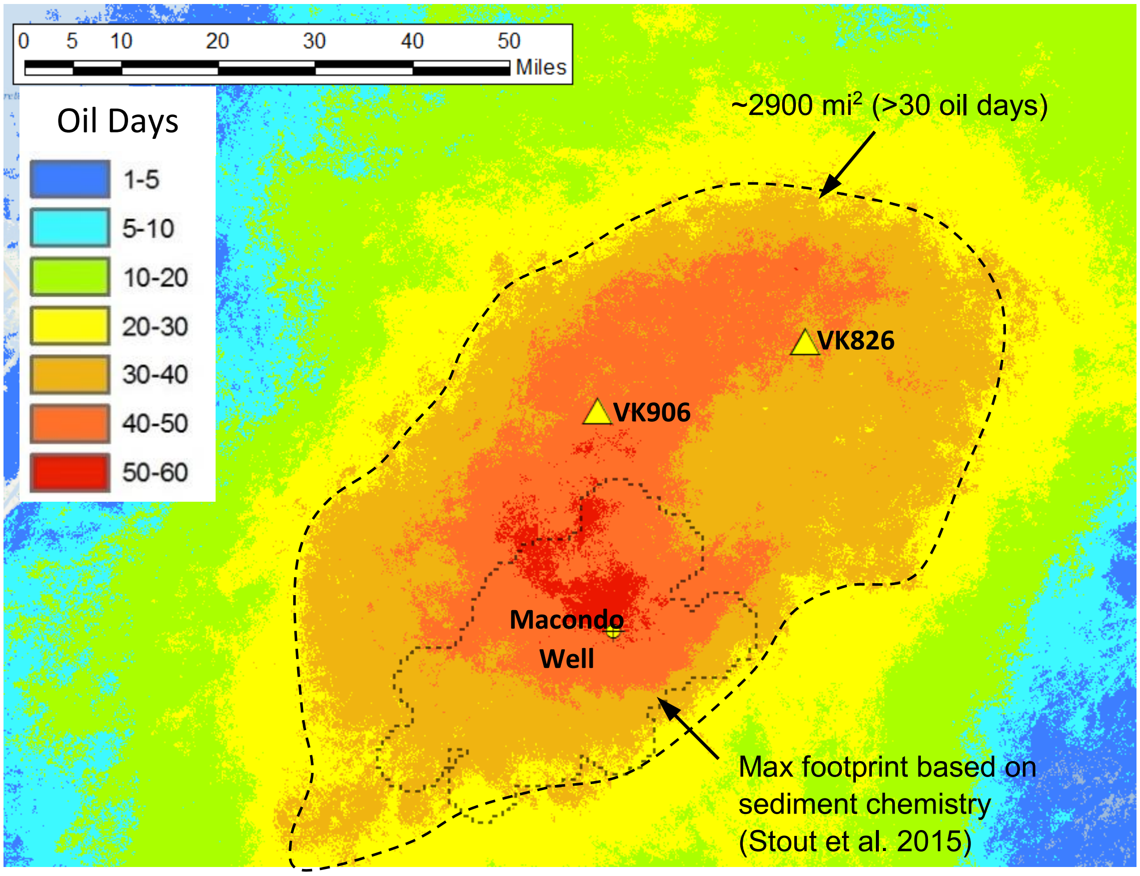
Notably, sediment cores collected early in the course of the incident during the response (and outside of the 2.5-square-mile area proximal to the well) did not indicate the presence of sea floor oil/floc beyond about 1 mile from the well—an observation that seemingly contradicts the finding presented above. This is attributed to the fact that response cores were not collected with the intention to retain any oily floc at the surface (which required great care) (Payne & Driskell 2015e). Additionally, if floc was collected, it was diluted when the entire top 3 centimeters of each sediment core were analyzed. Thus, the sediment cores collected during the NRDA were able to reveal much more detail than the response cores.

The deposition of oily floc predominantly occurred during and shortly after the active spill. Flux calculations based on oily particles collected in a deep-sea sediment trap showed 97 percent of the oil found on the sea floor was deposited before late August 2010 (i.e., within about 5 weeks after the spill ended) (Stout & Passow 2015). However, this same study found that trace amounts of DWH oil or oily particles "lingered" within the water column until August 2011, approximately 1 year after the spill had stopped.

As noted above, some evidence suggests that the impact to the sea floor may extend beyond the 400- to 700-square-mile "footprint" recognized through hydrocarbon fingerprinting of sea floor sediments (Figure 4.2-8). For example, some impacts to deep water corals beyond the recognized footprint have been reported (Fisher et al. 2014) and other data suggest impacts extended east in the DeSoto Canyon area (Brooks et al. 2015; Chanton et al. 2015). The Trustees collected data to support the contention that benthic exposures occurred outside the "footprints" that could be recognized through sediment chemistry. The following section reviews that evidence.

4.2.3.3.2 Marine Oil Snow on the Slope and Continental Shelf: Depths Less Than 1,000 Meters

Evidence for the deposition of marine oil snow and the resulting exposure of shallower benthic ecosystems (depths less than 1,000 meters) was found through chemical analysis of semi-permeable membrane devices (SPMDs) and sediment traps deployed within the water column. For example, SPMDs deployed near a shelf edge (Alabama Alps) mesophotic reef site north of the well also collected DWH oil within the water column (Stout & Litman 2015a). In addition, sediment traps captured marine oil snow settling through the water column along the shelf edge 37 miles northeast of the well (VK826; Figure 4.2-10) and proximal to *Lophelia* reef ecosystems (Stout & German 2015). Data showed approximately 26 barrels of oil per square mile were deposited on the sea floor in this area during the spill. Based on this result, marine oil snow deposition likely affected wide portions of the continental slope and shelf wherever floating surface oil occurred. Based on the sediment trap results, Stout and German (2015) estimate that more than 76,000 barrels of oil once present at or near the ocean surface sank within an approximately 2,900-square-mile area, which is much larger than the deep-sea sediment footprint (Figure 4.2-10).



Source: *Stout and German (2015); Stout et al. (2015).*

Figure 4.2-10. Map showing area where surface oil was present for 30 or more days (April 25 to July 28, 2010). This area spanned approximately 2,900 square miles, which is much larger than the footprint based on sediment chemistry (from Figure 4.2-8). The sediment trap results collected during the active spill at VK826 indicated marine oil snow was prolific and depositing at a rate of 26 barrels/square mile. Comparable oily marine snow deposition is assumed to have occurred within the entire 2,900-square-mile area during the active spill (Stout & German 2015). The legend in the figure refers to "cumulative oil days" (i.e., the total number of days when surface oil was determined to be present using remote sensing between April 25 and July 28, 2010).

4.2.3.4 Exposure from Sinking In Situ Burn Residue

In situ burning, whereby oil floating on the ocean surface was corralled and ignited to reduce the amount of oil that could continue to impact resources and eventually strand on shorelines, was a widely used countermeasure in response to the DWH oil spill (Section 2, Description of the Incident). Approximately 250,000 barrels of floating DWH oil reportedly were consumed during 411 separate burn events (Mabile & Allen 2010). The burning produced massive atmospheric emissions (Perring et al. 2011; Ryerson et al. 2011) and between 11,600 and 16,300 barrels of "stiff, taffy-like" burn residue that subsequently sank (see panel 5 in Figure 4.2-6).

Samples of burn residue were collected from the sea surface and sea floor, before and after it sank, respectively. The Trustees determined these samples were enriched in high molecular weight PAHs compared to unburned oil (Stout & Payne 2015). In addition, high molecular weight PAH-rich particles were collected in deep-sea sediment trap samples from late August 2010, which was 4 to 5 weeks after the last in situ burn (Stout & Passow 2015). Detection of burn-related PAH in these sediment trap samples suggests that atmospheric particles (soot) re-deposited to the Gulf surface (as can be seen in Figure 4.2-11) and subsequently sank. Thus, both residues of the unburned surface oil and soot particles generated during in situ burning sunk to the sea floor and further exposed benthic resources to this additional source of DWH oil-derived contaminants.

Source: *J.R. Payne aboard Jack Fitz III cruise.*

Figure 4.2-11. Photographs showing multiple in situ burn events (June 18, 2010). Note soot from plumes is settling back toward sea surface.

4.2.4 Exposure Within the Rising Plume

Key Points

- The majority (65 to 75 percent) of oil volume released from the Macondo well rose nearly 5,000 feet (1,500 meters) through the water column, eventually reaching the sea surface.
- Oil that remained in the deep plume consisted of very small droplets lacking the buoyancy to rise above this zone. In contrast, oil within the rising plume consisted of larger droplets with sufficient buoyancy to rise.
- As the oil droplets ascended, some of their lighter, more soluble hydrocarbons dissolved into the surrounding waters forming a rising plume of dissolved chemicals and droplets of partially weathered oil.
- Throughout the 87-day spill, water column biota between the wellhead on the sea floor and the ocean surface were exposed to high concentrations of dissolved and particulate oil within the rising plume.

Biota in the nearly 1,500-meter (5,000-foot) water column above the failed Macondo well were exposed to rising particles of (buoyant) oil and dissolved chemicals from the oil within a large vertical plume that persisted throughout the entire duration of the active DWH incident (see Figure 4.2-12 and Section 4.4, Water Column).

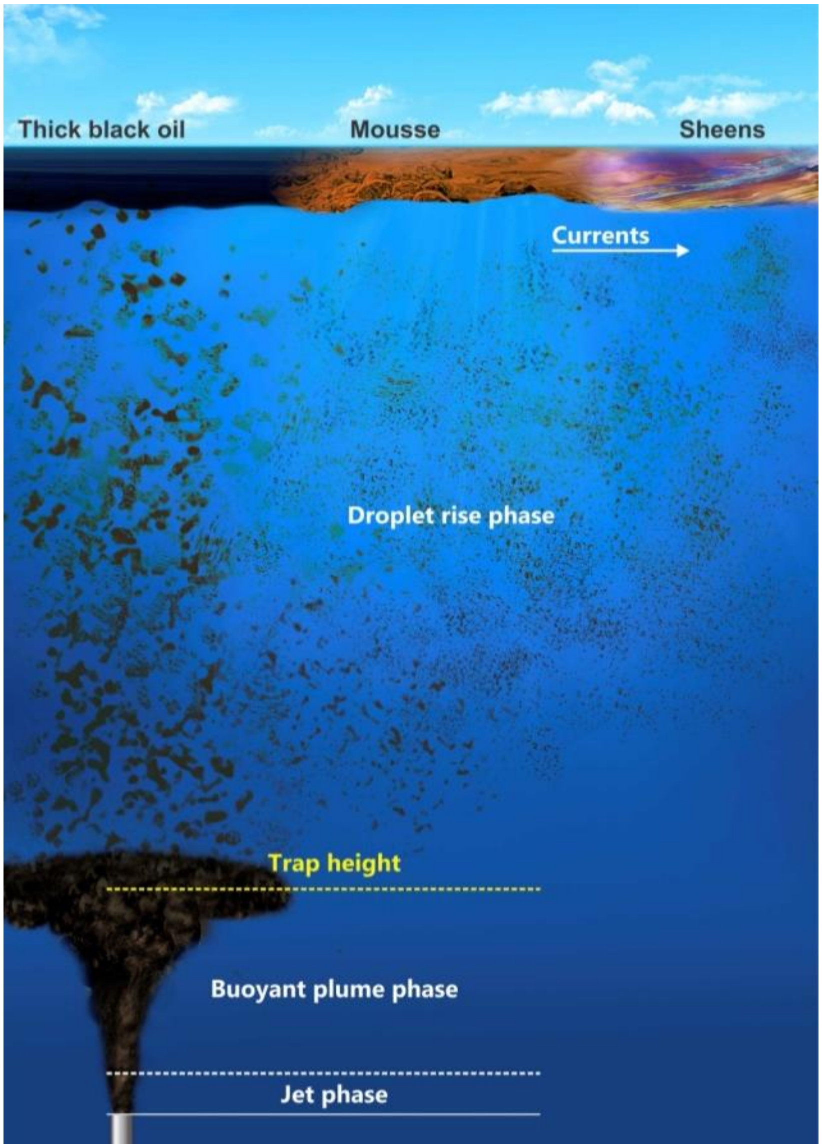
Oil released from the well rose quickly to a depth where it became neutrally buoyant, known as the "trap height" (Figure 4.2-12). Very small droplets of oil, formed by the combined effect of high energy, turbulent expulsion from the well, and the injection of dispersant at the wellhead, did not have sufficient buoyancy to rise farther and remained effectively trapped within the water at a depth between about 3,200 and 4,200 feet, (975 and 1,280 meters) thus creating a deep-sea plume (described in Section 4.2.3.2). However, approximately 65 to 75 percent of the oil released from the Macondo well consisted of larger droplets that had a sufficient buoyancy to continue through the "trap height" and rise another 3,200 feet (975 meters) through the water column to the sea surface (Li et al. 2015).

The rise of these larger droplets was mostly vertical, but some lateral spreading of the rising (buoyant) plume occurred (Spaulding et al. 2015) (Figure 4.2-12). The largest droplets within the rising plume surfaced within a few hours (French McCay et al. 2015), mostly within about 1 mile of the wellhead (Ryerson et al. 2011). However, some oil droplets were laterally spread during ascent and surfaced more slowly; some of these droplets surfaced at locations beyond a 2-square-kilometer area centered over the wellhead.

4.2.4.1 Exposure of Biota to Rising Oil

The rising oil exposed the entire 1,500-meter (5,000-foot) water column above the well to oil droplets as they spread laterally, aided by subsea currents, during their ascent. For two main reasons it was not possible to representatively sample the immense volume of water impacted by the rising plume during the 87-day release timeframe. First, it was difficult to capture the buoyant oil droplets in discrete "grab" water samples; second, researchers were excluded from conducting sampling near the wellhead during ongoing response activities. Thus, only a limited number of water samples empirically authenticated the existence of the rising plume.

To fill this gap in the sampling data, the Trustees used numerical models to estimate hydrocarbon concentrations within the rising plume. The models were based on scientific principles of the oil's behavior in water (Section 4.4, Water Column). The modeling results coupled with the limited authentic field sample data characterized the temporal and spatial variations in oil exposures for water column resources.



Source: French McCay et al. (2015).

Figure 4.2-12. Schematic drawing depicting the ascent of oil from the Macondo well with neutrally buoyant oil stopping around the "trap height" and buoyant oil ascending toward the surface, with larger particles rising more quickly and mostly vertically and smaller particles rising more slowly and spreading laterally by subsea currents.

Numerical modeling of the Macondo well blowout estimated the droplet sizes present at the trap height (Spaulding et al. 2015). Oil droplet mass, size, and location estimates from this analysis were used as input to the Spill Impact Modeling Application Package (SIMAP) oil fate model (French McCay 2003, 2004). SIMAP then simulated weathering (i.e., dissolution and biodegradation) and movements, as well as concentrations of oil and its individual or groups of constituents (e.g., PAHs), in the water between the trap height and the ocean surface (French McCay et al. 2015). The model provided an estimated range of oil concentrations in water within the rising plume at any given time and depth. The Trustees estimated the volume of water affected by both the larger, rapidly rising oil droplets and the smaller, more slowly rising oil droplets (that were spread more laterally; Figure 4.2-12).

Animals living within or passing through the rising plume were exposed to the concentrations of hydrocarbons from the liquid droplets of oil and its dissolved constituents, both of which were derived from the SIMAP analyses. These exposures occurred from the first day of the blowout on April 20, 2010, until the well was capped on July 15, 2010. In addition, some of the more slowly rising droplets likely persisted in the water column for some time after the well was capped.

4.2.4.2 Chemistry of Rising Oil and Changes in Oil During Transport

As noted above, sampling vessels were typically excluded from a zone around the Macondo well to avoid interfering with response assets or compromising safety. Therefore, large numbers of samples within the rising plume could not be collected as part of the NRDA.

Despite this constraint, response and NRDA personnel on several offshore cruises were able to collect 47 water samples in the rising plume between May and August 2010. These samples were collected within 3 miles (4.8 kilometers) of the well and at depths ranging from 130 to 3,200 feet (40 to 975 meters) (i.e., below the upper mixed zone and above the deep plume). Forensic analysis identified the samples as containing DWH oil, and the maximum TPAH50 concentration observed among these samples was 19 µg/L (Payne & Driskell 2015a). This measured concentration is about 10 times lower than 218 µg/L—the maximum TPAH50 concentration predicted by the model within the rising plume (Section 4.4, Water Column) (French McCay et al. 2015). However, this disparity is minor considering only 47 samples could be collected. This again emphasizes the importance of the model results on representing the true conditions within the rising plume.

As oil droplets within the rising plume ascended, lighter aromatic hydrocarbons and other relatively soluble hydrocarbons dissolved into the surrounding water column, effectively leaving behind a “cone” of dissolved chemicals ascending and spreading above the well. Transport of oil from the wellhead or broken riser pipe to the surface was nearly vertical, with most of the rising oil ultimately surfacing within about 1 mile (1.6 kilometers) of the wellhead (Ryerson et al. 2011).

Weathering of DWH oil occurred as the oil traveled to the surface through the water column. Chemical analysis of oil collected immediately upon surfacing showed the liquid oil had already lost approximately 15 percent of its original mass, at least some portion of which was through dissolution (Stout 2015h). Evaporation of the surfacing oil was rapid and aerial measurements identified massive amounts of spill-related hydrocarbons evaporating to the atmosphere over an area of approximately 0.75 square miles (1.9 square kilometers) around the Macondo well (Ryerson et al. 2011). Detection of spill-related hydrocarbons in the air indicated that not all volatile compounds (which are often also quite soluble in water) had dissolved during the oil’s ascent to the surface.

Dissolved concentrations of light hydrocarbons (e.g., benzene), which would normally quickly evaporate in a surface oil spill, were highly elevated within the rising plume, having largely dissolved during ascent (Payne & Driskell 2015a). This confirms that many of the most soluble hydrocarbons in the rising oil droplets dissolved out of the oil phase and into the water phase before the oil surfaced. Even some PAHs (e.g., naphthalene) dissolved into the water column, as evidenced by the excess loss of naphthalene relative to equally volatile, but less soluble hydrocarbons (Stout 2015h). Consequently, biota within the rising plume were exposed to a combination of dissolved chemicals (e.g., benzene and naphthalene) and to particulate oil.

4.2.5 Exposure at the Sea Surface

Key Points

- Oil slicks on the surface of the Gulf of Mexico cumulatively covered at least 43,300 square miles (112,115 square kilometers) of the northern Gulf of Mexico—an area about the size of the state of Virginia. At its maximum extent (on June 19, 2010), oil covered 15,300 square miles (39,600 square kilometers) of the Gulf of Mexico—an area about 10 times the size of Rhode Island.
- Surface oil was still detectable on August 11, 2010—113 days after the start of the incident and nearly a month after the Macondo well was capped.
- Once arriving at the sea surface, the oil was changed chemically and physically, as lighter hydrocarbons evaporated and dissolved into surface water, sunlight oxidized other oil components, and the oil mixed with water to form viscous water-in-oil emulsions ("mousse").
- Plankton, larvae, floating seaweed habitats (*Sargassum*), and larger animals living at the sea surface, such as sea turtles, marine mammals, and birds, were directly fouled by the surface oil. When fouled animals were directly observed, they were rescued if possible.
- Turbulence at the sea surface and use of chemical dispersants drove some surface oil back into the upper water column. This exposed the diverse biota living in near-surface habitats (less than 65 feet [19.8 meters] deep) to dissolved and particulate oil and biota near the sea surface to dispersant-derived chemicals.
- The air above surface oil contained elevated concentrations of volatile compounds that were evaporating from the surface oil. Small droplets (aerosols) also formed and traveled long distances through the air. Air-breathing animals were exposed to both evaporated compounds and aerosols.

During the DWH incident, surface and near-surface natural resources were exposed to oil that continually reached the sea surface throughout the 87-day spill. Upon reaching the surface, the oil spread horizontally and, to a lesser extent, vertically and 1) directly fouled (coated) some resources, 2) exposed biota to dissolved chemicals and particles of oil (and dispersant) within the upper water column, and 3) exposed biota to oil vapors above the water. In this section, the Trustees describe the exposure of natural resources to oil and dispersants at and near the sea surface (both offshore and nearshore), based on direct observations, remote sensing data, empirical chemical measurements, and modeling.

4.2.5.1 Chemical Fate of Oil and Dispersant at the Sea Surface

The oil that reached the sea surface after ascending more than 1,500 meters (5,000 feet) from the deep ocean was altered in composition from the oil expelled from the failed well at the sea floor. As noted previously, the expelled gas was dissolved and then consumed by bacteria within the waters of the deep sea and did not reach the sea surface (Section 4.2.3). Similarly, large fractions of soluble chemicals within the liquid oil were dissolved during the oil's buoyant ascent to the surface (Section 4.2.4).

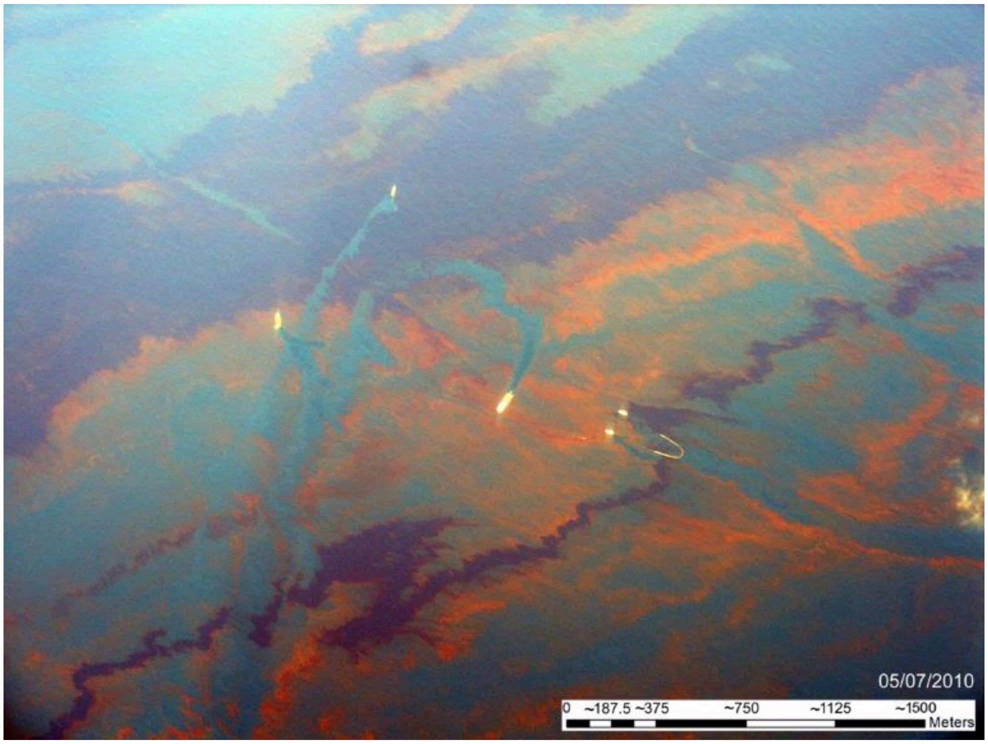
The Trustees studied the chemical compositions of dozens of floating surface oils collected during the active spill from the offshore oil slicks (Stout 2015h). These oils were variably weathered through the combined effects of dissolution and evaporation. While only traces of BTEX were detected in floating oil, their mere detection demonstrated that these compounds were not entirely dissolved during the oil's ascent—indicating some near-surface resources were exposed to these chemicals. TPAH50 concentrations in the surfacing oil rivaled or exceeded the concentrations in fresh DWH oil, because some PAHs were concentrated after the loss of more volatile and soluble hydrocarbons (Stout 2015h). Given the continuous resupply of surface oil for 87 days, PAHs were being regularly replenished in surface waters of the northern Gulf of Mexico, exposing resources there to these chemicals.

As the floating oils were transported and spread throughout the northern Gulf, they continued to weather through evaporation, dissolution into surface waters, emulsification, and photooxidation (the latter being a form of weathering in which ultraviolet light from the sun causes chemical reactions within the oil). Warm surface waters (28–30°C) and high solar radiation typical of the northern Gulf of Mexico in late spring and summer promoted both evaporation and photooxidation (Aeppli et al. 2012; Hall et al. 2013; Radovic et al. 2014; Ruddy et al. 2014; Stout 2015h). Photooxidation of the floating DWH oil is notable because the oxidized chemicals formed from this process can be toxic. Despite these weathering processes, there was no evidence that biodegradation of the floating oil slicks themselves had yet occurred (Stout 2015h).

The extent of oil weathering generally increased with increased distance from the wellhead, though not consistently given the variable conditions encountered. Most floating oil samples collected from across the widespread impacted region had lost more than one-third of their original mass due to weathering (Stout 2015h). TPAH50 concentrations in the floating oils ranged from 1,010 to 13,700 µg/g, the latter of which is somewhat higher than the fresh oil due to the concentrating effects of weathering on the PAHs (Stout 2015h). Most of the lightest PAHs (2-ring naphthalenes) were lost from the floating oils due to dissolution and evaporation, increasing the proportion of PAHs containing three or more rings as weathering progressed.

Chemical changes in the floating oil were accompanied by physical changes, including the increase in density and viscosity of the floating oils and concurrent formation of water-in-oil emulsions, sometimes referred to as "mousse." The oil emulsification often commenced within hours of the oil reaching the surface and was accompanied by the oil's color changing from dark brown to bright reddish brown or orange as the water content increased (Figure 4.2-13) (Leirvik et al. 2011). Animals and plants living at the surface were physically fouled upon contact with the sticky emulsions.

Over 1 million gallons of dispersant were sprayed directly on the sea surface (OSAT-1 2010); see Figure 4.2-4) in attempts to disperse the oil into the water and to reduce the overall amount available to reach the coastlines. Chemicals within the dispersant, particularly the surfactant DOSS, both persisted within undispersed oil on the sea surface and sank with dispersed oil into the waters below. DOSS and other dispersant chemicals were detected in samples of floc from the deep-sea floor collected 6 months after the spill (White et al. 2014) and at trace levels in some stranded oils that had reached shore (Stout 2015g). The latter observation indicates some dispersant was transported to shore as a residue in coalesced oil slicks.



Source: *NOAA.*

Figure 4.2-13. Variably emulsified oil on the sea surface, May 7, 2010. Scale is approximate. When the oil mixed with water to form emulsions, it changed from black to reddish-brown to orange. The oil typically sorted into long, relatively narrow strands of thicker oil, becoming particularly emulsified at the margins.

4.2.5.2 Aerial Extent of Surface Exposure to Oil: Remote Sensing Evidence

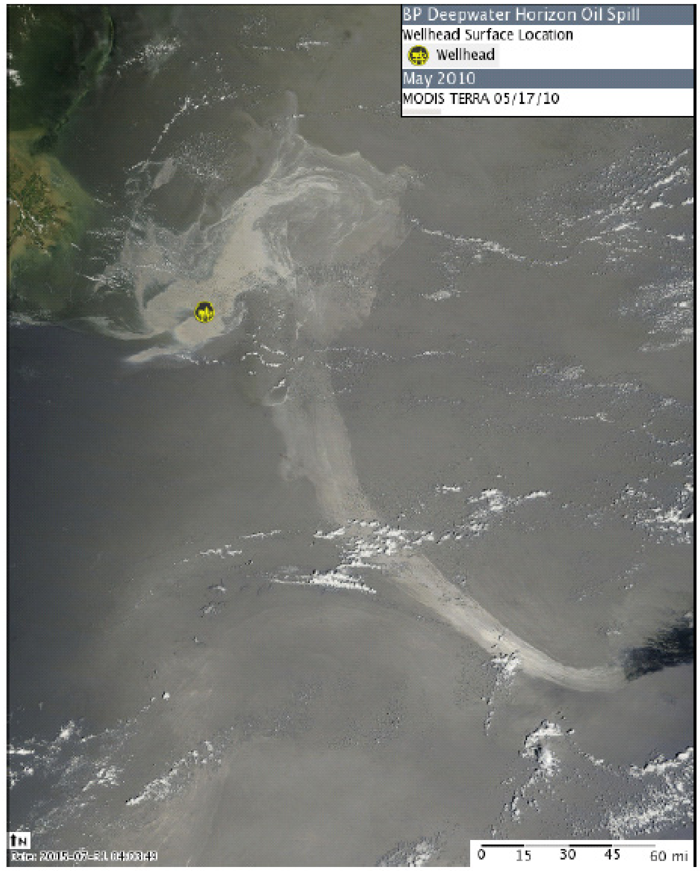
The Trustees used both airplane-mounted and satellite-based remote sensing methods to identify and quantify oil floating on the ocean surface. On most days between late April and late July 2010, the spatial coverage of floating oil exceeded what could be captured from an airplane. The full extent of the surface oil could only be captured from space.

The Trustees utilized several different sensors to detect oil on the sea surface during the spill, developed algorithms for classifying oil slicks of different relative thicknesses, and integrated the results of multiple analyses into a single model (Graettinger et al. 2015). The satellite sensors that collected the most data over the northern Gulf of Mexico during the spill were synthetic aperture radar (SAR) and NASA's Moderate Resolution Imaging Spectroradiometer (MODIS).

When meteorological conditions were optimal, MODIS images clearly showed the extent of the surface oil (Figure 4.2-14) (ERMA 2015). Unfortunately, such optimal conditions did not always exist during the spill. SAR, on the other hand, does not require daylight or clear skies, and it is particularly useful for detecting the presence of oil slicks (Garcia-Pineda et al. 2009). In addition, SAR data were more widely available with at least eight different satellites collecting data during the spill. Thus, the Trustees relied primarily on SAR data to estimate the spatial extent of surface oil during the spill.

The Trustees analyzed SAR data collected on 89 days between April 23 and August 11, 2010. Some days included multiple images per day, covering a broad swath of the northern Gulf of Mexico. Other days, the SAR data captured only a portion of the surface slicks.

Results revealed that floating DWH oil was not uniformly distributed on the ocean surface. Instead, oil concentrated along convergence zones created by ocean circulation patterns (Langmuir cells)—the same zones where sea creatures converge. The SAR images are snapshots in time and thereby revealed the oil on the ocean surface was constantly moving with the winds and currents. The surfaced oil sorted into patchy lines of thick oil and thin sheen, and "new" oil was constantly arriving at the sea surface near the well and replenishing the slicks at the surface.



Source: *ERMA (2015).*

Figure 4.2-14. MODIS image of DWH surface oil on May 17, 2010. On this date, the slick was over 200 miles (322 kilometers) long (light grey areas).

In the model integrating remote sensing data from numerous sources (Graettinger et al. 2015), the Trustees estimated the coverage of thicker oil emulsions and thin oil sheens when sufficient data were available. The vast majority of surface coverage of oil occurred as sheen, which is consistent with other

Remote Sensing of Oil

The Trustees used remote sensing data from both airplane- and satellite-mounted sensors that collected a wide range of spectra, including:

- Visible light wavelengths, similar to a camera.
- Infrared wavelengths, including thermal infrared that can show oil slicks that are warmer or cooler than surrounding sea water.
- Microwave (radar) wavelengths that penetrated clouds and did not require daylight.

Sensors mounted on airplanes had high resolution but could only capture a small sliver of sea surface covered with oil slicks. Sensors mounted on satellites could take an image of the entire northern Gulf of Mexico but with relatively coarse resolution. The Trustees integrated data from numerous sensors to evaluate surface oil coverage over the northern Gulf of Mexico during and after the spill.

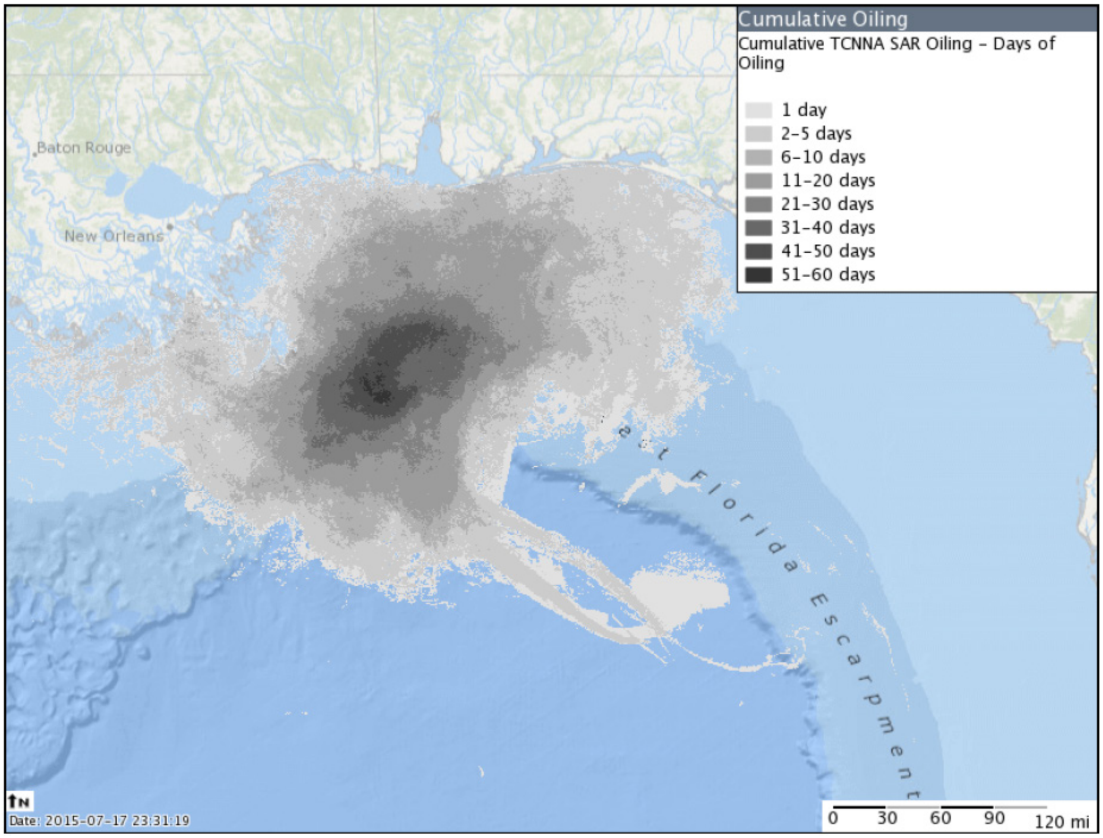
oil spills (Leifer et al. 2012). Thicker oil was often evident in long, narrow strands along the convergence zones, with sheen evident between the thick oil strands (see Figure 4.2-13).

After overlaying all available SAR images, the Trustees developed a "cumulative surface oil days" footprint that covered 112,115 square kilometers (43,300 square miles) (Figure 4.2-15)—an area approximately the size of the state of Virginia. This cumulative footprint shows the area where SAR detected oil at any time during the 89 days for which images are available. Because each SAR image provides only a snapshot in time, and the oil was constantly moving across the ocean surface, the SAR imagery likely missed some locations where oil was present. Thus, the Trustees' estimate of cumulative oil coverage based solely on oil present in SAR images likely underestimates the cumulative extent of surface oiling. Other researchers using different methods estimated a greater cumulative extent of oiling. For example, MacDonald et al. (2015) calculated a cumulative surface oil footprint of 57,500 square miles (149,000 square kilometers) using statistical interpolation to estimate oil coverage where and when SAR images were not available. Not surprisingly, areas closest to the wellhead had the most number of days with detectable oil, and areas farthest from the wellhead had the least number of days with detectable oil (Figure 4.2-15); see also MacDonald et al. (2015).

The maximum extent of surface oil detected in the SAR imagery on any single day was 39,600 square kilometers (15,300 square miles) on June 19, 2010 (Figure 4.2-16); this area is about 10 times the size of Rhode Island.

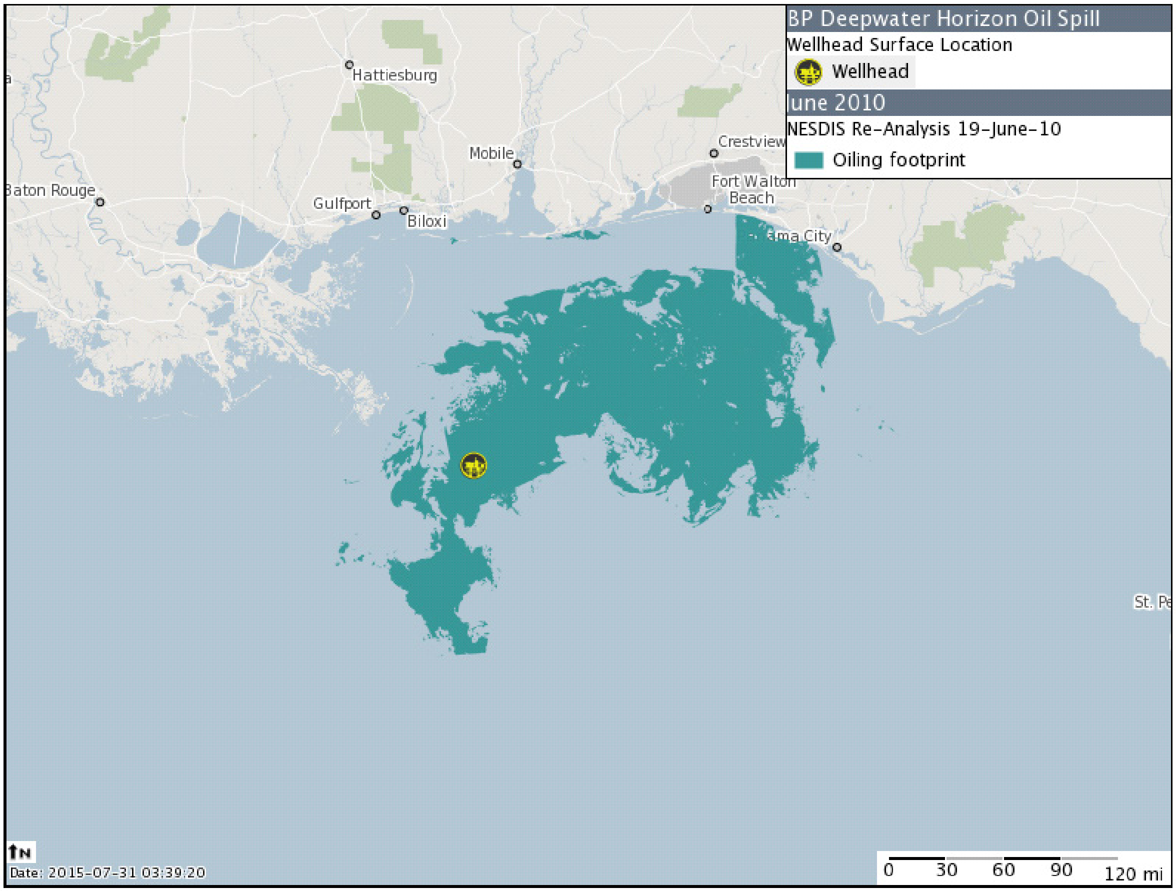
DWH oil slicks were detectable using remote sensing from the start of the incident until at least August 11, 2010. As evidenced in the various figures (Figure 4.2-15 and Figure 4.2-16), despite all the response activities that were conducted (e.g., dispersant application, skimming, and in situ burning) to control the spread of oil, the oil slicks, sheens, and emulsions were continually observed on the sea surface of the northern Gulf of Mexico over this 113-day period, with a substantial amount of oil eventually reaching shorelines.

In addition to quantifying the extent of oil slicks in the open ocean, the Trustees closely examined the extent of surface oil found in nearshore environments, such as the many sounds, bays, and bayous, where diverse resources exist. SAR imagery (e.g., Figure 4.2-17) clearly shows oil slicks repeatedly approaching the shoreline from early May through early August 2010. The Trustees also examined aerial photographs and remote sensing images collected from airplanes flying over the coastline's many marshes. These images showed, for example, the extent of oil movement into a marsh (Figure 4.2-18) and oil near shorelines where response teams did not observe oil (Figure 4.2-19). These remote sensing data provide additional lines of evidence for the spatial extent of oil in nearshore waters.



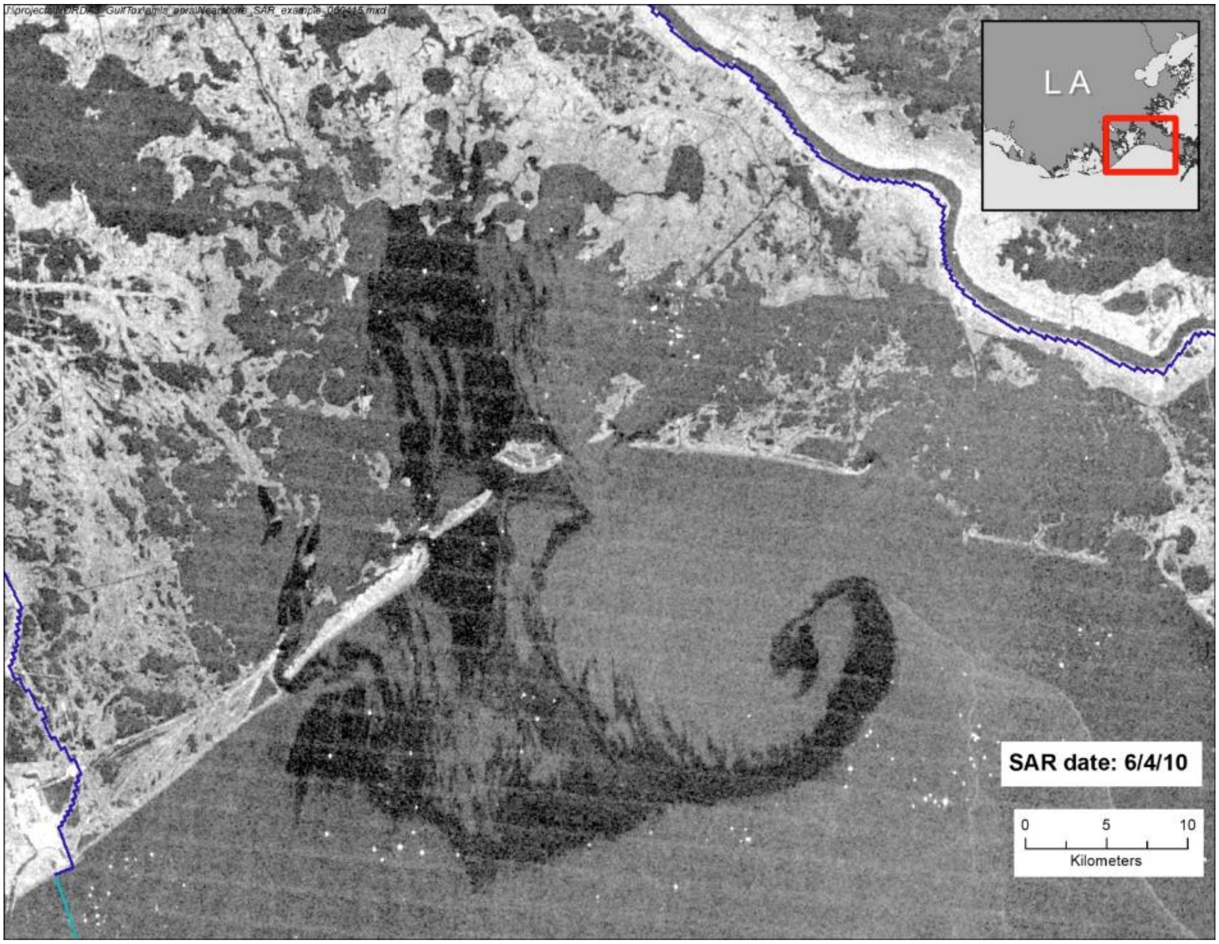
Source: ERMA (2015).

Figure 4.2-15. Cumulative footprint of surface oil coverage and the total number of days that oil was detectable on the ocean surface based on SAR imagery. The cumulative area where surface oil was detected covers approximately 112,115 square kilometers (43,300 square miles).



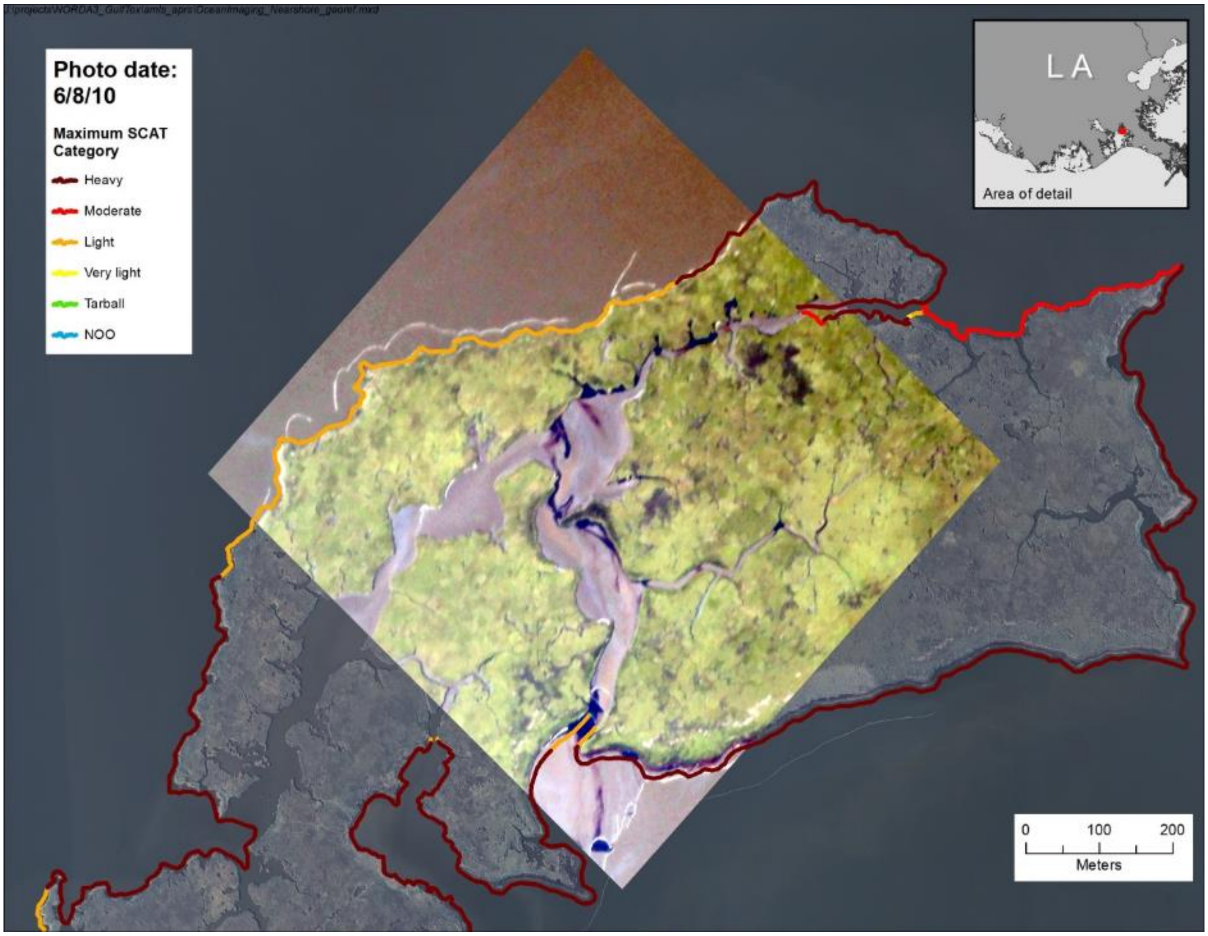
Source: ERMA (2015).

Figure 4.2-16. Extent of surface oil detected by SAR on June 19, 2010. The oil slick on this date covered approximately 39,600 square kilometers (15,300 square miles).



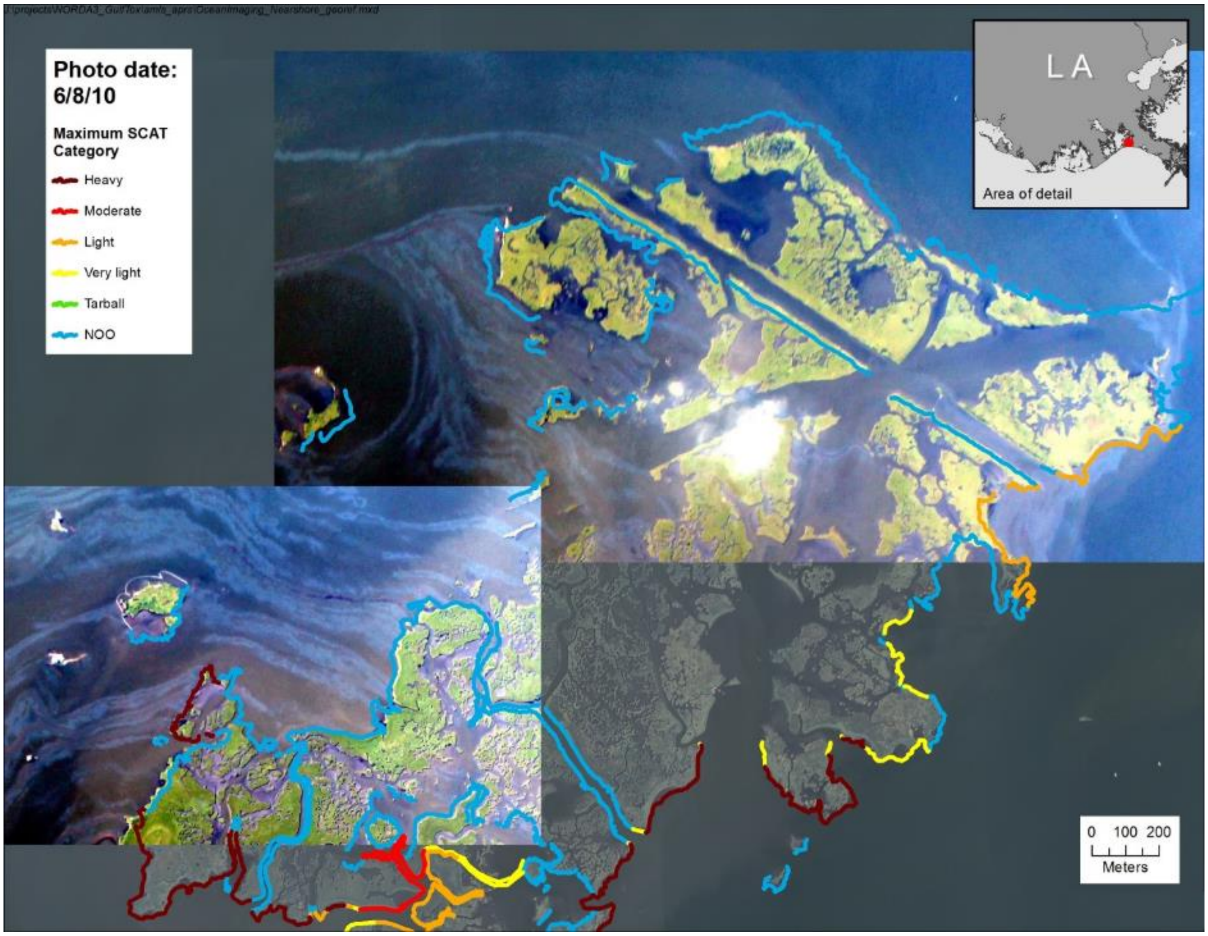
Source: Abt Associates.

Figure 4.2-17. DWH surface oil entering Barataria Bay, Louisiana, on June 4, 2010. In SAR images, oil slicks appear dark.



__Source:__ Abt Associates; photograph © Ocean Imaging Corp., used by permission.

Figure 4.2-18. Movement of DWH thick oil (black) and sheen (grey) into marsh inlets.



***Source**: Abt Associates; photographs © Ocean Imaging Corp. Used by permission.*

Figure 4.2-19. Surface oil (wavy grey bands) intersecting a marsh shoreline where Shoreline Cleanup Assessment Technique (SCAT) response teams recorded "no observed oiling."

4.2.5.3 Exposure of Biota to Surface Oil

The water-in-oil emulsions that developed on the sea surface formed viscous oil masses that stuck to larger biota and engulfed smaller biota. The Trustees conducted numerous surveys with photographic documentation confirming that, for example, turtles and *Sargassum* (Figure 4.2-20) and dolphins (Figure 4.2-21) were exposed to oil during the spill. In addition to visual observations, the Trustees conducted chemical analyses of oil from hundreds of samples of *Sargassum*, turtles, and dolphins to confirm the oil observed was derived from the DWH oil—and to determine the state of weathering of the oil to which these animals were exposed.

Source: The Washington Post/Getty Images.

Figure 4.2-20. A sea turtle and *Sargassum* exposed to surface oil during the DWH spill.

The Trustees identified more than 200 turtles with DWH oil on their shells, and some of these turtles were recovered up to 150 miles from the well (Stout 2015e). Floating *Sargassum* samples collected up to 100 miles from the well were also shown to have been impacted by DHW oil (Stout & Litman 2015b). *Sargassum* exposure is notable as it forms an important habitat for sea life, including juvenile sea turtles.

***Source**: NOAA.*

Figure 4.2-21. Dolphins exposed to DWH oil.

The Trustees also identified DWH oil swabbed from the exterior of 14 stranded dolphin carcasses, mostly collected from Port Fourchon and Grand Isle beaches in Louisiana (Stout 2015c).

Birds too were exposed to surface oil. Birds feed on fish and zooplankton and rest on the water. Some areas, such as *Sargassum* mats, attract birds due to the abundance of food and areas to rest.

The following sections further document exposure of a broad range of aquatic, terrestrial, and avian biota, and the injuries to those biota as a result of the exposure.

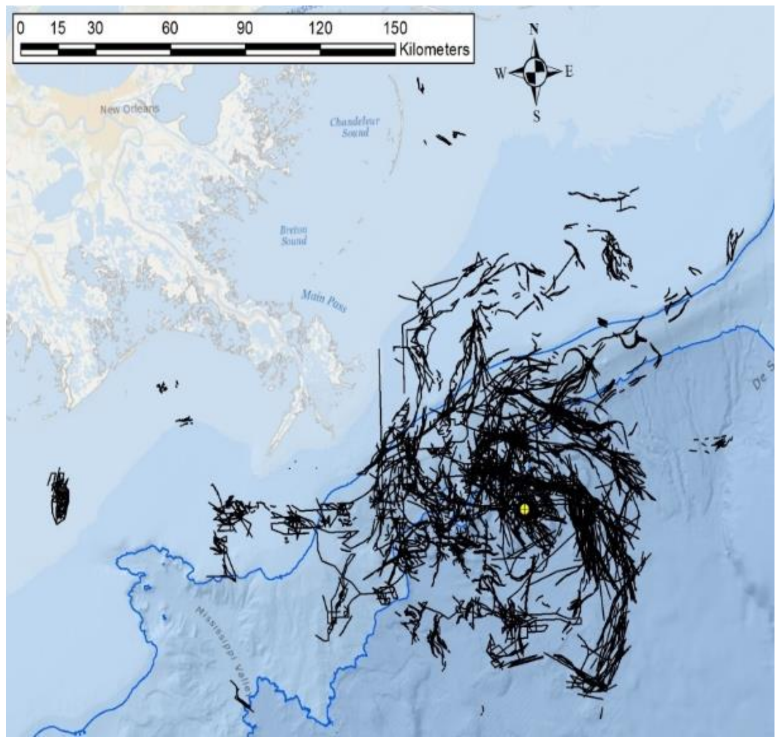
4.2.5.4 Exposure to Dissolved and Particulate Oil in Surface Water

Wave action and turbulence within the upper water column naturally disperses and entrains droplets of oil into the upper water column. Lighter oil constituents can dissolve directly from the surface slick and form droplets entrained into the upper water column. The buoyancy of these droplets will dictate whether or when they resurface, with smaller droplets remaining submerged. Some natural entrainment of the floating DWH oil occurred across the northern Gulf of Mexico where floating oil existed. Also, as discussed previously, planes and vessels sprayed some floating oil with 1.07 million gallons of chemical dispersants that were intended to break up the oil slicks into smaller droplets that would then disperse or become entrained in the water column.

Dispersant was applied by aircraft over 305 square miles (790 square kilometers) of floating oil that was at least 3 nautical miles offshore, with 98 percent of the dispersant applied more than 10 nautical miles offshore (Houma 2010). Most of the surface applications occurred within 55 kilometers of the well (Figure 4.2-22).

Both wave action and chemical dispersion drove oil back below the sea surface. This dispersion exposed upper water column biota, including plankton, fish, and invertebrates, to dispersed oil droplets, chemicals that dissolved from the oil, and dispersant chemicals (Hemmer et al. 2011) (Section 4.3, Toxicity; Section 4.4, Water Column).

In addition, as mentioned previously (Section 4.2.3.3), a proliferation of bacterial activity at the sea surface in response to the oil led to the formation of marine oil snow, which ultimately lost its buoyancy and sank, carrying oily biomass to the sea floor (Passow 2014; Passow et al. 2012; Stout & Passow 2015). This process was likely enhanced in areas where dispersants were applied to the surface, as bacteria preferentially acted upon the dispersed oil. Some studies indicated that plankton were unable to consume bacteria in the presence of dispersant, thereby disrupting the base of the pelagic food web (Ortmann et al. 2012).



Source: *BP (2014b).*

Figure 4.2-22. Recorded tracks of aerial and surface vessel applications of dispersants; 200- and 1000-meter bathymetric contours are shown.

4.2.5.4.1 Empirical Evidence—Water Chemistry and Forensic Analyses

Surface waters (defined roughly as the upper 20 meters or 65 feet in depth) associated with DWH slicks in the northern Gulf of Mexico were sampled and found to be contaminated with PAHs and other oil-derived chemicals. From May through July 2010, PAH concentrations sufficient to be harmful to sensitive life stages of biota (Section 4.3, Toxicity) were present in a wide geographic area in the northern Gulf surface waters (Section 4.4, Water Column) (Payne & Driskell 2015a; Rice 2014). The water chemistry data provide direct evidence of DWH oil exposure experienced by resources living within the Gulf surface waters.

Although thousands of water samples were collected during the spill, sampling in the region immediately surrounding the well (within approximately 1 mile) was restricted during the response. The Trustees evaluated TPAH50 concentrations in water samples collected from the upper water column under floating oil on the Gulf surface. These data are compiled in NOAA's data management system

(DIVER) and include samples collected by BP and the Trustees as part of the NRDA, samples collected during spill response, and data that BP posted on its public website. A total of 378 samples in the upper 65 feet (19.8 meters) of the water column were identified that were co-located with oil slicks based on SAR analysis (Section 4.4, Water Column) (Travers et al. 2015).

A subset of water samples collected at and immediately below the surface appeared to contain mixtures of water and entrained surface oil, having TPAH50 concentrations as high as 90,500 µg/L (ppb). These elevated concentrations are representative of exposures experienced by aquatic biota that passed through or stayed within inches of the water surface when surface oil was present. The remaining shallow subsurface water samples were collected below the surface but at depths of less than 20 meters (65 feet) deep. The TPAH50 concentrations of these subsurface samples ranged from undetectable to 240 µg/L (ppb).

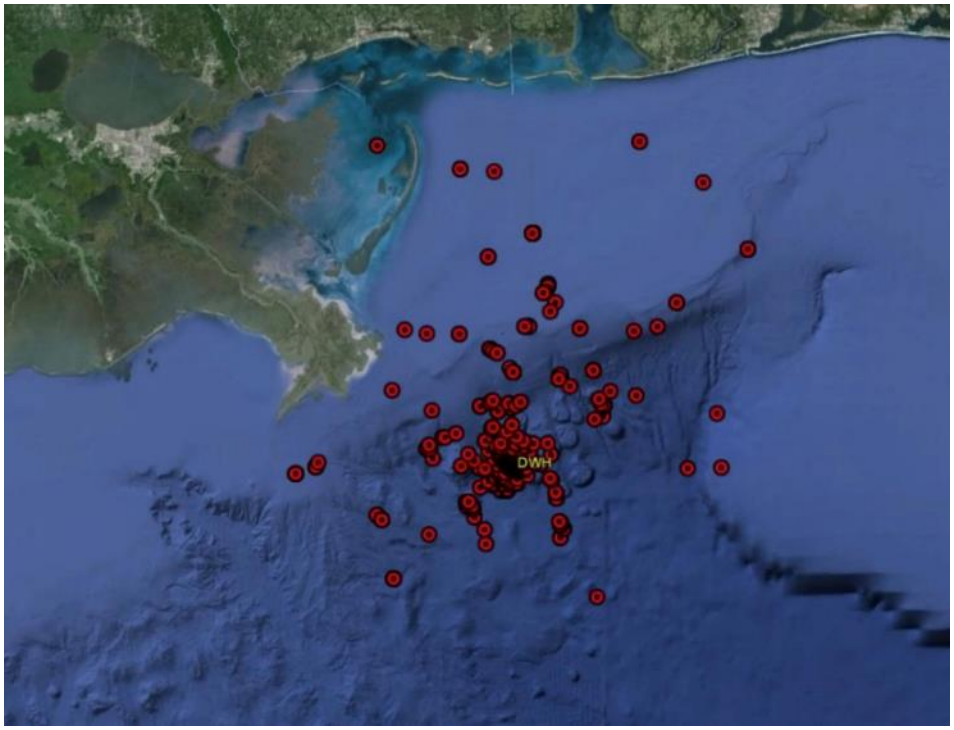
The concentrations reported in the previous paragraph are notable because, the Trustees determined through extensive sampling that ambient ("background") water in the northern Gulf of Mexico has almost undetectable concentrations of PAHs. Specifically, NRDA water samples collected in the upper 20 meters (65 feet) of the water column in areas unaffected by the DWH incident had an average TPAH50 concentration of less than 0.06 µg/L (ppb) (Payne & Driskell 2015a).

As Section 4.3 (Toxicity) discusses in detail, the amount of TPAH50 that is toxic depends on many factors, including species and life stage. To verify potential exposure of biota to PAHs in the upper water column, the Trustees evaluated water samples with TPAH50 concentrations that exceeded 0.5 µg/L (ppb), which is a concentration sufficiently high to harm sensitive life stages of biota (Section 4.3, Toxicity). Of samples co-located with oil slicks based on SAR analysis, 54 percent collected from 0 to 2 meters beneath the surface exceeded a TPAH50 concentration of 0.5 µg/L (ppb). The percentage of samples exceeding 0.5 µg/L (ppb) decreased with depth:

- 35 percent of samples at 2–10 meters depth exceeded this threshold.
- 15 percent of samples at 10–30 meters depth exceeded this threshold.
- 6 percent of samples from 30–50 meters exceeded this threshold.

Further information is available from the technical memorandum on analysis of water column TPAH50 data (Travers et al. 2015). These data suggest that the majority of PAHs entrained in surface waters remained close to the surface at depths of less than 65 feet (20 meters). Similar declining concentrations with depth were found during a test of dispersant effectiveness conducted during the response, in which hydrocarbon concentrations and fluorescence (an indicator of hydrocarbons) were measured beneath undispersed and dispersed slicks at varying depths (Bejarano et al. 2013).

The Trustees forensically evaluated selected water samples collected in the upper 20 meters (65 feet) of the water column (Figure 4.2-23). For this analysis, care was taken to exclude samples that may have included oil from the surface slick. The Trustees confirmed that DWH oil was present in 359 of these near-surface water samples, which were collected at locations as far as 97 kilometers (60 miles) in most directions from the wellhead (Figure 4.2-23) (Payne & Driskell 2015a).



Source: *Payne and Driskell (2015a).*

Figure 4.2-23. Locations of upper water column samples (depths less than 20 meters [65 feet]) that were collected as part of the NRDA, in which forensic analyses confirm the presence of DWH oil.

For insights on contamination in nearshore/estuarine areas, the Trustees evaluated water chemistry data in floating oil from Terrebonne, Barataria, and Mobile Bays, and Chandeleur and Mississippi Sounds (Payne & Driskell 2015c). Consistent with methods used for the offshore areas, the Trustees considered the sample locations relative to the oil slicks detectable in SAR imagery collected on the same day. Of the more than 3,700 nearshore/estuarine water samples collected between April and August 2010, most were collected *prior* to the arrival of floating oil or in places away from floating oil; only 121 of these samples were collected within 1 kilometer (0.6 mile) of an oil slick detectable in a SAR image on the same day (Travers et al. 2015).

The evaluation also considered how oil concentrations collected near SAR-detected oil slicks varied with depth in the nearshore/estuarine water column. Most of the samples were either collected at the water surface (i.e., a sample depth of 0 meters) or the sample depth was not reported. Within this group of surface samples, TPAH50 concentrations ranged from 0 to 29 µg/L (ppb). Some of these surface water samples likely included traces of surface slick oil. Of the nearshore/estuarine water samples associated with surface slicks that were collected below the water surface, on the other hand, the TPAH50 concentrations were lower, ranging from 0 to 0.7 µg/L (ppb) (Travers et al. 2015).

The Trustees conducted a forensic assessment of nearshore water samples collected during the year after the spill. DWH oil was forensically identified in 361 samples that supported two inferences: DWH oil was present in the nearshore/estuarine water column during the DWH incident, and DWH oil persisted in some nearshore/estuarine waters into 2011 (Payne & Driskell 2015c). Oil in nearshore/estuarine waters was primarily in a dissolved form rather than particulate oil. This differs from most samples collected in the offshore surface waters, which also contained oil droplets. Two explanations were considered for the presence and predominance of dissolved oil in the nearshore/estuarine waters. For the nearshore samples collected in summer 2010, the dissolved oil may have been related to dissolution from floating oil arriving at the coastline. On the other hand, for samples that found dissolved oil in nearshore waters months after floating oil was arriving, the dissolved oil likely resulted from dissolved components leaching from previously deposited oil sources in the nearshore environment (Payne & Driskell 2015c).

4.2.5.5 Oil Vapors and Airborne Droplets (Aerosols)

Previous work has demonstrated the presence of volatile hydrocarbons in air samples collected less than 2 feet (0.5 meters) above floating oil (Payne et al. 1980). Research during the DWH incident demonstrated the widespread occurrence of volatile and less volatile (intermediate and semi-volatile) compounds in atmospheric plumes emanating from the floating oil (de Gouw et al. 2011; Ryerson et al. 2011). Because marine mammals, sea turtles, and birds breathe just above the water-air interface, it is likely that evaporated (volatile, intermediate, and semi-volatile) constituents from the oil were inhaled. Some evidence for this was found in a lung tissue sample of a dolphin carcass that contained hydrocarbons reasonably derived from inhalation of DWH vapors (Stout 2015c).

In addition, disruptions to the air-water interface (e.g., by the action of breaking waves, wind, raindrops, animals breaking the surface) may have caused oil to be suspended in the air above surface slicks (Haus 2015; Murphy et al. 2015). Also, volatiles and particles in the air column can undergo chemical transformations and coalesce to form suspended particulates (de Gouw et al. 2011). Thus, it is also possible that liquid (or aerosolized) oil could have entered the lungs of an animal swimming among floating oil.

4.2.6 Exposure in the Nearshore

Key Points

- Animals and habitats along the northern Gulf of Mexico coastline were exposed to weathered DWH oil as the oil slicks made landfall along the region's beaches and marshes and entered shallow water ecosystems within sounds, bays, and bayous.
- Floating DWH oil entered coastal and estuarine waters in early- to late-May and was present at times through mid-August, nearly 4 months after the spill started. Animals and plants in the water column were exposed to the oil slicks and oil that was dissolved and entrained beneath them.
- DWH oil was stranded along the coastline spanning at least 1,300 miles (2,100 kilometers) of Louisiana, Mississippi, Alabama, Florida, and Texas, including beaches (51 percent of affected coastline), marshes (45 percent), and other (mostly) manmade shorelines (4 percent).
- The Trustees found that oil-derived chemicals measured in nearshore sediments and soils forensically matched the DWH oil and were far higher in concentration than existed in samples collected prior to the spill or from apparently unoiled locations.

- Oil persisted in some areas for years after the spill. For example, for the shorelines observed to be oiled in 2010, approximately 48 percent still had some degree of oiling after 1 year and 39 percent had some degree of oiling after 2 years.
- Some SOMs that formed in 2010 persisted and were broken up during subsequent storms, causing re-oiling of some shorelines.

Along the northern Gulf of Mexico coast, many different types of habitats were exposed to weathered DWH oil when slicks came ashore during the spring and summer of 2010. The slicks made landfall on the region's beaches and marshes and entered shallow water ecosystems within sounds, bays, and bayous.

In the initial days of the spill, prevailing currents and winds kept oil slicks offshore, but eventually winds played a major role in pushing the floating oil toward the northern Gulf of Mexico coasts (Boesch 2014). The first oil reportedly reached shorelines in Louisiana on approximately May 15, 2010, and about 2 weeks later in Mississippi, Alabama, and Florida (OSAT-1 2010). The heaviest shoreline oil deposition took place from May to August. Almost all coastal oiling took place in Louisiana, Mississippi, Alabama, and Florida; light to trace amounts of DWH oil were also found along the Texas coast during Rapid Assessment Team surveys (Nixon et al. 2015).

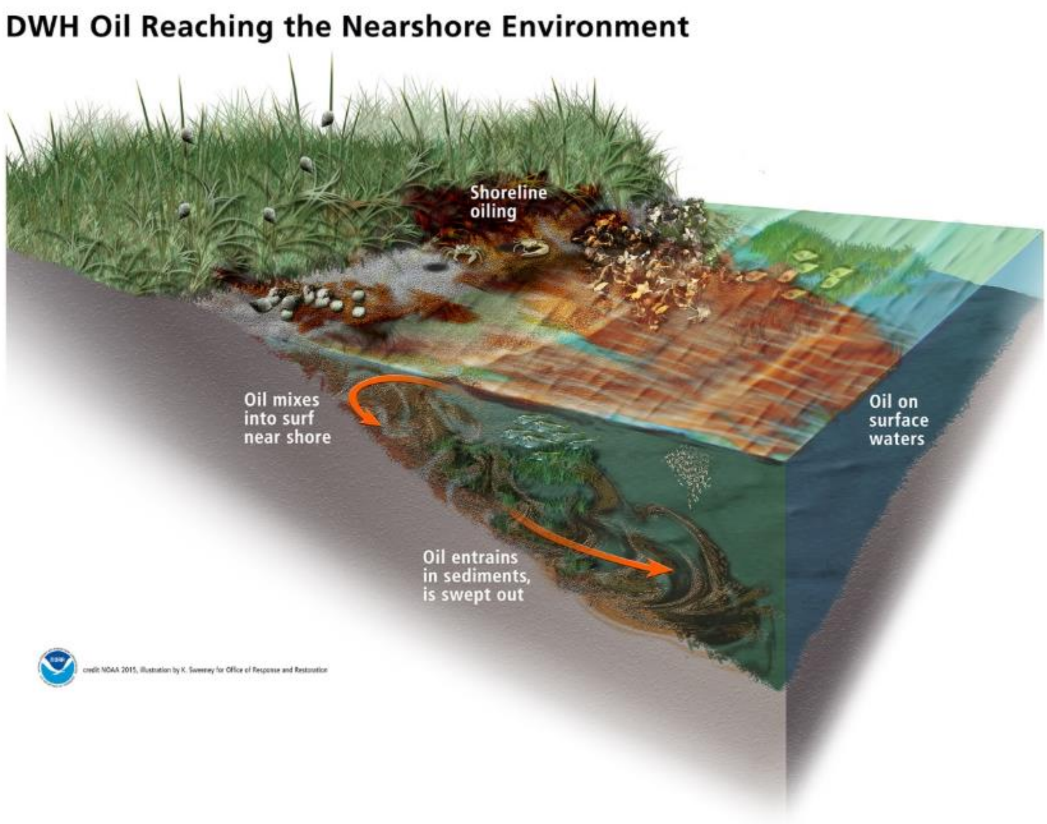
The degree of oil exposure was not uniform along the expansive and complex coastlines characteristic of the northern Gulf. Some areas were heavily oiled, and many of these were repeatedly oiled. Other areas were moderately oiled, some lightly, and some were found to not have any observable oil. Many areas could not be surveyed given the overall geographic scale of the spill and the complex and remote nature of Gulf Coast marshes and swamps. In some cases, evidence of oiling varied among multiple surveys at the same locations. This generally occurred due to patchiness of oil, the timing of field work (e.g., some sites where no oil was observed by a response team were later observed to be oiled by a NRDA team), or the differences in survey objectives for response (i.e., cleanup planning) and for NRDA (i.e., exposure assessment).

4.2.6.1 Pathway of Floating Oil to and Within the Nearshore Environment

For purposes of this document, nearshore refers to both shoreline and shallow water habitats adjacent to marsh and beach shorelines and includes oyster reefs, SAV, and unvegetated areas.

As discussed in previous sections, the DWH oil that reached the northern Gulf of Mexico nearshore environment was significantly weathered by its 1,500-meter (5,000-foot) vertical ascent through the offshore water column and its lateral transport across many miles of open ocean over many days and weeks. By the time the floating oil reached shorelines, much of it was in the form of viscous emulsions that stuck to sand, mud, sediment, vegetation, and biota. Some oil also arrived in the nearshore as thinner sheens and slicks (Zhang et al. 2015a).

As the DWH oil floating on the sea surface was carried toward shore and washed up ("stranded") on various types of shore, it was deposited and redeposited in several ways (Figure 4.2-24) (Zhang et al. 2015a). Some of the oil stranded on, coated, or was incorporated into marsh and beach substrates, including soils and plants. Some of the oil mixed with nearshore sediments in the surf zone and was carried back out into nearshore sediments, most notably within 50 meters (160 feet) of oiled vegetated shorelines. Some of this oil was incorporated into the shallow sediments and into SAV, and some remained on the nearshore bottom as SOMs. Later, storms buried or exposed stranded oil and broke up and redistributed SOMs, periodically re-oiling the shoreline (Stout & Emsbo-Mattingly 2015; Zhang et al. 2015a). Although dispersants were not applied to oil near coastlines, traces of dispersant chemicals were found on some stranded oils in the nearshore (Stout 2015g; White et al. 2014).



Source: *Kate Sweeney for NOAA.*

Figure 4.2-24. Conceptual illustration showing arrival of oil and oil exposure in a nearshore marsh environment. Oil on the sea surface was carried toward the shore. The oil then stranded onshore and some was mixed with nearshore sediments. A portion of the oil in nearshore sediments was swept farther out into shallow offshore areas and in some locations formed SOMs.

4.2.6.2 Observations of Nearshore Oil Exposure

The location, magnitude, and persistence of exposure of nearshore habitats to DWH oil was documented through 4 years of field surveys that included observations, measurements, and the collection and analysis of thousands of samples. Survey teams evaluated nearshore oiling on foot and by boat, occasionally with direction from response teams in airplanes. Nearshore exposure to DWH oil was also documented by aerial observations and remote sensing.

The U.S. Coast Guard (USCG) and other agencies conducted shoreline surveys to characterize and prioritize shorelines for cleanup. These were referred to as Shoreline Cleanup Assessment Technique

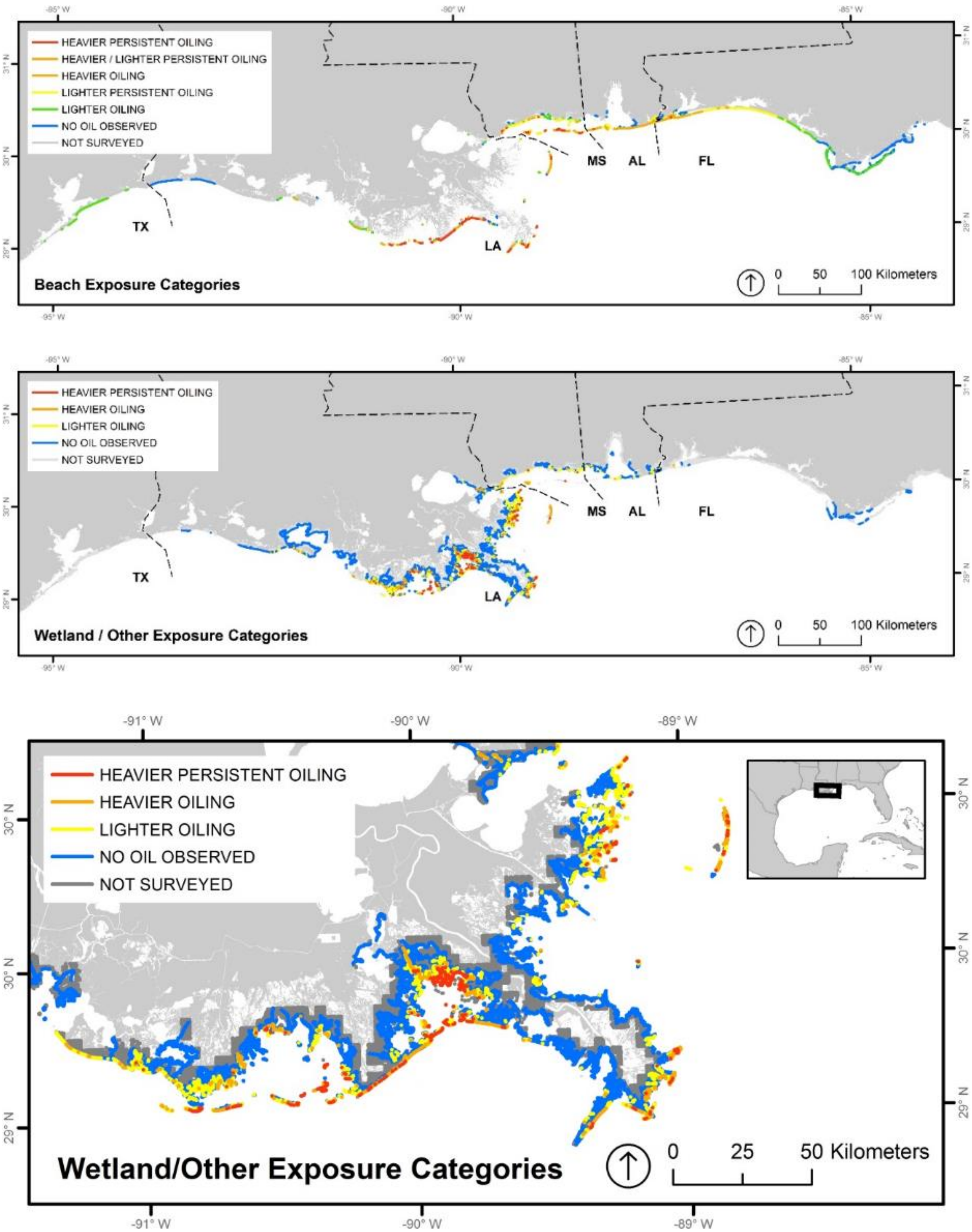
(SCAT) surveys. Trustees also conducted separate surveys intended to characterize linear extent of shoreline oiling. For example, the state of Texas collected data between July 7, 2010, and July 15, 2010, using the Rapid Assessment Teams (TCEQ, 2013, as cited by Nixon & Michel 2015). These data were collected as part of the response by cooperative survey teams and documented observed oiling conditions during cleanup operations.

Based on all of these surveys, oil was observed on over 2,100 kilometers (1,300 miles) of shoreline from Texas to Florida, out of 9,545 kilometers (5,931 miles) of surveyed shoreline (Figure 4.2-25) (Nixon et al. 2015). The shoreline lengths reported are 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 SCAT survey dataset was supplemented with other available observational shoreline oiling data, including that collected by operational cleanup efforts and data collected under the NRDA (most notably, the Rapid Assessment survey). The Rapid Assessment data provide additional coverage in describing shoreline oiling in some marsh areas of Louisiana between August 14, 2010, and October 16, 2010. As part of the response effort, Rapid Assessment Teams surveyed the Texas coastline from Corpus Christi to the Texas/Louisiana border during the period of July 5, 2010, through September 9, 2010 (Unified Command for DWH Incident Command Post 2010). These surveys represent a supplemental source of surface shoreline oiling data for these locations.

By state, the majority of oiled shoreline (approximately 65 percent) was in Louisiana, including the vast majority of oiled wetland shorelines (95 percent; Nixon et al. 2015). The heaviest and most persistent shoreline oiling occurred in salt marshes in northern Barataria Bay (Michel et al. 2013; Zengel & Michel 2011). Most of the oil that response teams observed was along the shoreline edge; oil that penetrated marshes was often inaccessible to response teams and thus not observed.

These oiled shoreline lengths are based on a compilation of the results of many shoreline surveys but are not comprehensive of all oil observations. Surveying a given segment of shoreline did not ensure that all oil on that segment was observed; oil can be difficult to find in marshes (see Figure 4.2-18) and sometimes washed ashore after the segment was surveyed. Thus, shorelines were sometimes documented as “no oil observed,” but this does not preclude the possibility that oil was present and not observed or that oil did not arrive after the survey (see Figure 4.2-19).

The geographic extent of shoreline oiling caused by the DWH incident is the largest of any marine spill globally (Nixon & Michel 2015; Nixon et al. 2015). Further analysis by the Trustees indicates that the 2,100-kilometer (1,300-mile) estimate is less than the actual length of oiled shoreline. Shoreline delineation used by SCAT to support cleanup operations represents the land-water interface at a 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 of the details of the vegetated land-water interface where the majority of marsh oiling occurred. Consequently, marsh shoreline lengths that are based on the 2008 data layer underestimate the 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 most of the oil exposure occurred (Wobus et al. 2015). This analysis indicated that the length of the oiled marsh edge in



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

Figure 4.2-25. Extent of shoreline oiling by oil exposure categories for beaches (top) and coastal wetland and other shoreline habitats (middle and zoomed in, bottom). Oil was observed from Texas to Florida (Nixon & Michel 2015). 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.

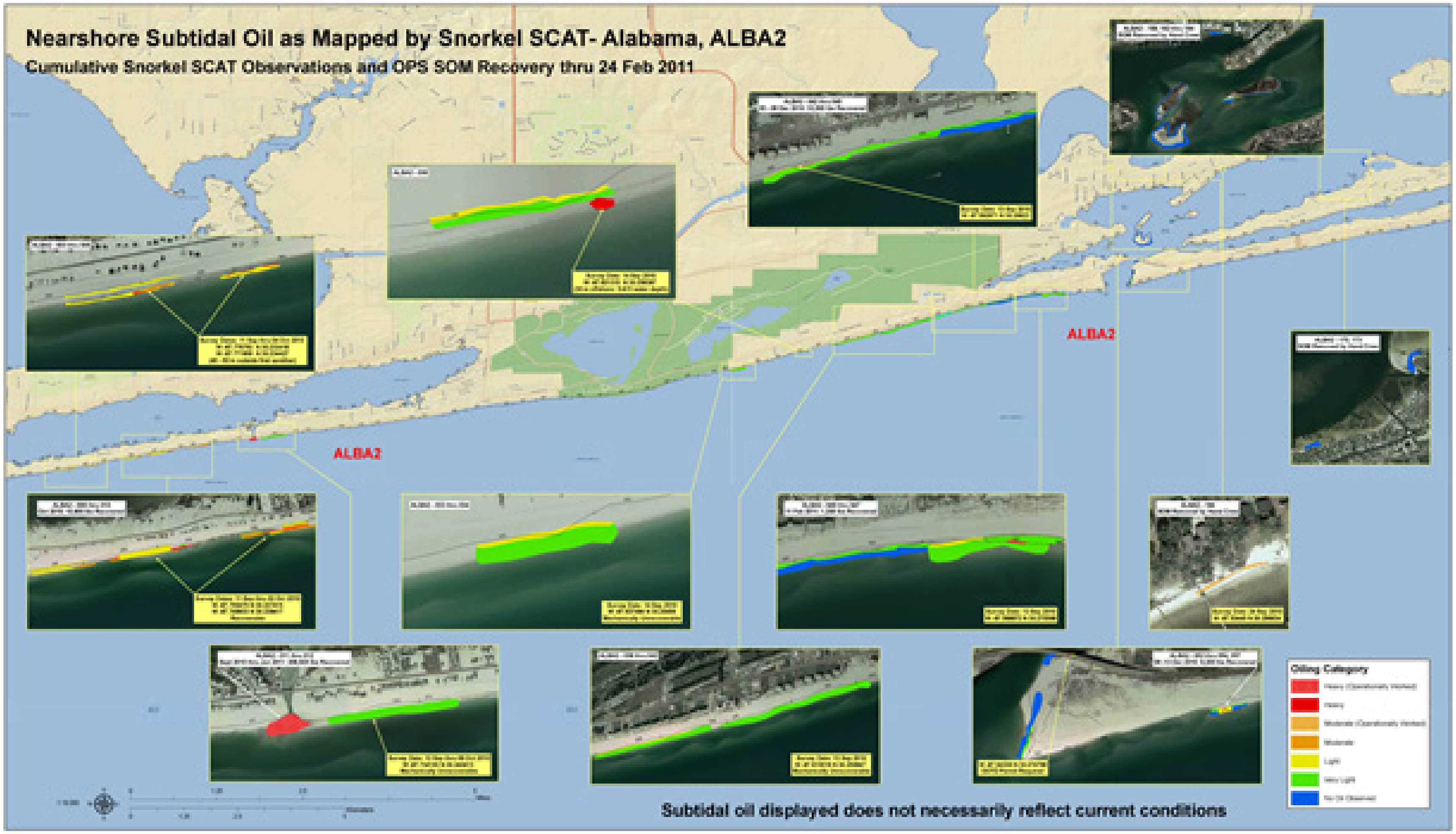
Louisiana exceeds estimates based on the 2008 shoreline by up to 40 percent in some areas. These factors indicate that the actual shoreline oiling was greater than the 2,100 kilometers (1,300 miles) reported in this section. It should also be noted that these shoreline lengths represent the cumulative shoreline observed to be oiled at any time over 4 years of observations. Some shorelines were oiled only once, others were repeatedly oiled.

Oil from the DWH incident persisted in the nearshore environment. Of the shorelines observed by response workers as having been oiled in 2010 (excluding Texas), approximately 48 percent still had some degree of oiling after 1 year and 39 percent had some degree of oiling after 2 years (Michel et al. 2013).

Stranded oil was observed in various forms, including discrete tar balls (less than 10-centimeter diameter), patties (10- to 51-centimeter diameter), and oil mats (greater than 51-centimeter diameter). These forms sometimes occurred as “pure” viscous emulsions of oil, but more often were mixtures of sand bound by lesser amounts of oil. Shoreline response teams found that oil stranded in coastal wetlands was typically pooled on the surface. Response teams did not frequently document oil penetrating into the marsh soils (Michel et al. 2013). However, pooled oil at the surface would have been considerably easier to observe than oil that penetrated the marsh and deposited in the soil. Oil in marshes also coated the stems of coastal wetland vegetation. 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, and wildlife.

Oil that sunk within subtidal areas to form SOMs was less visible than oil that stranded on shorelines. SOMs typically formed in the areas between the toe of the beach and the first offshore bar (BP 2014a; Hayworth et al. 2015; Michel et al. 2013; OSAT-2 2011; Urbano et al. 2013; Wang & Roberts 2013). The full extent to which SOMs formed along the coasts is unknown. SCAT efforts conducted throughout 2010 and 2011 via snorkeling revealed they could be common. For example, Figure 4.2-26 shows the frequency of SOMs observed to have formed along an approximately 32-kilometer (20-mile) stretch of sandy shorelines in Alabama. Comparable snorkel survey maps exist for other areas (ERMA 2015). Despite remedial efforts, DWH SOMs have continued to be reported and cleaned up, most recently offshore of East Grand Terre Island Louisiana in March 2015.

During the erosional winter months and during storms, SOMs from subtidal areas were eroded and re-deposited on shorelines (BP 2014a; Clement et al. 2012; Michel et al. 2013; OSAT-2 2011; Stout & Emsbo-Mattingly 2015; Urbano et al. 2013). This was especially true during high-energy storms, such as Tropical Storm Lee (September 2011) and Hurricane Isaac (August 2012). Thus, SOMs in the subtidal zone became a chronic source of DWH oil to beaches, which may continue for some time. This phenomenon has been observed following other oil spills as well (e.g., Gundlach et al. 1983).



Source: *ERMA (2015).*

Figure 4.2-26. Map showing the extent of SOMs in subtidal areas along a portion of the Alabama coastline determined via snorkel SCAT (through Feb. 24, 2011). Maps showing the frequency of SOMs in other areas are available at ERMA (2015).

4.2.6.2.1 Overview of Sampling and Chemistry Analysis in the Nearshore Environment

Soon after oil from the DWH incident came ashore, the Trustees deployed teams to survey and collect samples of stranded and nearshore floating oils. These samples were chemically analyzed to determine whether they could be forensically compared to DWH oil. Overall, 1,300 stranded and floating oil samples were collected under this effort (Stout 2015g), and the results are discussed in Section 4.2.6.3.

In addition, as part of the nearshore injury assessment, the Trustees collected thousands of samples of coastal wetland soils, nearshore shallow water sediments, and plants and animals. These samples were chemically analyzed, specifically to determine TPAH50 concentrations. These results were also forensically analyzed to identify DWH oil (Emsbo-Mattingly & Martin 2015). This section summarizes chemistry and forensic analysis findings. More detailed discussion is found in Section 4.6, Nearshore Marine Ecosystem. Concentrations of TPAH50 in nearshore water were also evaluated, but they are summarized earlier in this section; the implications of this exposure are discussed in Section 4.4, Water Column.

4.2.6.2.2 Chemical Results in the Nearshore Environment

DWH oil was detected in coastal wetland soils, beaches, sediments adjacent to beaches, wetlands, SAV, and tissues of SAV and nearshore animals. TPAH50 concentrations in coastal wetland soils were most significantly elevated along shorelines where oiling was observed and generally mirrored the oil exposure categories that were based on visual observations of oiling (Nixon & Michel 2015). This was especially true for mainland salt marshes in Louisiana. Offshore of Louisiana mainland salt marshes, sediment TPAH50 concentrations were generally elevated along oiled shorelines, especially for sediments within 160 feet (50 meters) of shore (Zhang et al. 2015a). Such correlations between elevated coastal marsh soil TPAH50 concentrations and the adjacent shallow water sediments were generally not seen in the 2011 Mississippi and Alabama sampling surveys (Zhang et al. 2015a).

Biological tissue PAH concentrations often did not display clear trends in association with observed oiling (Oehrig et al. 2015), although high concentrations in SAV environmental samples were found adjacent to the most heavily oiled areas of the Chandeleur Islands (Cosentino-Manning et al. 2015). PAH contamination in biological tissue samples may not correlate with results in sediment or water due to differences in how these contaminants are retained and metabolized in different animals. Greater specificity on spatial and temporal chemistry findings in the nearshore environment is provided in Section 4.6 (Nearshore Marine Ecosystem). Some notable highlights of the results in vegetated coastal wetlands, in shallow subtidal sediments adjacent to marshes and beaches, and in nearshore biological tissues are provided in the following sections.

Vegetated Coastal Wetland Shorelines (Marshes and Adjacent Shallow Subtidal Areas)

The Trustees collected soil samples in coastal wetlands across Louisiana, Mississippi, and Alabama from 2010 to 2013. Sampling locations represented the full range of shoreline oil exposure categories based on survey data.

For Louisiana mainland marshes, TPAH50 concentrations in marsh soil samples collected along oiled shorelines in fall 2010 were orders of magnitude higher than baseline concentrations. This contrast is especially apparent along heavier persistent and heavier oiling shorelines and along the seaward marsh edges. TPAH50 concentrations generally mirrored observations of shoreline oiling and the resulting shoreline oiling categories described in previous sections and by Nixon and Michel (2015). Along heavier persistent oiling marshes, TPAH50 concentrations remained elevated through 2013, the last year measured. Along more lightly oiled shorelines, concentrations generally decreased from 2010 to 2013 (Zhang et al. 2015a).

In other Louisiana coastal wetland habitats and in Mississippi and Alabama, soil TPAH50 concentrations tended to increase with increasing shoreline oiling categories, and concentrations decreased over time. These trends were less defined, however, than those observed in Louisiana mainland salt marshes, because of the more patchy distribution of DWH oil in these regions (Zhang et al. 2015b).

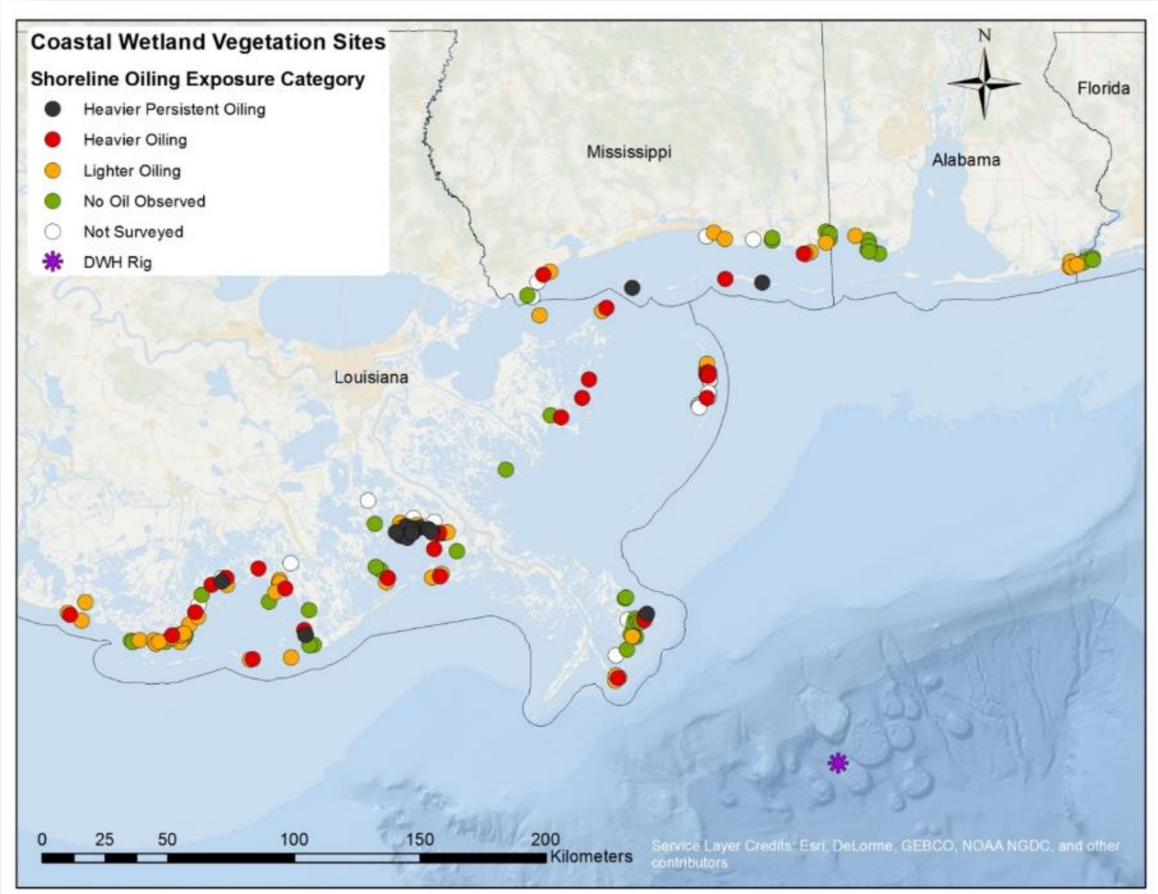
TPAH50 soil concentrations in Louisiana mainland herbaceous marsh sites were first measured in fall 2010. The average soil concentration in the most heavily exposed areas adjacent to the shoreline exceeded 127 milligram per kilogram (mg/kg) (ppm). Concentrations were not as high in sites further inland from the coast, but the average TPAH50 concentration for the most heavily oiled inland areas in 2010 was still above 15 mg/kg (ppm). In contrast, TPAH50 concentrations in soil samples taken from marshes where "no oil was observed" generally averaged under 0.5 mg/kg (ppm).

Weighted average TPAH50 concentrations across the various wetland types and oiling categories are provided in Table 4.2-1. Corresponding sampling sites and their observed oiling categories are shown in Figure 4.2-27.

Table 4.2-1. Soil TPAH50 average concentrations in Louisiana mainland salt marshes in zone 1 of coastal wetland vegetation sites. Concentrations along oiled shorelines were orders of magnitude higher than concentrations measured at "no oil observed" sites (Zhang et al. 2015b).

State	Habitat	Shoreline Exposure	Season: Average TPAH50 Concentrations (ppb)[a] Fall 2010	Spring 2011	Fall 2011	Fall 2012	Fall 2013
Louisiana	Mainland Herbaceous Salt Marsh	HEAVIER PERSISTENT OILING	127,558	130,351	95,842	126,834	31,920
		HEAVIER OILING	12,398	4,875	2,552	520	260
		LIGHTER OILING	4,477	2,707	681	1,785	674
		NO OIL OBSERVED	394	556	527	711	918
	Back-Barrier Herbaceous Salt Marsh	HEAVIER OILING	8,695	36,285	884	486	NA
		LIGHTER OILING	43	26	37	NA	NA
		NO OIL OBSERVED	33	41	59	15	NA
	Coastal Mangrove Marsh	HEAVIER PERSISTENT OILING	1,065	1,623	353	NA	NA
		HEAVIER OILING	966	658	555	682	841
		LIGHTER OILING	311	337	304	594	70
		NO OIL OBSERVED	711	343	270	200	371
	Delta Phragmites Marsh	HEAVIER PERSISTENT OILING	281	896	529	48	NA
		HEAVIER OILING	1,128	1,233	377	913	NA
		LIGHTER OILING	1,350	1,229	1,223	1,415	NA
		NO OIL OBSERVED	1,763	3,690	2,004	4,991	NA
Mississippi/ Alabama—Mississippi Sound	Mainland Herbaceous Salt Marsh	HEAVIER OILING	NA	362	89	851	NA
		LIGHTER OILING	NA	283	408	161	212
		NO OIL OBSERVED	NA	202	277	258	57
	Island Herbaceous Salt Marsh	HEAVIER PERSISTENT OILING	NA	446	415	730	782
		HEAVIER OILING	NA	11	4	8	1
		NO OIL OBSERVED	NA	71	130	NA	NA

[a] Average soil TPAH50 concentrations are weighted to account for stratified random sampling and preferential analysis of samples indicating likely presence of oil. See Table 4.6-8 of Zhang et al. (2015b) for standard error associated with these average values.



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

Figure 4.2-27. Observed degrees of oil exposure at coastal wetland locations sampled and analyzed for TPAH50. Concentrations in shallow water sediments just offshore from these sites tended to correlate with the oil exposures in the adjacent marshes.

Shallow Subtidal Sediments Adjacent to Marshes and Unvegetated Shorelines (Primarily Beaches)

The Trustees collected samples from sediments beneath the water in the nearshore environment in 2010 and 2011. For sediment 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" (Zhang et al. 2015a). This pattern is especially strong for sediments within 50 meters of shore. In 2011, concentrations in the areas where surveys found the heaviest oiling caused by the DWH incident were two to three times higher than ambient concentrations. In 2010, concentrations in the heavier and lighter oiling areas were significantly higher than those in the heavier persistent oiling areas. This seemingly contradictory finding may be explained by the timing of sample collection: samples were collected before the heaviest oil washed ashore in the heavier persistent oiling areas of Barataria Bay (Zhang et al. 2015a).

In 2010 and 2011, the Trustees also collected sediment samples adjacent to unvegetated shorelines, primarily beaches, in Florida, Alabama, Mississippi, and Louisiana. No relationship was apparent between TPAH50 concentrations in shallow water sediments and the degree of oiling on adjacent unvegetated shoreline. Average TPAH50 concentrations in these sediments were as high as 4.8 mg/kg (ppm) in some areas and below 0.01 mg/kg (ppm) in other areas (Zhang et al. 2015a).

TPAH50 concentrations in sediment samples collected offshore from Louisiana mainland marshes in 2010 generally corresponded to the degrees of oiling found in the adjacent marshes (Zhang et al. 2015a). Also, sediment contamination was highest within the first 50 meters from the shoreline. For instance, the average sediment TPAH50 concentration was highest within 50 meters of more heavily oiled marshes (5.4 mg/kg [ppm]). TPAH50 sediment concentrations offshore from lightly oiled marshes or marshes where no oil was observed were generally below 0.1 mg/kg (ppm, Zhang et al. 2015a).

Sediment samples were also collected in seagrass beds surrounding the Chandeleur Islands before (May to July 2010) and after (August to September 2010) oil reached the beds. Concentrations of sediment TPAH50 were eight to 12 times greater, on average, than ambient conditions (Cosentino-Manning et al. 2015).

Of special note for exposure to DWH oil adjacent to unvegetated shorelines was the documented presence of SOMs just offshore in certain regions from Louisiana to the Florida Panhandle (Pensacola Bay). Along the more heavily oiled sand beaches in Florida, Alabama, and the offshore barrier islands of Mississippi, some of the oil/sand mixture accumulated in the nearshore subtidal, forming weathered SOMs mostly between the toe of the beach and the first offshore bar (e.g., Figure 4.2-26). These mats were repeatedly buried and then re-exposed by sand migration. The SOMs became chronic sources of tar balls on the adjacent shoreline as they broke up (Michel et al. 2013).

Along the Louisiana barrier islands, oil/sand mixtures accumulated on portions of the lowermost intertidal zone, particularly where erosion exposed former marsh habitat. The oil/sand residues adhered to these surfaces, forming mats that were up to 100 meters long and 20 centimeters thick. These mats were only exposed during the lowest of tides and/or were buried by beach accretion, making it difficult to delineate and remove them. These mats were also chronic sources of tar balls on the adjacent beaches, as described above (Michel et al. 2013; Stout & Emsbo-Mattingly 2015).

Nearshore Biological Tissues

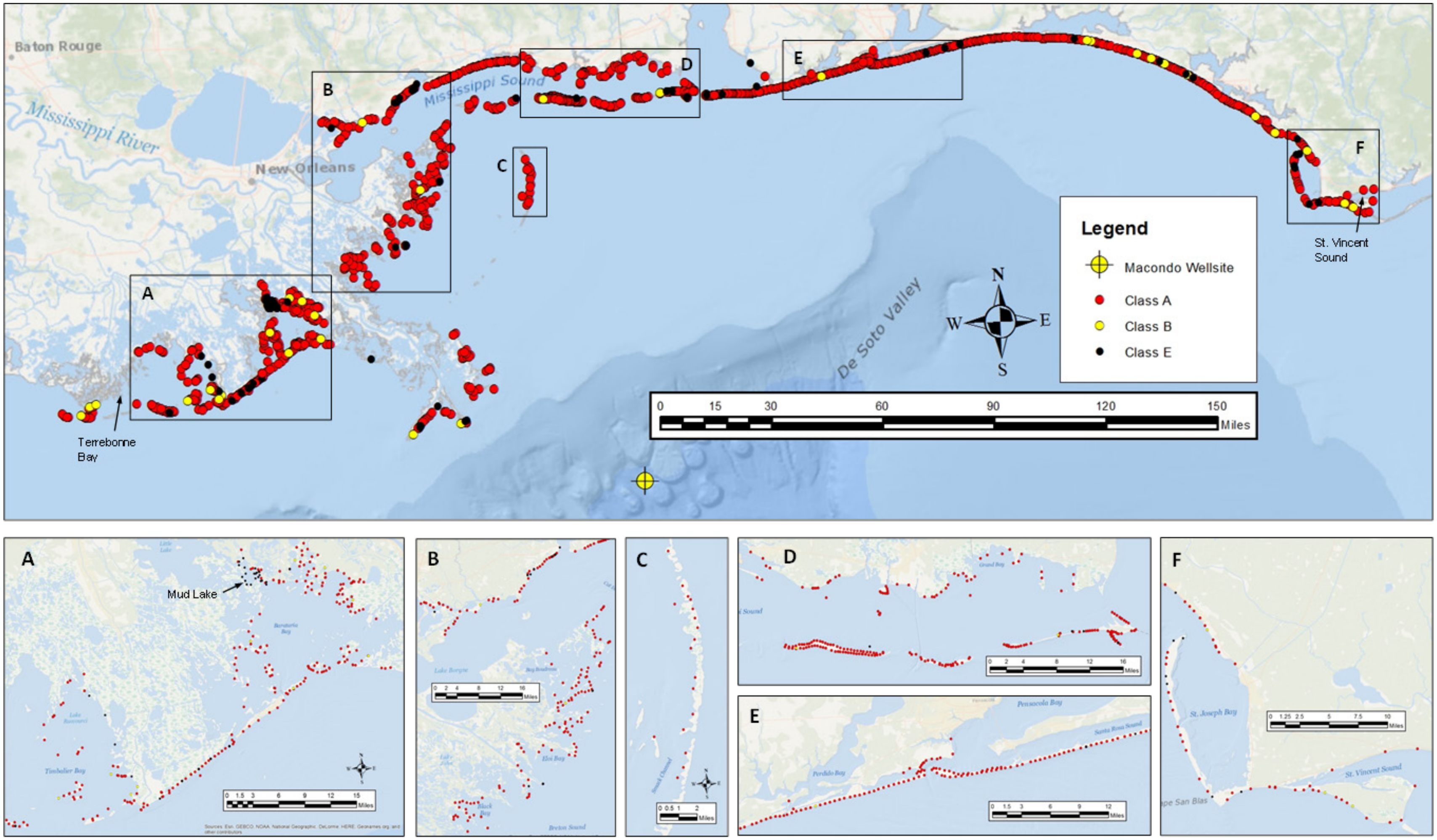
A limited number of nearshore biota samples were collected for PAH analysis offshore of coastal wetlands and beaches, in SAV beds, and in oyster reefs (Oehrig et al. 2015). The focus of several sampling efforts was on fish, invertebrates within the sediment (e.g., polychaete worms and amphipods), crustaceans (e.g., penaeid shrimp, blue crabs), bivalves, and oysters. The PAH chemistry data were highly variable; clear relationships generally were not found between PAH concentrations in biota and presence of DWH oil in sediments (Emsbo-Mattingly & Martin 2015; Oehrig et al. 2015). Such a relationship was found, however, in the case of SAV beds (Cosentino-Manning et al. 2015). TPAH50 concentrations in invertebrate tissue samples collected in the oiled SAV beds of the Chandeleur Islands in 2010 were over 400 times higher than the pre-spill baseline. Concentrations in SAV tissue were 13 times higher than baseline. Concentrations in June 2011 continued to be higher than pre-spill baseline conditions (Cosentino-Manning et al. 2015).

4.2.6.3 Forensic Analysis of Nearshore Samples

Forensic analysis for DWH oil presence in nearshore environmental samples (e.g., stranded oil, soils, sediments, water, and biota) found widespread distribution of DWH-derived oil along the northern Gulf of Mexico coast. The Trustees conducted forensic analyses of thousands of stranded oil, soil, sediment, and tissue samples collected in the nearshore environment. Certain samples, primarily sediments and soils, were collected seasonally for up to 4 years after the oil spill. These analyses confirmed the widespread exposure of the nearshore environment to DWH oil. The analysis of more than 1,300 stranded oil samples (usually in the form of tar balls, tar patties, and mats) indicated the presence of DWH oil from western Terrebonne Bay, Louisiana, to St. Vincent Sound, Florida (Stout 2015g) (Figure 4.2-28).

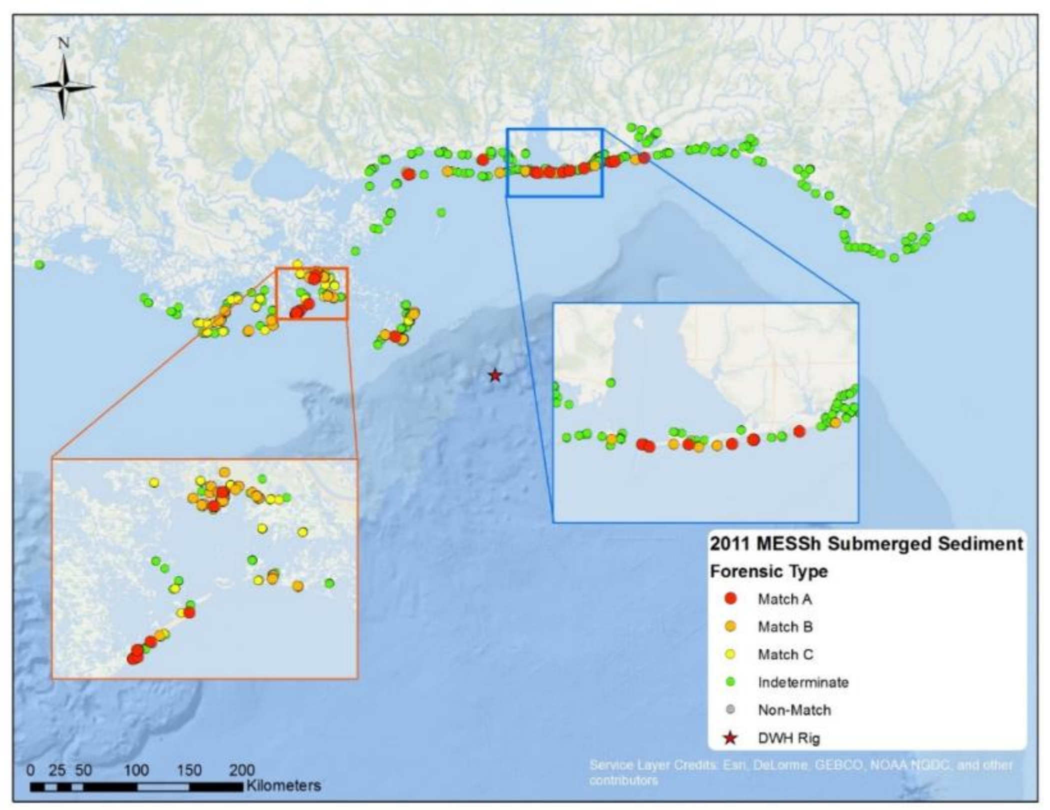
In addition, a total of 107 stranded oil samples were collected on Texas shorelines between July 5 and September 9, 2010, from Corpus Christi to the Texas/Louisiana border. Samples were collected as part of response efforts by the USCG, assisted by the Texas General Land Office, and contractors representing BP. Sample results indicated the presence of DWH oil from Galveston Island to the McFaddin National Wildlife Refuge near the Louisiana border (Unified Command for DWH Incident Command Post 2010).

More than 5,500 coastal wetland soil, nearshore sediment, and nearshore tissue samples also underwent forensic analysis (Emsbo-Mattingly & Martin 2015). These evaluations again revealed widespread distribution of DWH oil (Figure 4.2-29 and Figure 4.2-30). In addition, nest materials (e.g., sticks) collected at three osprey nests from Horn Island (Gulf Islands National Seashore) also contained DWH oil, indicating exposure of the osprey to DWH oil (Stout & Litman 2015c).



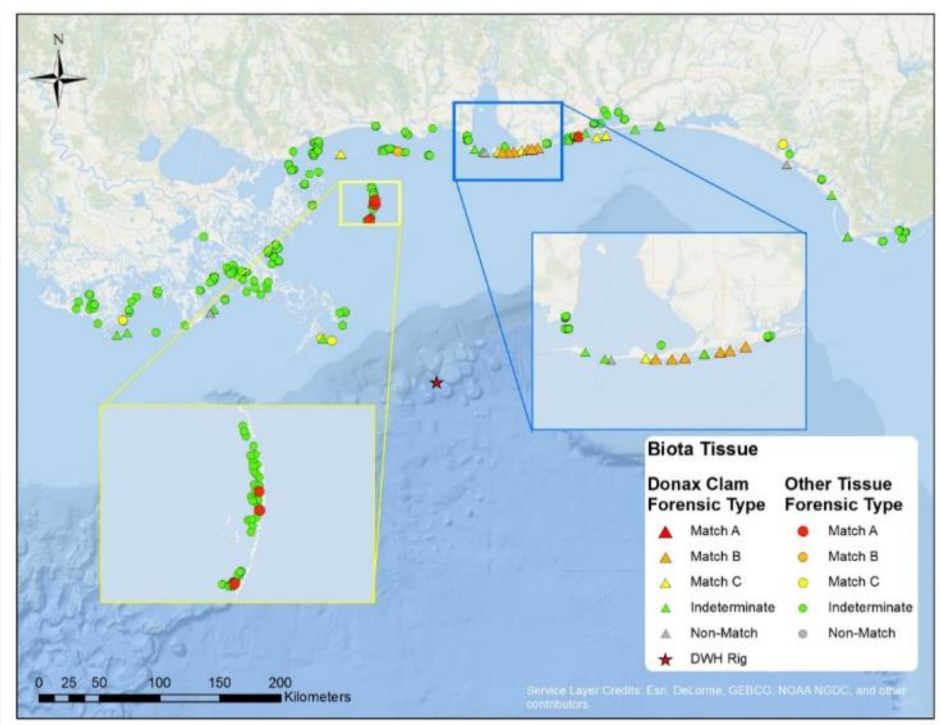
Source: Results from Stout (2015g).

Figure 4.2-28. Maps showing the spatial extent of stranded oils from supratidal and intertidal zones collected between May 24 and November 14, 2010, derived from Macondo oil (i.e., Class A and B; n=1188 and 31, respectively) and from non-Macondo oil (i.e., Class E; n=76) as determined by chemical fingerprinting. (A) Timbalier and Barataria Bays area, (B) Eastern delta area, (C) Chandelier Island, (D) Gulf Islands National Seashore area, (E) Pensacola Bay area, and (F) St. Vincent Sound area.



***Source**: Emsbo-Mattingly and Martin (2015).*

Figure 4.2-29. Locations of nearshore sediment samples collected along the northern Gulf of Mexico that matched the DWH oil fingerprint. Match A and Match B both refer to samples that matched the DWH fingerprint.



Source: *Emsbo-Mattingly and Martin (2015).*

Figure 4.2-30. Locations of nearshore tissue samples collected along the northern Gulf of Mexico that matched the DWH oil fingerprint. Match A and Match B both refer to samples that matched the DWH fingerprint.

4.2.7 Conclusions

Table 4.2-2 summarizes this section's findings regarding exposures of natural resources to DWH oil, dispersants, and synthetic-based drilling mud. The table cross-references the habitats, pathways, exposures and resource groups, and the subsequent Final PDARP/PEIS sections addressing each. Complete pathways from sources to exposed animals, plants, and habitats have been demonstrated through observations, empirical data, and modeling. These pathways link the widespread exposures to oil and dispersants to the release and responses to the DWH oil spill.

Based on the data presented in this section, the Trustees conclude the following:

- The DWH disaster released 3.19 million barrels (134 million gallons) of oil and 1.84 million gallons of dispersant into the environment.
- Every day for 87 days, the Macondo well released an average of nearly 1.5 million gallons of oil into the ocean. This essentially created a massive new oil spill every day for nearly 3 months.

- Combining direct observations, remote sensing data, field sampling data, and other lines of evidence, the Trustees documented that oil spread across an ocean surface about the size of the state of Virginia, washed onto at least 2,100 kilometers (1,300 miles) of shoreline, sank onto at least 400 square miles of sea floor sediments, and was transported as small oil droplets and dissolved chemicals within deep ocean water currents hundreds of miles away from the failed well.
- Natural resources were exposed to oil and dispersants across a broad range of habitats, including the deep sea; more than 1,500 vertical meters (5,000 vertical feet) of water column; the sea surface; and nearshore habitats, such as beach, marsh, mangrove, and SAV.
- Evidence of exposure includes numerous observations and collections of animals from within surface oil slicks and collections of carcasses confirmed to be contaminated with DWH oil. Biological tissues also contained elevated concentrations of compounds derived from the DWH oil.
- Despite natural weathering processes over the 5 years since the spill, oil persists in some habitats where it continues to expose resources in the northern Gulf of Mexico.

Table 4.2-2. Inventory of pathways, exposures, and resources in different habitat zones. Details available in subsequent Final PDARP/PEIS sections indicated.

Habitat	Transport Pathways	Contaminants	Resource Groups	Chapter 4 Sections
Deep-Sea, Slope, and Shelf	Direct fallout around wellhead	oil, subsea injected dispersant, and drilling mud in sediment	benthic sediments and biota	4.2.3
	Direct deposition due to impingement of deep-sea plume particulate due to bathymetry	oil with or without dispersant in sediment		
	Sinking marine oil snow originating at/near sea surface or within deep-sea plume	oil-containing flocculent with or without dispersant in sediment		
	Fallout of in situ burn residue and particulates	burn residue within water column or sediment		
	Dissolved and particulate oil within deep-sea plume	deep-sea plume containing dissolved gas and soluble components of oil, dispersant	water column and biota	4.2.4
Rising Plume	Ascending buoyant oil and (limited) gas plume	dissolved and particulate oil with or without subsea injected dispersant in water column		
Surface and Near-Surface	Floating oil slick, sheen, mousse	floating oil with or without dispersant	surface water and biota	4.2.5
	Uppermost water column (less than 10 meters below surface)	dissolved and entrained particulate oil with or without dispersant in water column		
	Oiled *Sargassum*	floating oil with or without dispersant		

Habitat	Transport Pathways	Contaminants	Resource Groups	Chapter 4 Sections
	Fallout of in situ burn residue and particulates	floating burn residue or settling particulates		
Nearshore/ Onshore	Floating oil slick, sheen, mousse	floating oil with or without dispersant		
	Stranded oil mats, coatings, tar balls	stranded oil with or without dispersant	shoreline soils, sediments and biota	4.2.6
	Sunken oil (SOMs)	sediment-oil mixtures		
	Resuspended sediment	oil in sediment		
Air	Evaporation of floating oil	volatile chemicals in oil	air-breathing biota	4.2.5
	Formation of aerosols of floating oil	micro-droplets of floating oil		
	Formation of combustion-derived particulates from in situ burning	soot		

4.2.8 References

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4.3 Toxicity

What Is in This Section?

- **Executive Summary**
- **Introduction (Section 4.3.1):** What is toxicity? What is known about the toxic effects of oil?
- **Approach to the Assessment (Section 4.3.2):** How did the Trustees assess the toxicity of *Deepwater Horizon* (DWH) oil?
- **Toxic Effects of DWH Oil (Section 4.3.3):** Summary of the results of the Trustees' toxicity testing program.
  - What are the effects of DWH oil to water column resources?
  - What are the effects of DWH oil on benthic invertebrates?
  - What are the effects of DWH oil on nearshore resources?
  - What are the effects of DWH oil on birds?
  - What are the effects of DWH oil on sea turtles and marine mammals?
- **Conclusions (Section 4.3.4)**
- **References (Section 4.3.5)**

Executive Summary

The Trustees designed and implemented a comprehensive program to evaluate the toxic effects of DWH oil on natural resources of the northern Gulf of Mexico. This program entailed performing a series of controlled laboratory studies that were designed to support the Trustees' resource and habitat-specific injury assessments. Through this comprehensive toxicity testing program, the Trustees created a body of information that greatly expands on the scientific literature available prior to the spill and provides an unprecedentedly large, coherent dataset from which conclusions about injury could be drawn.

Overall, the Trustees found that exposure to DWH oil causes a wide range of toxic effects to natural resources, including death, impaired reproduction, disease, and other physiological malfunctions that reduce the ability of organisms to survive and thrive. Measured and modeled concentrations of DWH oil in surface water and sediments in the Gulf of Mexico at a number of locations and times during and following the spill exceeded concentrations at which the Trustees documented toxicological effects from oil exposure in the laboratory.

As part of the evaluation of toxicity to **water column resources**, the Trustees exposed fish and invertebrates (both offshore and nearshore species) to DWH oil mixed into water and in surface sheens. The results of Trustee studies demonstrated that the embryos and larvae (i.e., early life stage) of fish (ichthyoplankton) and various stages of pelagic invertebrates (zooplankton) are particularly susceptible to the toxic effects of DWH oil. Measured and modeled concentrations of DWH oil in the Gulf of Mexico exceeded lethal levels in a number of locations and times during and following the spill. Thin, rainbow

sheens of surface slick oil were also found to be lethal to fish embryos and invertebrates. Oil is known to be more toxic in the presence of natural sunlight. The Trustees determined that DWH oil was roughly 10 to 100 times more toxic to semi-transparent invertebrates and early life-stage fish when exposed to ultraviolet (UV) light, an effect that is of particular concern for fish embryos and larvae at or near the surface of the water column. In addition to lethality, exposure to DWH oil also caused developmental abnormalities, growth inhibition, immunosuppression, and decreased swim performance. The lethal effect of DWH oil on the embryos and larvae of fish and several life stages of invertebrates has important ecological implications. In addition to sustaining fish and invertebrate populations, these small, planktonic organisms are an important base of the marine food web.

The evaluation of toxicity to **benthic resources** involved tests in which bottom-dwelling invertebrates were exposed to sediments contaminated with DWH oil. Testing demonstrated that exposure of amphipods to contaminated sediments resulted in mortality at concentrations observed in nearshore and deep-sea sediments following the spill.

The **nearshore resources** toxicity testing program involved studies on species selected to represent injury to the marsh faunal community and to serve as an overall indicator of adverse effects on nearshore marsh habitats. In addition to evaluating toxicity to water column resources described above, the Trustees evaluated the following: nearshore fish and invertebrate species exposed to contaminated sediment, nearshore species exposed to combinations of water and suspended sediment, and nearshore species exposed to contaminated marsh soil and vegetation.

Exposure of marsh organisms to sediments contaminated with DWH oil resulted in a series of adverse effects, including death, reduced growth, and reduced reproductive success. Higher concentrations of total polycyclic aromatic hydrocarbons (TPAH50; the sum of 50 individual PAHs measured) in sediments resulted in more severe adverse effects in more test species. Adverse effects were observed at concentrations as low as approximately 1 milligram of TPAH50 per kilogram of sediment (mg/kg). Toxicological responses to DWH oil observed in Trustee studies included damage to gill and liver tissues, reduced growth rates, and mortality in flounder; growth inhibition in juvenile red drum and Pacific white shrimp; reduced reproduction and survival in Gulf killifish; mortality to fiddler crab offspring exposed to relatively low concentrations of oil in or on sediments, when followed by exposure to sunlight; and increased mortality and an impaired ability to move away from oil in marsh periwinkles. Exposure to DWH oil caused adverse effects in all oyster life stages tested, at varying concentrations.

The Trustees performed studies on the effects of ingesting DWH oil to freshwater turtle species that could serve as surrogates for **sea turtles**. Test animals that ingested DWH oil exhibited alterations in multiple toxicity endpoints, such as oxidative damage, and DNA damage. The Trustees also observed evidence of dehydration, decreased digestive function, and poor absorption of nutrients.

In studies with cell lines and an experimental surrogate organism for **marine mammals**, exposure to DWH oil also was found to cause problems with the regulation of stress hormone secretion from adrenal cells and kidney cells (Gulf toadfish). Impacts such as these on the endocrine system will affect an animal's ability to regulate body functions and respond appropriately to stressful situations and will lead to reduced fitness.

There was a considerable degree of consistency among the types of toxic responses observed across the different organisms tested in the Trustees' toxicity testing program. For example, effects observed by the Trustees in laboratory studies using DWH oil included cardiac effects in both fish and birds; disruption of blood cells and function in fish and birds; evidence of oxidative damage in fish, birds, and turtles; and impairment of immune system function in fish and birds. Evidence of impairment to stress responses and adrenal function was also observed in fish, birds, and mammalian cells tested in the laboratory. Evidence of similar physiological impairments were observed in a number of different types of organisms in the wild that were exposed to DWH oil.

4.3.1 Introduction

Oil is known to be toxic to organisms. However, the toxicity of different oils can vary, and both the toxic effects and the concentrations of oil at which those effects occur can differ across species and types of exposures. To understand the toxic effects of DWH oil exposure on the natural resources of the northern Gulf of Mexico, the Trustees conducted a series of controlled laboratory studies. These studies contributed to the Trustees' understanding of the range of natural resource injuries that occurred as a result of the spill, and informed the injury determination and injury quantification findings that are presented in the remainder of Chapter 4.

Section 4.3 describes the rationale (Section 4.3.1), design (Section 4.3.2), and results (Section 4.3.3) of the Trustees' toxicity testing program.

4.3.1.1 What Is Toxicity?

Toxicity refers to the nature and degree to which a chemical substance is poisonous to organisms. The toxicity of oil has been studied extensively, through both laboratory investigations and in response to oil spills (e.g., Douben 2003; Kingston 2002; Leighton 1993; Peterson 2001; Reynaud & Deschaux 2006; Teal & Howarth 1984). In many hundreds of published reports, oil has been shown to be toxic to fish (e.g., de Soysa et al. 2012; Hemmer et al. 2010), invertebrates (e.g., Baussant et al. 2011; Hannam et al. 2010), birds (e.g., Balseiro et al. 2005; Stubblefield et al. 1995), mammals (e.g., Bowyer et al. 2003; Duffy et al. 1994), reptiles (e.g., Fritts & McGehee 1989; Lutcavage et al. 1995), plants (e.g., DeLaune et al. 2003; Ibemesim & Bamidele 2008), plankton (Bender et al. 1977; Gardiner et al. 2013), and bacteria (Fuller et al. 2004; Hodson et al. 1977; Suarez-Suarez et al. 2011). Some of the toxic effects of oil include death (Aurand & Coelho 2005; Perkins et al. 2005); reduced growth rates (Barron et al. 1999; Cajaraville et al. 1992; Scarlett et al. 2007); impacts on tissues, such as lesions in the liver, skin, or elsewhere (e.g., Khan 2013; Lipscomb et al. 1993); developmental abnormalities and cardiac damage (e.g., de Soysa et al. 2012; Hatlen et al. 2010); reproductive impairment (Baussant et al. 2011; Truscott et al. 1992); immune effects, which can increase susceptibility to disease (Hannam et al. 2010; Payne & Fancey 1989); and cancer (Hawkins et al. 1990; Suchanek 1993).

Many of the original studies of oil toxicity focused on lethal effects to older organisms. More recently, scientists have found that oil causes a much wider variety of toxic effects, and that the early life stages of many animals (e.g., embryos and larvae for fish; eggs for birds and reptiles) are particularly sensitive to the toxic effects of oil (e.g., Carls et al. 1999; Colavecchia et al. 2004; Couillard & Leighton 1991; Finch et al. 2011). In aquatic habitats, this can have an especially significant effect on the ecosystem, since, in addition to sustaining populations of fish and invertebrates, the food web depends upon early life-stage fish and other plankton (see Chapter 3).

When describing toxicity, scientists often refer to "lethal" and "sublethal" effects (Rand 1995). Lethal toxicity occurs when exposure to a chemical results in observable mortality (death) to an exposed animal. Sublethal toxicity refers to effects that do not result directly in observable death. However, sublethal toxic effects can shorten the life expectancy of organisms by reducing their overall health or "fitness." For example, animals whose fitness is compromised by sublethal toxic effects may 1) have more difficulty finding prey or avoiding predators, 2) exhibit greater susceptibility to disease, 3) demonstrate a reduced ability to tolerate natural stresses (such as elevated temperatures or reduced dissolved oxygen in water), or 4) have more difficulties reproducing. In the wild, organisms whose fitness is compromised are more likely to die (Rice 2014).

In addition to its chemical toxic effects, oil can also harm organisms through physical fouling (Fowler et al. 1995; Hurst et al. 1991; Pezeshki et al. 2000). Fouling refers to the physical coating of oil on an organism and is not strictly the same as "toxicity" because it does not involve the chemical interaction of oil compounds with physiological processes. Nonetheless, fouling can be lethal to organisms and can cause a range of effects, such as smothering (lack of oxygen), clogging tissues (e.g., eyes, nasal cavities), losing insulation from feathers (which can result in death from hypothermia), or impairing movement (inefficient flight or swimming). Fouling resulted in mortality to birds and turtles and other taxa (see Section 4.7, Birds, and Section 4.8, Sea Turtles).

4.3.1.2 What Factors Influence the Toxic Effects of Oil?

A number of different factors can influence the toxicity of oil. Animals can be affected in different ways and at different oil concentrations, depending on their physiology, behavior, and life history. For example, the effects of oil on fish, which "breathe" through the passage of water over their gills, can differ from the effects of oil on dolphins, which, like humans, have lungs and must surface to breathe air. Some of the factors that influence the toxic effects of oil and were considered by the Trustees are described below. Section 4.3.2, Approach to the Assessment, discusses how these different factors were addressed in the Trustees' toxicity testing program.

4.3.1.2.1 Species and Life Stage

The toxic effects of oil and the concentrations at which those effects occur can vary considerably between species. These variations can be caused by differences in anatomy and physiology, metabolism, and how an organism is exposed to a chemical (e.g., gill exposure, inhalation, or ingestion) (Rand 1995). The Trustees evaluated toxicity across a suite of representative organisms that utilize the different habitats of the northern Gulf of Mexico, including animals that live in the open ocean and in nearshore waters, animals that live in and on bottom sediments, and animals that rely on important shoreline habitats such as intertidal marshes. In addition, the Trustees considered the life stage and life history of organisms that were exposed in these habitats by including evaluations of the toxicity of oil to embryonic, juvenile, and adult life stages.

4.3.1.2.2 Chemical Composition of Oil

As described in Section 4.2 (Exposure), oil is a complex mixture of chemicals, and this chemical mixture, or composition, can change in the environment through the process of weathering (Morris et al. 2015c). Although the toxicity of oil results from exposure to this complex chemical mixture, many scientists evaluate oil toxicity based on the concentration of polycyclic aromatic hydrocarbons (PAHs), a group of chemical compounds that are known to be among the most toxic components of oil (Box 1). The Trustees generally have employed this convention in evaluating laboratory and field data, but recognize that oil toxicity is caused by a more complex mixture of chemicals.

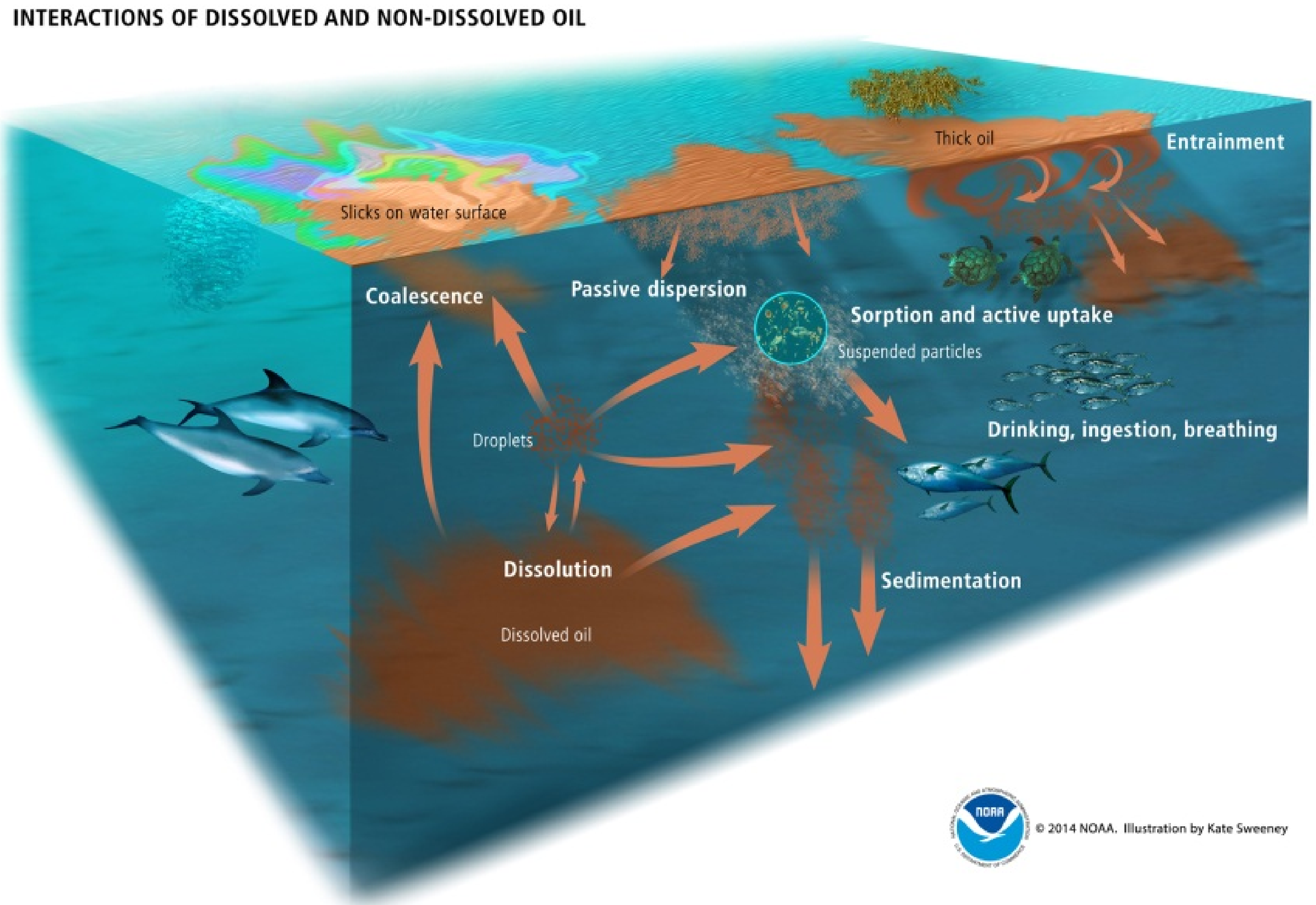
Because it is such a complex mixture of chemicals, the toxicity of different oils can vary, and the toxicity of a single type of oil (such as DWH oil) can change based on weathering and other environmental factors (NRC 2003). The Trustees evaluated the toxicity of DWH oils that were collected from the environment after the spill, including the relative toxicity of different field-collected oils that covered a range of weathering conditions (Morris et al. 2015c).

4.3.1.2.3 Oil Mixing, Dispersion, and Partitioning

Aquatic organisms in the Gulf of Mexico were exposed to DWH oil that was present as slicks floating on the surface of the water, oil that was mixed with and dissolved into water, and oil that was associated with sediments and organic material (Figure 4.3-1). The toxic effects of oil in water can be influenced by how the oil mixes and dissolves into water, and how it partitions between small droplets, water, and sediment (Payne et al. 2003). Oil-water dispersions can be natural, such as the formation of droplets of oil through physical action (e.g., wave action or the violent release of DWH oil from the well) (Delvigne & Sweeney 1988), or can be created by application of dispersant chemicals (NRC 2005).

Box 1. Why Are PAHs Important for Toxicity?

Oil is composed of thousands of chemical compounds, and the composition of the oil can affect its toxicity. PAHs are some of the most toxic components of oil and have been the subject of extensive study over the past several decades (Adams et al. 2014; Billiard et al. 2008; Douben 2003; NRC 2003). Toxicologists often represent the toxicity of different oils in terms of the summed concentrations of many individual PAHs (e.g., EPA 2003; Incardona et al. 2014; NOAA 1991). Although this measure should not imply that PAHs are the only toxic components of oil, the Trustees largely have employed this convention in their chemical analyses of samples from the field and laboratory, particularly when evaluating potential toxicity. The specific measurement adopted in the Natural Resource Damage Assessment (NRDA) describes concentrations in terms of the sum of 50 individual PAH compounds (H.P. Forth et al. 2015a). This value is referred to as total PAH50 (referring to the 50 individual PAHs measured), or TPAH50.



Source: *Kate Sweeney for NOAA.*

Figure 4.3-1. Natural resources of the Gulf of Mexico were exposed to oil floating on the water surface, oil mixed into the water through dispersion and natural mixing/dissolution processes, and suspended and bottom sediments.

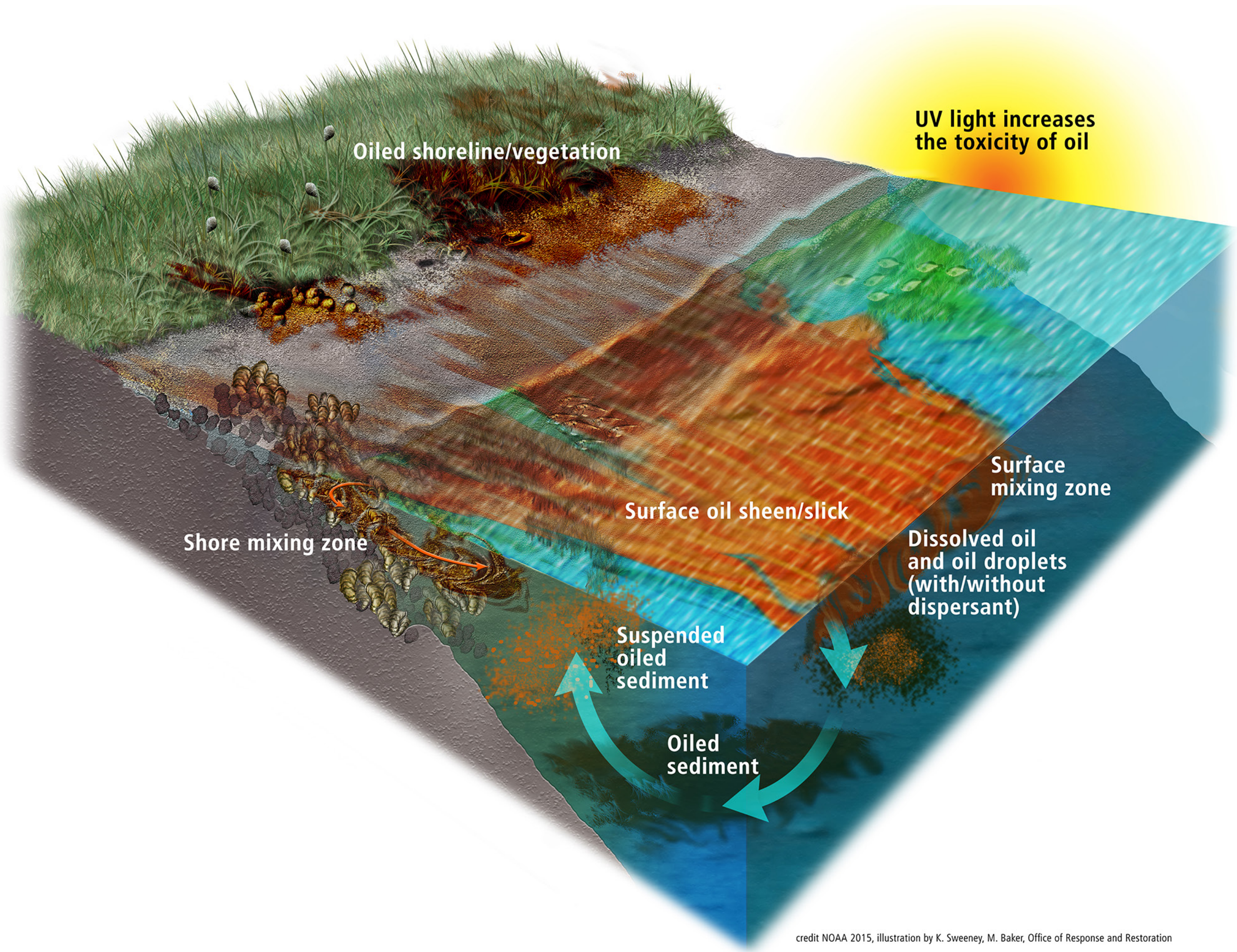
Chemical oil dispersants are mixtures of solvents (like paint thinner), surfactants (like soaps), and other additives that are applied to oil slicks to break up the slick and mix the oil into the water column (Section 4.2, Natural Resource Exposure). Although dispersants are used to reduce the amount of thick, floating oil that can reach sensitive shoreline habitats, adding dispersants can increase the toxicity of oil to aquatic organisms by mixing more of it into the water column (NRC 2005). The Trustees' toxicity evaluation considered these different types of exposures to aquatic resources by evaluating the toxicity of different oil-water mixtures, as well as the toxicity of oil with added chemical dispersants (H.P. Forth et al. 2015b; Morris et al. 2015b). Dispersants were used during the spill at depth near the wellhead and in offshore waters (see Section 4.2, Exposure).

4.3.1.2.4 Sunlight (Photo-Induced Toxicity)

When organisms are exposed to oil, the toxicity of the oil can increase substantially in the presence of natural sunlight (Oris & Giesy 1985; Sellin Jeffries et al. 2013). Known as photo-induced toxicity, this occurs because some PAH compounds in oil absorb UV light from the sun and produce toxic byproducts that can damage DNA, cell membranes, and other tissues (Arfsten et al. 1996). This reaction occurs in transparent tissues, such as gills, and in transparent organisms (Arfsten et al. 1996; Oris & Giesy 1985), such as the embryos and larvae of many fish and shellfish. The Trustees evaluated the toxicity of DWH oil with and without UV light to represent the range of conditions that occur in the environment.

4.3.1.2.5 Exposure Pathways

The natural resources of the northern Gulf of Mexico were exposed to oil across a wide variety of habitats and through a number of different exposure pathways (Section 4.2). Plants and animals were exposed to oil floating on the ocean surface, mixed in the water column, mixed in sediments and marsh soils, and on plants (Figure 4.3-2). As described in greater detail in the following sections of Chapter 4, animals were exposed to oil through breathing (including exposure through gills and lungs), inhalation and aspiration of oil into lungs, drinking and incidental intake of water, ingestion of sediment and food, and through physical contact (Figure 4.3-3). The Trustees designed studies to consider these different pathways and types of exposure.



Source: *Kate Sweeney for NOAA.*

Figure 4.3-2. The Trustees' toxicity testing evaluated the effects of DWH oil across a variety of habitats.

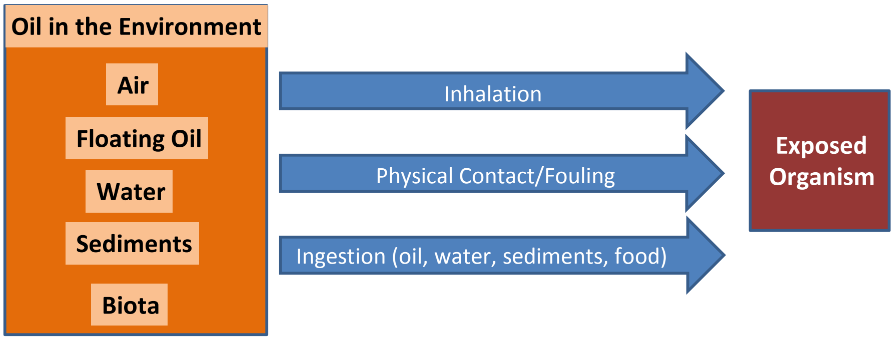


Figure 4.3-3. Potential pathways of exposure of organisms to DWH oil. Animals may be exposed through one or more of these pathways.

4.3.2 Approach to the Assessment

The Trustees' assessment approach involved performing a series of controlled laboratory studies that were designed to support the Trustees' resource and habitat-specific injury determination and quantification. As illustrated in Figure 4.3-4, the Trustees' toxicity testing program comprised studies designed to evaluate the toxicity of DWH oil for the following resource categories and exposure pathways in the injury assessment:

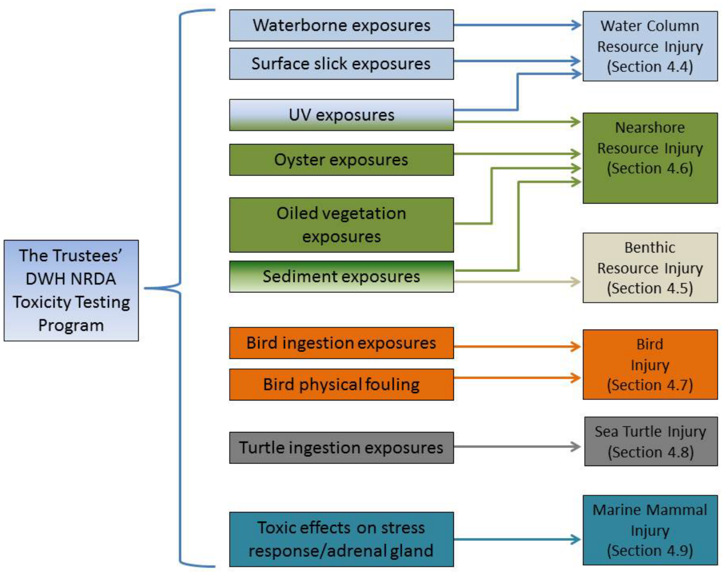


Figure 4.3-4. Overview of the Trustees' toxicity testing program showing the relationship between laboratory testing and resource-specific injury assessment.

- **Water column resources (Section 4.4)**, including fish and invertebrates found in offshore and nearshore areas. As part of the evaluation of toxicity to water column resources, the Trustees exposed fish and invertebrates to DWH oil mixed into water and in surface slicks.
- **Benthic resources (Section 4.5 and 4.6)**, specifically benthic invertebrates. The Trustees exposed bottom-dwelling invertebrates to sediments contaminated with DWH oil. The Trustees also evaluated the toxicity of oil to other bottom-dwelling organisms, including fish, oysters, and crustaceans, as part of the nearshore resource toxicity testing work.
- **Nearshore resources (Section 4.6)**, including fish, crustaceans and other invertebrates, oysters, and snails. The Trustees exposed resources that live in nearshore habitats to DWH oil in sediments and on marsh vegetation.

- **Birds (Section 4.7)**. The Trustees performed focused studies to evaluate the toxicity of DWH oil to birds when ingested and when externally exposed on feathers.
- **Sea turtles (Section 4.8)**. The Trustees performed studies on the effects of ingesting DWH oil to turtle species that could serve as surrogates for sea turtles.
- **Marine mammals (Section 4.9)**. To support evaluation of field data collected from bottlenose dolphins, the Trustees performed limited and focused laboratory studies using a mammalian cell line culture to evaluate the effects of DWH oil on stress response and adrenal gland function.

The Trustees designed their toxicity testing program to investigate the nature and extent of different types of adverse impacts to a variety of organisms based on observed, measured, and modeled exposure to oil and dispersants. The testing program was designed to address different types of exposures to DWH oil (e.g., exposure to weathered oil, dispersed oil, oil-water mixtures, surface slicks, and sediments), different environmental variables that can influence toxicity (primarily UV light), different test species, different life stages, and a series of different lethal and sublethal effect endpoints (Figure 4.3-5). To address the role of weathering on toxicity, a range of weathered DWH oils were used in laboratory tests (Box 2). Through this comprehensive toxicity testing program, the Trustees created a body of information that greatly expands on the scientific literature available before the spill and provides an unprecedentedly large, coherent dataset from which conclusions about injury could be drawn.

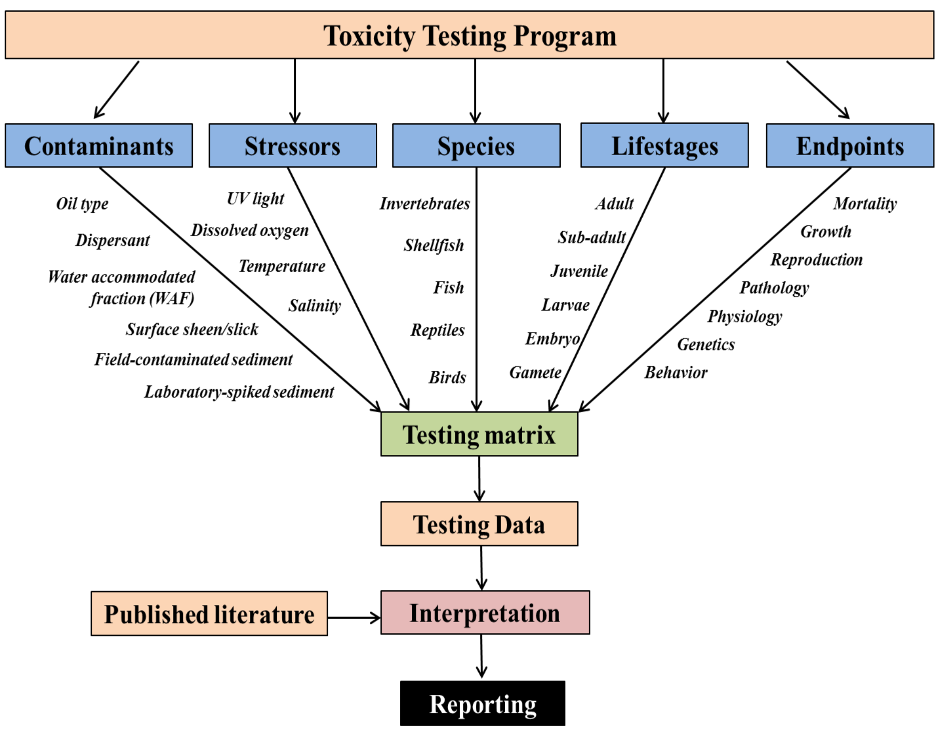


Figure 4.3-5. The Trustees' toxicity testing program was designed to evaluate different types of exposure to DWH oil, different environmental variables, a variety of Gulf of Mexico species and life stages, and a series of lethal and sublethal toxicological effects endpoints.

Box 2: What Types of Oil Were Used in the Toxicity Tests?

To better understand the influence of weathering on toxicity, four DWH oil samples at varying degrees of weathering were included in the toxicity testing program (H.P. Forth et al. 2015b). The percent depletion of TPAH relative to a stable marker compound, hopane, is used as an indicator of the relative degree of weathering and is generally presented as a percent (Morris et al. 2015c). The four oils tested were:

- "Source oil": DWH (BP's Macondo well) riser oil collected on July 26, 2010. This oil was almost entirely unweathered (8 percent weathered).
- "Artificially weathered oil": Source oil heated in a laboratory, removing the lightest components of the oil (e.g., volatile components like BTEX), to represent a slightly weathered oil (27 percent weathered).
- "Slick A": Surface slick oil collected on July 29, 2010, from the hold of a barge that was receiving oil from various skimmer vessels responding to the spill, therefore representing a natural degree of weathering that occurred in the Gulf environment (68 percent weathered).
- "Slick B": Surface slick oil collected on July 19, 2010, by a skimmer vessel near the Mississippi River Delta. This oil represents a higher degree of weathering that occurred naturally in the environment (85 percent weathered).

The Trustees' toxicity testing program evaluated a variety of species and included a series of laboratory tests (or "bioassays").

- **Test species.** Because it is impractical to test every species of organism that was exposed to DWH oil, the Trustees' testing program focused on using representative Gulf of Mexico species. The Trustees selected species for testing that are native to the northern Gulf of Mexico, could serve as example species from which generalizable inferences can be drawn (i.e., species whose physiology and life histories are representative of many species, or that occur in multiple habitats), could be tested in a laboratory setting, and play important or unique roles in the Gulf of Mexico ecosystem and/or economy. In some cases, tests were conducted with organisms closely related to Gulf of Mexico species. The Trustees conducted these tests using surrogate species to support evaluation of the toxicity of DWH oil to animals that cannot be tested in the laboratory (e.g., endangered sea turtles), or to gain a more mechanistic understanding of the toxic effects of DWH oil to facilitate broader inferences across different types of organisms. As shown in Table 4.3-1, the Trustees' toxicity testing program included 21 different species of fish and 12 species of invertebrates, as well as phytoplankton, freshwater turtles (as surrogates for sea turtles), and four different species of birds. The table also identifies the type(s) of exposures used in evaluating toxicity for each test species.
- **Toxicity testing procedures: bioassays.** Scientists typically evaluate the toxicity of environmental samples or chemicals by exposing test organisms to a range of concentrations under controlled conditions (Rand 1995). Such tests are often referred to as bioassays. Data generated from these tests are used by scientists to determine the types of adverse effects that

occur at different oil concentrations, under specific exposure conditions. The way that scientists often evaluate these adverse effects concentrations is through the determination of "dose-response relationships," which show the relationship between the concentration of a toxicant and the degree of adverse effects (Box 3). To simplify comparative analysis across bioassays, scientists refer to the toxic concentrations of a contaminant that cause a specific amount of some adverse effect, such as the concentration of oil that causes mortality to 50 percent of the test organisms exposed in a given study. These effects levels are known as "lethal concentration" (LC), or more broadly, "effects concentration" (EC) values. Using a standardized effects level, such as an LC50 or an LC20 (that is, the concentration of a chemical that causes mortality to 50 percent or 20 percent of the test organisms, respectively), enables easier comparison between studies. For example, the relative sensitivity of two different species may be evaluated by comparing their respective LC50s or LC20s. It should be emphasized, however, that these ECs do not represent "thresholds" for the onset of toxicity.

Table 4.3-1. Species and exposures included in the Trustees' toxicity testing program.

Species	Scientific Name	Exposure				
		WAF[a]/ Surface Slick	Sediment/ Oiled Substrate	Dietary	Dermal	UV[d]
Fish						
Atlantic croaker	*Micropogonias undulates*	X				
Bay anchovy	*Anchoa mitchilli*	X				X
Cobia	*Rachycentron canadum*	X				
Gulf killifish	*Fundulus grandis*	X	X			X
Gulf menhaden	*Brevoortia patronus*	X				
Gulf toadfish	*Opsanus beta*	X				
Inland silverside	*Menidia beryllina*	X				
Mahi-mahi	*Coryphaena hippurus*	X				X
Pacific bluefin tuna[b]	*Thunnus orientalis*	X				
Pacific mackerel[b]	*Scomber japonicus*	X				
Red drum	*Sciaenops ocellatus*	X	X			X
Red snapper	*Lutjanus campechanus*	X				X
Sand seatrout	*Cynoscion arenarius*	X				
Sheepshead minnow	*Cyprinidon variegatus*	X				X
Shovelnose sturgeon[b]	*Scaphirhynchus platorynchus*	X				
Southern bluefin tuna[b]	*Thunnus maccoyii*	X				
Southern flounder	*Paralichthys lethostigma*	X	X			
Speckled sea trout[c]	*Cynoscion nebulosus*	X				X
Yellowfin tuna	*Thunnus albacares*	X				
Yellowtail amberjack[b]	*Seriola lalandi*	X				
Zebrafish[b]	*Danio rerio*	X				

Species	Scientific Name	Exposure				
		WAF[a]/ Surface Slick	Sediment/ Oiled Substrate	Dietary	Dermal	UV[d]
Invertebrates						
Amphipod	*Leptocheirus plumulosus*		X			
Blue crab	*Callinectes sapidus*	X	X			X
Brown shrimp	*Farfantepenaeus aztecus*		X			
Copepod	*Acartia tonsa*	X				X
Eastern oyster	*Crassostrea virginica*	X	X	X		X
Fiddler crab	*Uca longisignalis*	X	X			X
Fiddler crab	*Uca minax*		X			
Grass shrimp	*Palaemonetes pugio*	X	X			X
Marsh periwinkle	*Littoraria irrorata*		X			
Mysid shrimp	*Americamysis bahia*	X				X
White shrimp	*Litopenaeus setiferus*	X	X			
Pacific white shrimp[b]	*Litopenaeus vannamei*	X	X			X
Phytoplankton						
Diatom	*Skeletonema costatum*	X				
Reptile						
Common snapping turtle[b]	*Chelydra serpentina*			X		
Red-eared slider[b]	*Trachemys scripta elegans*			X		
Birds						
Double-crested cormorant	*Phalacocorax auritus*			X	X	
Homing pigeon[b]	*Columba livia*			X	X	
Laughing gull	*Leucophaeus atricilla*			X		
Western sandpiper	*Calidris mauri*			X	X	

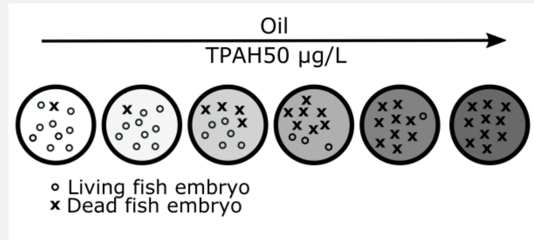
[a] Water accommodated fraction.

[b] Surrogate species.

[c] Many common names for this species, including spotted seatrout.

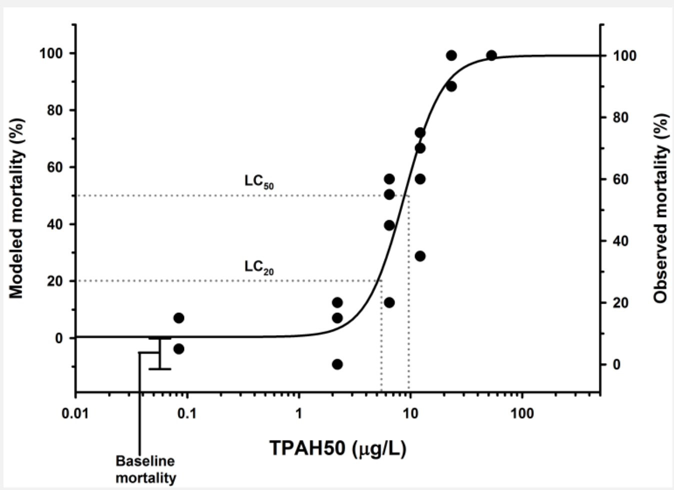
[d] UV light exposure in addition to oil exposure.

Box 3: What Is a Toxicity Test and How Is Toxicity Measured?



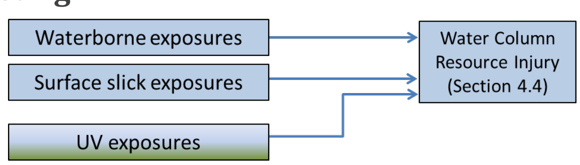
A toxicity test, or bioassay, is a controlled study in which the lethal or sublethal effects of a chemical are evaluated in test organisms (Rand 1995). One common approach to toxicity testing is to place a similar number of organisms into replicate exposure chambers (such as tanks or beakers) that have different concentrations of oil. An unoiled "control" treatment is used to quantify the measured effect under the conditions of the test, but without oil. After exposing the organisms to oil for a given period of time, scientists determine the frequency or degree of adverse effects, such as the number of living and dead organisms in each exposure chamber.

Data from a toxicity test then are compiled to evaluate the dose-response relationship between the exposure concentration, such as the concentration of TPAH50 in water, and the degree of effect, such as the percent of test organisms killed by the exposure.



The figure to the left shows an example of a dose-response curve derived from one of the Trustees' toxicity tests (Morris et al. 2015b). For this toxicity test, mahi-mahi embryos were exposed to increasing concentrations of oil, shown in terms of the TPAH50 concentration in micrograms of oil per liter of water (µg/L). The dose-response curve shows the relationship between TPAH50 and mortality based on the experimental data, illustrating that greater concentrations of TPAH caused higher mortality rates. LC50 and LC20 values are shown as the points on the horizontal axis that are associated with 50 percent and 20 percent mortality.

4.3.2.1 Water Column Resource Toxicity Testing



The water column resource toxicity testing program focused on evaluating the toxicity of DWH oil—with and without added dispersant—to fish, invertebrates, and diatoms, a kind of phytoplankton (see Table 4.3-1). Test organisms were exposed to oil mixed in water and as surface slicks. Because of the important role that natural sunlight plays in enhancing the toxicity of PAH, extensive testing was performed to evaluate the degree of this photo-induced toxicity.

Toxicity tests included studies to determine the concentrations of oil that kill organisms, as well as studies to determine concentrations of oil that cause adverse effects on the health or viability of organisms. In addition to mortality, the Trustees' toxicity tests documented the following adverse effects on health and viability:

- Impaired early life-stage growth and development (Brown-Peterson et al. 2015; Incardona et al. 2014; Incardona et al. 2013; Lay et al. 2015b; Morris et al. 2015b).
- Impaired reproductive success (Morris et al. 2015b; Vignier et al. [In Press]).
- Impaired cardiac development and function (Bursian et al. 2015b; Bursian et al. 2015c; Dorr et al. 2015; Incardona et al. 2014; Incardona et al. 2013; Morris et al. 2015b).
- Reduced immune system function and increased susceptibility to disease (Morris et al. 2015b; Ortell et al. 2015).
- Biochemical, cellular, and genetic alterations, and adverse changes to organ tissue (Brown-Peterson et al. 2015; Bursian et al. 2015b; Bursian et al. 2015c; Dorr et al. 2015; Morris et al. 2015b; Takeshita et al. 2015).

For purposes of the summary provided in this Final PDARP/PEIS, the Trustees' focus is primarily on the results of tests incorporated directly in the resource-specific injury determination and quantification presented in the following sections of this chapter. More detailed information is provided in related technical appendices and publications contained in the References section.

Prior to initiating the testing program, the Trustees developed a standardized set of methods and procedures to ensure consistency across the testing laboratories, and to ensure that quality assurance and quality control (QA/QC) was maintained. These efforts included using the same oil samples for testing, creating exposure solutions using the same methods, reviewing testing plans and procedures prior to conducting each test, collecting data in a standardized format, and implementing a detailed data validation and verification process (Morris et al. 2015b).

Toxicity tests with water column organisms included bioassays performed with oil-water mixtures and tests in which organisms were exposed to surface slicks. For oil-water mixtures, the Trustees developed a suite of methods to achieve a range of chemical concentrations and compositions similar to the range of conditions that organisms would have encountered in the ocean, but in a standardized manner that could be easily replicated by different laboratories conducting toxicity studies (Box 4). By using four different oils (Box 2) and three different mixing methods, the Trustees were able to develop test solutions that captured variability in conditions encountered in the environment and bolstered our understanding of oil toxicity across that range of conditions (H.P. Forth et al. 2015b; Incardona et al. 2013).

Box 4: Producing Oil-Water Mixtures for Toxicity Testing

During the DWH spill, oil mixed with water in the environment through a variety of ways, including high energy mixing near the wellhead, wave action at the ocean surface, and application of chemical dispersants. While toxicity testing laboratory procedures do not need to mimic how oil mixed into water in the environment, toxicologists do want to test solutions with similar chemical compositions and concentrations to those found in the environment. Three methods were used to create oil-water mixtures, known as water accommodated fractions (WAFs): a low-energy mixing procedure (LEWAF); a high-energy mixing procedure (HEWAF); and a medium-energy, chemically enhanced mixing procedure (CEWAF) (H.P. Forth et al. 2015b; Morris et al. 2015b). Researchers have used LEWAFs and CEWAFs for many years (Aurand & Coelho 2005; H.P. Forth et al. 2015b; Singer et al. 2000), and these mixing methods produce solutions with chemical compositions similar to those measured in the field (H.P. Forth et al. 2015b). Although HEWAFs have also been used in the field of petroleum toxicology for many years (Aurand & Coelho 2005; Echols et al. 2015; Girling 1989; Incardona et al. 2014), the Trustees developed a standardized blender-mixing method to achieve better consistency across test solutions and laboratories (H.P. Forth et al. 2015b; Incardona et al. 2014; Morris et al. 2015b). Although DWH oil did not mix into the ocean like a blender, the method was used to prepare reproducible oil-water mixtures with chemical compositions and concentrations that were also similar to those measured in the environment (H.P. Forth et al. 2015b).

In addition to tests with oil mixed into water, the Trustees also conducted a series of toxicity tests on different fish and invertebrate species using very thin surface slicks or sheens similar to the surface slicks that covered large expanses of the Gulf of Mexico during and following the spill (Figure 4.3-6).

Source: *Abt Associates (top), NOAA (bottom).*

Figure 4.3-6. *Top*: Thin oil sheen, about 1 micron (µm) in thickness—see H.P. Forth et al. (2015)—generated in a beaker using DWH oil (Slick A). This oil, visible as a slightly reflective sheen on the water surface, was used to conduct bioassays with fish and invertebrates. *Bottom*: DWH oil sheen in the northern Gulf of Mexico, photographed from an airplane.

4.3.2.1.1 Photo-Induced Toxicity Testing

As described previously, photo-induced toxicity can increase the toxicity of oil (Alloy et al. 2015; Oris & Giesy 1985; Sellin Jeffries et al. 2013). This reaction is important because, in the Gulf of Mexico, many developing fish and invertebrates live in the upper water column near the surface of the ocean where they are exposed to UV light (see Section 4.4).

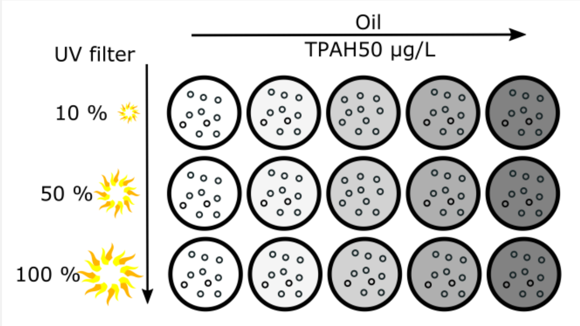
The Trustees evaluated the UV photo-induced toxicity of DWH oil in a number of controlled tests using both natural sunlight and artificial lights that produced UV exposures similar to those encountered in nature (Box 5).

Box 5: What Are Photo-Induced Toxicity Tests?

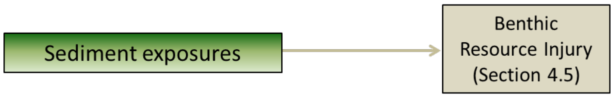
The upper 15 to 30 meters of the water column in the northern Gulf of Mexico are clear enough that UV light penetrates at intensities sufficient to cause photo-induced toxicity (Lay et al. 2015a). When UV light is absorbed by PAHs in the tissues of semi-transparent organisms, damaging byproducts are produced that can destroy tissues and cells. Thus, during the DWH oil spill, semi-transparent organisms that came into contact with oil and UV in both offshore and nearshore environments would be susceptible to photo-induced toxicity. The Trustees carried out a series of toxicity tests to determine how much more toxic DWH oil became in the presence of UV light.

A typical bioassay included 2 to 8 hours of exposure to waterborne (WAF) or surface slick oil followed by or in conjunction with 6 to 9 hours of exposure to outdoor sunlight or UV light generated with special light bulbs in a laboratory. Filters were used to vary the amount of UV exposure to test organisms.

The amount of sunshine can vary from day to day, depending on weather and time of day, during summers in the northern Gulf of Mexico. Therefore, researchers measured the amount of UV light exposure experienced by organisms during each test (Lay et al. 2015b) for comparison to field measurements to enable quantification of photo-induced toxic effects in the environment.

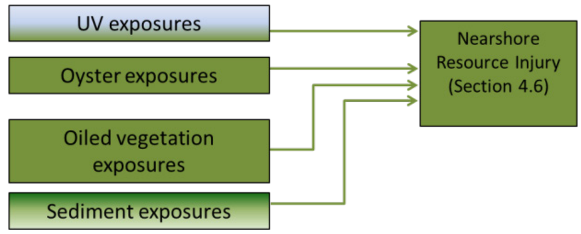


4.3.2.2 Benthic Resource Toxicity Testing



The benthic resource toxicity testing focused on evaluating the toxicity of sediments contaminated with DWH oil to bottom-dwelling invertebrates (Table 4.3-1). Extensive areas of bottom sediment were contaminated with DWH oil in nearshore and deep-sea habitats (see Section 4.2, Exposure, and Section 4.5, Benthic Resources). Therefore, bioassays used contaminated sediment collected from coastal areas of the northern Gulf of Mexico that experienced heavy oiling and added varying amounts of DWH oil to clean sediments collected from the field (Krasnec et al. 2015a). Additionally, BP conducted bioassays using contaminated sediment collected from deep-sea habitats, which are discussed in Section 4.3.3.2 (Krasnec et al. 2015b). In addition to being used for the benthic resource injury assessment, sediment toxicity testing was also used in the nearshore resource assessment, discussed below.

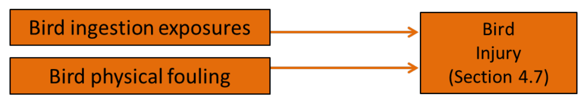
4.3.2.3 Nearshore Resource Toxicity Testing



As part of the nearshore resource injury assessment (Section 4.6), the Trustees studied the effects of DWH oiling on marsh species (Table 4.3-1) that were selected to represent injury to the marsh faunal community more broadly, and as an overall indicator of adverse effects on nearshore marsh habitats.

In addition to evaluating toxicity to water column resources (Section 4.3.2.1), the Trustees evaluated nearshore species exposed to contaminated sediment, combinations of water and suspended sediment, and contaminated marsh soil and vegetation. The nearshore studies included evaluating the toxicity of DWH oil to Gulf sturgeon because this species migrates through nearshore areas when they move from salt water into coastal rivers to spawn. Nearshore studies also included tests to evaluate UV effects.

4.3.2.4 Bird Toxicity Testing



The Trustees conducted a series of focused laboratory toxicity studies to evaluate the toxicity of ingested oil, and to evaluate the implications of physical fouling of feathers.

4.3.2.4.1 Evaluating Effects of Ingested Oil

DWH response workers documented the presence of thousands of dead birds during and after the spill (see Section 4.7, Birds). Field surveys also documented the occurrence of thousands of oiled birds that were not sufficiently impaired to allow their capture and cleaning by oil spill responders (Section 4.7). When birds preen to clean oil off their feathers, they inevitably swallow oil. In order to assess the toxic effects of ingested oil on Gulf of Mexico bird species, the Trustees conducted a series of tests using representative and surrogate bird species, such as double-crested cormorants, spotted sandpipers, laughing gulls, and homing rock pigeons. Oil ingestion studies for gulls and cormorants were performed by injecting dead fish with DWH oil and feeding the contaminated fish to birds. As described later in this section, when birds ingested oil-contaminated fish, they suffered from a variety of adverse health effects, including anemia, liver dysfunction, kidney damage, hypothermia, weight loss, lethargy, abnormal feces, feather damage, heart abnormalities, moribundity (near death), and death (Bursian et al. 2015b; Bursian et al. 2015c; Dorr et al. 2015).

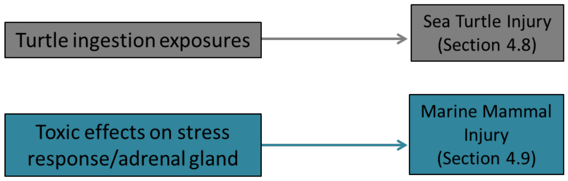
4.3.2.4.2 Evaluating Physical Fouling Effects of Oil

In addition to its chemical toxicity, the viscous, sticky nature of oil can adversely affect the ability of birds to take off, fly, and follow efficient flight paths. A wind tunnel was used to measure how oil on the body (not the wings) affects flight energetics and flight ability for western sandpipers (Maggini et al. 2015). The Trustees also employed high-speed video to determine how trace levels of oil on wing and tail feathers affects the speed and angle of a bird's takeoff movements, and how that affected flight energy costs.

To further investigate how DWH oil affects a bird's ability to fly in a natural environment, the Trustees assessed the effects of externally applied oil on the field-based flight performance of homing pigeons.

The Trustees compared the flight performance of individual homing pigeons before and after oiling over trips of 50, 85, and 100 miles (from the release site to their home loft). Studies also were designed to observe adverse health effects in oiled animals, including effects on body mass and effects on organ function (Pritsos et al. 2015).

4.3.2.5 Sea Turtles and Marine Mammals Toxicity Testing



The Trustees' injury assessment of sea turtles and marine mammals included very limited toxicity testing because these organisms are federally protected.

4.3.2.5.1 Sea Turtles

During the spill, response workers collected many oiled sea turtles from the Gulf of Mexico. Sea turtles had DWH oil covering their bodies and coating their esophagi; this physical fouling with heavy oil was associated with mortality (Section 4.8, Sea Turtles). To help investigators evaluate the toxicity of ingested oil on sea turtles, the Trustees conducted a limited series of tests with surrogate turtle species (red-eared sliders and snapping turtles), in which animals were exposed to DWH oil to emulate ingestion exposures. Testing focused on the evaluation of sublethal effects on health and fitness (Mitchelmore & Rowe 2015).

4.3.2.5.2 Marine Mammals

As described in Section 4.9 (Marine Mammals), field studies demonstrated that coastal bottlenose dolphins suffered from adrenal gland disease and dysfunction as a result of the DWH spill. As additional support in evaluating the physiological mechanisms associated with this injury, a limited series of laboratory studies were performed with mammalian adrenal cell lines to study cellular dysfunction (Takeshita et al. 2015). In addition, studies were performed with the Gulf toadfish to further evaluate the effects of DWH oil on the release of stress response hormones, which are related to adrenal function (Morris et al. 2015b).

4.3.3 Toxic Effects of DWH Oil

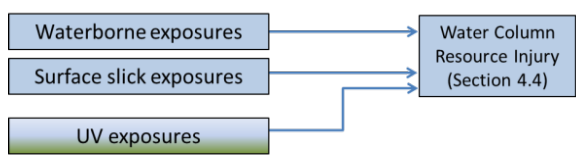
This section summarizes important findings from the Trustees' toxicity testing program. This summary emphasizes toxicity testing data used directly in the resource-specific sections of this chapter. A considerable amount of other supporting toxicity data were developed through the comprehensive testing program (see technical appendices and published manuscripts included in the References section). Although not all these data were included directly in the resource-specific injury determinations or quantifications, the data and findings informed the Trustees' decision-making regarding the nature and scope of injuries and will help inform future restoration planning efforts. Finally, the data developed through this program contribute substantially to the Trustees' understanding of the toxicological effects of oil and the susceptibility of Gulf of Mexico species to oil pollution.

4.3.3.1 Toxicity of DWH Oil to Water Column Resources

Key Findings

- Fish embryos and larvae and invertebrates are particularly susceptible to the toxic effects of DWH oil, both when mixed with water and when present in the form of a surface slick. Measured and modeled concentrations of DWH oil in the Gulf of Mexico exceeded lethal levels in a number of locations and times during and following the spill.
- Thin, rainbow sheens of surface slick were lethal to developing fish and invertebrates.
- DWH oil is roughly 10 to 100 times more toxic to invertebrates and developing fish in the presence of natural sunlight.
- In addition to lethality, exposure to DWH oil causes developmental abnormalities, including heart and spinal defects. Many of these developmental abnormalities are severe enough to kill early life-stage fish.
- Older fish (juveniles or adults) are less susceptible than embryos and larvae to lethal effects of DWH oil exposure. At most of the oil concentrations that occurred after the spill, the toxic effects of oil on older fish are more likely to have manifested as sublethal injuries, including growth inhibition, immunosuppression, decreased swim performance, and an abnormal stress response.
- The lethal effect of DWH oil on fish embryos and larvae and invertebrates has important ecological implications. In addition to sustaining fish and invertebrate populations, these small, planktonic organisms are an important base of the marine food web.

4.3.3.1.1 Lethality



Oil causes a range of adverse effects to organisms. Short-term lethality, sometimes referred to as "acute lethality" is the most severe of those effects. The Trustees conducted extensive laboratory testing to evaluate the concentrations of oil that cause mortality. Lethal responses were used for purposes of quantifying water column resource injuries (Section 4.4). However, sublethal toxicity occurs at lower oil concentrations than mortality, and the sublethal responses observed in the laboratory can result in reduced survival and reproduction in the wild. Consequently, lethality-based injury quantification underestimates the full scope of injuries.

Toxicity of Oil-Water Mixtures (WAF Tests)

The Trustees evaluated the lethal toxicity of DWH oil-water mixtures (WAF bioassays) to a large number of species of Gulf fish (or close surrogates) included as part of the water column resources assessment. Tested species included offshore, pelagic fish, such as mahi-mahi, tunas, and cobia, as well as fish that live along the continental shelf or in more nearshore waters, such as sea trout, red drum, menhaden, and bay anchovy (see Table 4.3-1). The Trustees also evaluated the toxicity of WAF exposure to shrimp and to several invertebrates (blue crabs, oysters, and fiddler crabs), the larvae of which occupy the water column during part of their developmental cycle. Testing was also performed with two species that make up part of the marine food web: a diatom phytoplankton and a copepod zooplankton.

As illustrated in Figure 4.3-7, which provides examples of WAF tests performed with embryos of mahi-mahi, red drum, and bay anchovy, the studies demonstrated a clear dose-response relationship between exposure to DWH oil and increased mortality rates. Table 4.3-2 summarizes LC20 values—concentrations that cause mortality in 20 percent of the test organisms—determined for water column fish species from these dose-response relationships. As shown in Table 4.3-2, LC20 values were as low as approximately 1 μg/L TPAH50, which can also be expressed as 1 part per billion (ppb) (Box 6). As discussed in greater detail below and in Section 4.4 (Water Column), many of these lethal ECs for early life-stage fish were exceeded in the environment during and following the spill in some locations and times.

Box 6: What Is a Part Per Billion?

Toxicologists typically report the results from tests involving oil-water mixtures in units of micrograms of oil or TPAH per liter of water (μg/L). 1 μg/L is the same as 1 part per billion (ppb). 1 ppb is roughly equal to thoroughly mixing *one drop* of ink (about 0.0017 liquid ounces) into a large tanker truck full of water (about 11,600 gallons).

For a given species and life stage, the toxicity of DWH oil to fish was generally similar across WAF preparation methods when toxicity is expressed in terms of the concentration of TPAH50 (Morris et al. 2015b). In our tests with oils at four different weathering states, the toxicity of the WAF, expressed in TPAH50, generally increased with increasing degree of weathering (Morris et al. 2015c).

Table 4.3-3 presents the results of toxicity tests with invertebrates in terms of LC20 values. These effects values were somewhat higher than were observed in the more sensitive early life-stage fish tests and any exceedances of these higher invertebrate toxicity values in the environment would have been less frequent.

As with fish, the toxicity of weathered DWH oil exceeded that of less weathered oils (Morris et al. 2015c). In invertebrates, the toxicity of CEWAFs (containing dispersant) often was greater (i.e., lower LC values) than in LEWAFs/HEWAFs (without dispersant; Table 4.3-3). In toxicity tests with dispersant alone, invertebrates tended to be somewhat more sensitive than fish (Box 7). Therefore, the dispersant itself may have contributed to the toxicity of CEWAFs in some invertebrate tests (Morris et al. 2015b). However, the cumulative surface area of the ocean over which dispersant was applied (305 square-mile [mi^2] days) (Houma 2010)[1] was only 0.06 percent of the cumulative area of surface oiling (475,000 mi^2 days; Section 4.4, Water Column).

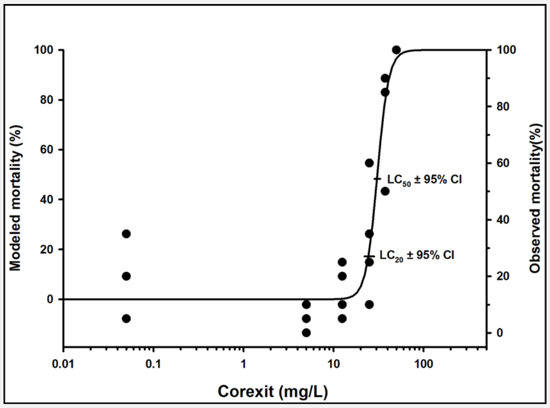
[1] A mi^2 day is a compound unit that means 1 square mile for 1 day, in any combination of area and time. For example, 100,000 mi^2 days could mean 1,000 mi^2 for 100 days, 10,000 mi^2 for 10 days, or 100,000 mi^2 for 1 day.

Box 7: Dispersant Toxicity

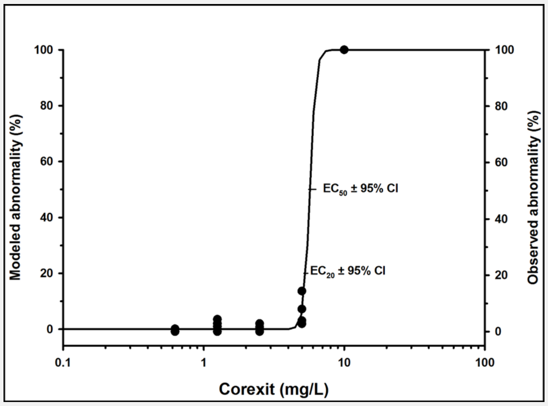
In addition to evaluating dispersant toxicity (Corexit 9500) in combination with oil in CEWAF tests, the Trustees performed bioassays with dispersant alone. Lethal effects concentrations for dispersant generally occurred in the parts-per-million (ppm, or mg/L) range of Corexit in water, whereas effects concentrations for TPAH50 were in the parts-per-billion (ppb, or µg/L) range (about 1,000 times lower).

For example, the left panel below shows the results of a test using larval mahi-mahi. In this test, the LC20 value was 25 ppm and the LC50 value was 31 ppm after 72 hours. The right panel in the figure below shows the results of a bioassay in which abnormal development in oyster larvae was measured. In this test, the EC20 and EC50 concentrations were 5.3 and 5.7 ppm respectively, after 24 hours, (Morris et al. 2015b).

It is possible that dispersant may have contributed to the observed toxicity in the Trustees' toxicity testing program CEWAF exposures to invertebrates (Morris et al. 2015b). However, the cumulative surface area of the ocean over which dispersant was applied was only 0.06 percent of the cumulative area of surface oiling (Section 4.4, Water Column). Consequently, any potential contribution from dispersant to total toxic effects would have been minimal relative to the injury caused by oil.



Effects of dispersant on survival of larval mahi-mahi.



Effects of dispersant on abnormal development of oyster larvae.

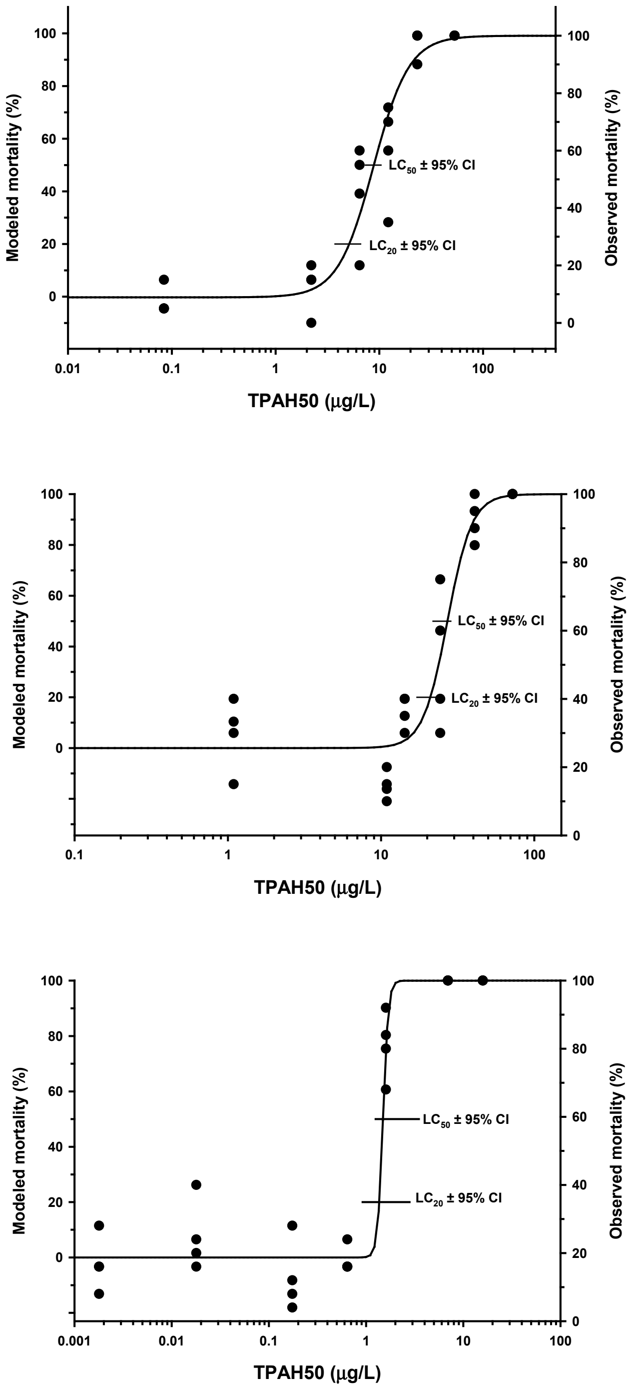


Figure 4.3-7. Results of HEWAF toxicity testing showing the relationship between the exposure concentration of TPAH50 and percent mortality (Morris et al. 2015b). **Top panel:** Mahi-mahi embryo/larvae exposed to Slick A oil. The LC20 for this test after 96 hours of exposure was 5.1 (95 percent confidence interval [CI] 3.7–6.6) µg/L TPAH50. **Middle panel:** Red drum embryo/larvae exposed to Slick A oil. The LC20 for this test after 72 hours was 21.9 (95 percent CI 18.4-24.5) µg/L TPAH50. **Bottom panel:** Bay anchovy embryo/larvae exposed to Slick B oil. The LC20 for this test after 48 hours was 1.3 (95 percent CI 0.9-2.8) µg/L TPAH50.

When evaluating the data presented in Table 4.3-2 and Table 4.3-3, it should be emphasized that the toxicity of DWH oil increased considerably in the presence of UV light, for both fish and invertebrates. Consequently, these LC20 values would be overestimated for organisms exposed to UV light at or near the surface of the water. Additional information about UV phototoxicity and the relationship to environmental conditions is presented below.

Table 4.3-2. Ranges of LC20 values observed in toxicity tests with early life-stage fish exposed to different DWH oils using different WAF preparation methods (Morris et al. 2015b).

Species	Test Duration (hours)	LC20 (µg/L TPAH50)
Bay anchovy	48	1.3–3.3
Cobia	96	17.3–27.5
Mahi-mahi	96	0.95–40.2
Red drum	60–72	7.0–21.7
Speckled sea trout	72	6.2–25.6
Yellowfin tuna	24	0.7

Table 4.3-3. Ranges of LC20 values observed in toxicity tests with invertebrates exposed to oil-water mixtures (e.g., Morris et al. 2015b). Invertebrates tended to be less sensitive to DWH oil than many of the early life-stage fish tested. However, invertebrates generally appeared to have greater sensitivity to dispersants than the early life-stage fish. Invertebrates typically were more sensitive to DWH oil and dispersant (CEWAF) than oil alone (HEWAF).

Species	Life Stage	Duration (hours)	LC20 µg/L TPAH50
CEWAF			
Blue crab	Zoea	96	12.7–12.9
White shrimp	Juvenile	96	50.9
Grass shrimp	Adult	96	73
HEWAF			
Blue crab	Zoea	48	56.8–105.1
Copepod	Adult	96	33.5
White shrimp	Juvenile	96	80.4

Section 4.4 (Water Column) describes how these dose-response relationships were used to quantify lethality injuries to fish and invertebrates.

Toxicity of Surface Slicks

DWH oil covered an extremely large area of the ocean during summer 2010. The embryos and larvae of many fish and many life stages of invertebrates live in the upper water column near the ocean surface where they could come into contact with floating oil. The Trustees performed toxicity tests in which early life stages of different fish species and mysid shrimp were held in water covered with very thin sheens of DWH oil (see Figure 4.3-6).

The results of these studies demonstrated that exposure to even thin sheens of oil were extremely toxic to developing fish. The longer organisms spent in contact with the oil sheen, the more likely they were to die. For example, red drum embryos exposed to sheens made with weathered oil (Slick A) for 24, 48, or 60 hours experienced an average of 34, 68, or 74 percent mortality in excess of controls, respectively (Figure 4.3-8). Similarly, bay anchovy exposed to sheens made with weathered oil (Slick A or Slick B) for 48 hours suffered 94 and 88 percent mortality, respectively (Morris et al. 2015a). These results demonstrate that direct exposure to the surface sheen and/or the water in close proximity to the sheen is highly toxic to early life-stage fish and invertebrates.

Photo-Induced Toxicity of DWH Oil

The Trustees evaluated the UV photo-induced toxicity of DWH oil in a number of controlled tests with fish and invertebrates that live in surface waters and are exposed to natural sunlight. Following a short exposure period (less than 8 hours) to relatively low concentrations of oil, semi-transparent organisms appear normal, but then die when they are exposed to UV light for short durations (less than 8 hours). Each of the species tested was susceptible to photo-induced toxicity when exposed to oil and UV light (Lay et al. 2015b). Toxicity in the presence of UV light was generally on the order of 10 to more than 100 times greater than toxicity without UV light. For example, the toxicity of waterborne DWH oil to speckled sea trout embryos/larvae was nearly 200 times greater in the presence of UV light than without UV (Figure 4.3-9).

As with photo-induced toxicity in WAF exposures, exposure to UV light also increased the toxicity of oil in thin surface sheens. Exposure to surface sheens and UV light resulted in very high mortality in tests with speckled sea trout embryos (70 percent mortality), juvenile mysid shrimp (97 percent mortality), and bay anchovy embryos (92 percent mortality).

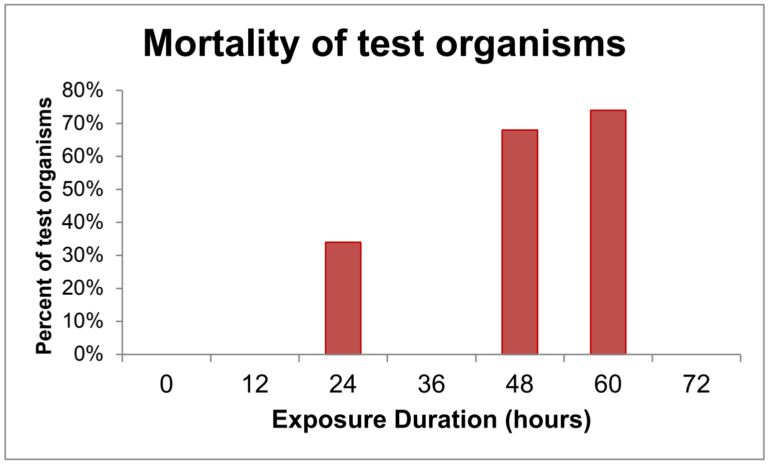


Figure 4.3-8. Percent mortality observed in red drum embryos following exposure in water with a thin sheen of floating DWH oil (Morris et al. 2015b).

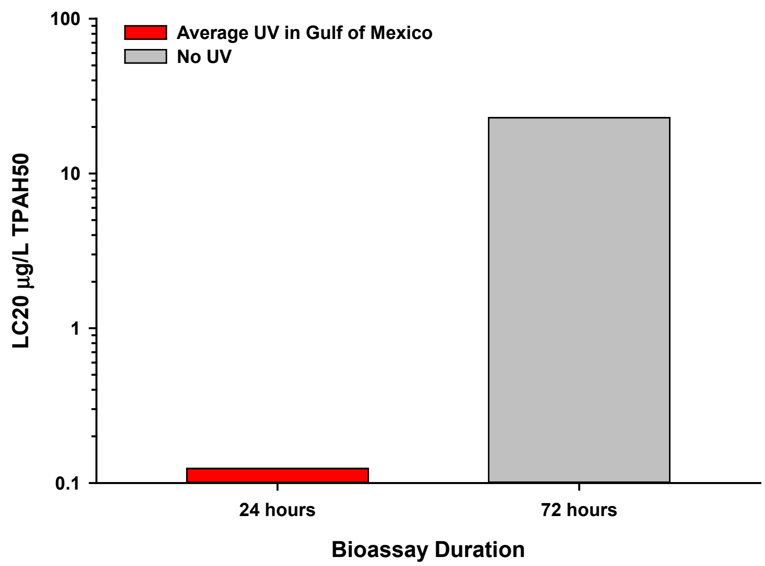


Figure 4.3-9. Comparison of lethal concentration (LC20) of DWH oil to early life-stage speckled sea trout in the presence of UV (red bar) at average UV levels recorded during the spill (Lay et al. 2015a) with bioassays conducted with embryos/larvae exposed to oil but no UV (gray bar) (Morris et al. 2015b). LC20 values for 72-hour exposures are generally lower than 24-hour exposures (i.e., more toxicity). However, the toxicity to sea trout larvae for a 24-hour exposure in the presence of UV light was nearly 200 times greater than the toxicity for a 72-hour exposure without UV light.

The degree of photo-induced toxicity was found to be a function of the amount of incident UV light exposure (Lay et al. 2015b; Morris et al. 2015a).

Data from the Trustees' UV/oil bioassays were used to develop a method to adjust the dose-response curves for fish and invertebrates to account for photo-induced toxicity (Lay et al. 2015b). UV-adjusted toxicity values were used to quantify mortalities in water column resources near the ocean surface (Table 4.3-4; Section 4.4).

Table 4.3-4. LC50 values for fish and invertebrates showing adjustment for phototoxicity. Toxicity increased (i.e., lower LC50s) with exposure to UV (Lay et al. 2015b).

Species	Oil	Duration (hours)	LC50 µg/L TPAH50 No UV	LC50 µg/L TPAH50 UV-Adjusted
Ichthyoplankton				
Bay anchovy	B	48	1.4	0.1
Speckled sea trout	B	72	24.7	0.2
Red drum	A	72	27.1	0.2
Bay anchovy	A	48	3.9	0.2
Speckled sea trout	A	72	30.3	0.2
Red drum	B	60	30.9	0.2
Mahi-mahi	A	96	8.8	0.6
Zooplankton				
Copepod	A	96	64.4	2.4
Blue crab	B	48	79.0	2.9

Comparing Lethal Effects Concentrations Determined from Laboratory Testing to Conditions in the Environment During and Following the Spill

The laboratory toxicity tests described in this section are important for establishing quantitative relationships between oil concentrations and toxic effects (i.e., dose-response relationships). As illustrated in Figure 4.3-10, many water samples collected from the Gulf during the spill exceeded concentrations that would cause mortality to water column resources.

Analysis of the degree of toxicity that occurred in the environment is presented in Section 4.4 (Water Column).

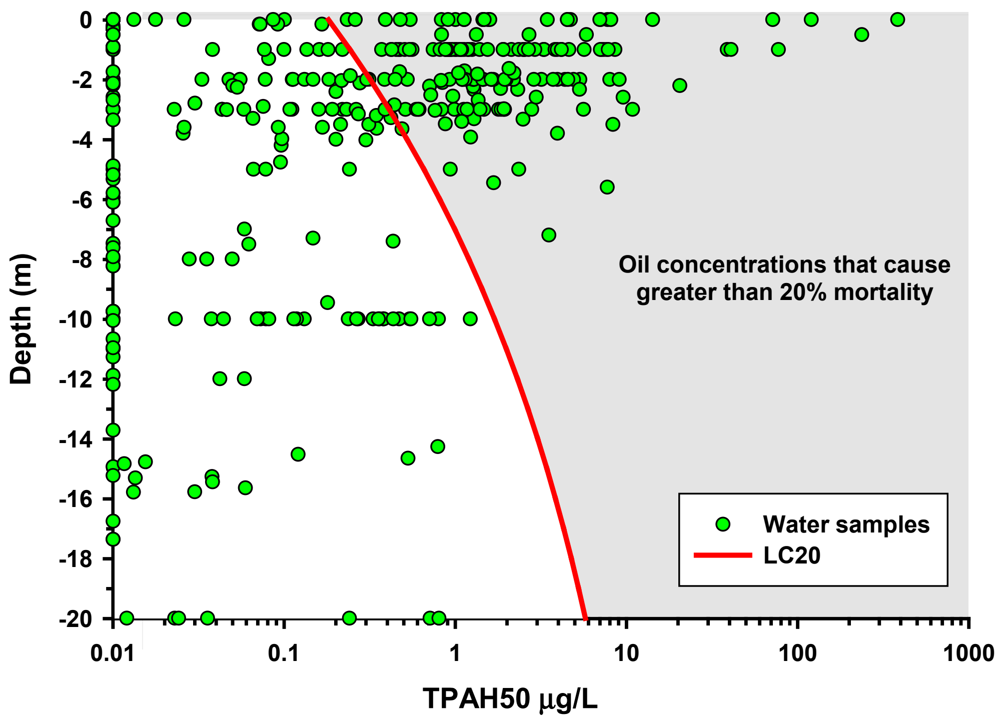


Figure 4.3-10. Many water samples collected during the spill had concentrations of TPAH that exceeded lethal levels determined from the Trustees toxicity test program (see Section 4.4, Water Column). TPAH50 concentrations in water samples collected at different depths during the spill (green dots) are plotted against the LC20 values for early life-stage speckled sea trout adjusted for photo-induced toxicity (red line). The LC20 value (red line) increases (i.e., less toxicity) with depth because ambient UV light is attenuated in the water. Samples falling in the gray-shaded area represent conditions in which mortality to ichthyoplankton would be expected to exceed 20 percent (see Section 4.4, Water Column) (Lay et al. 2015b).

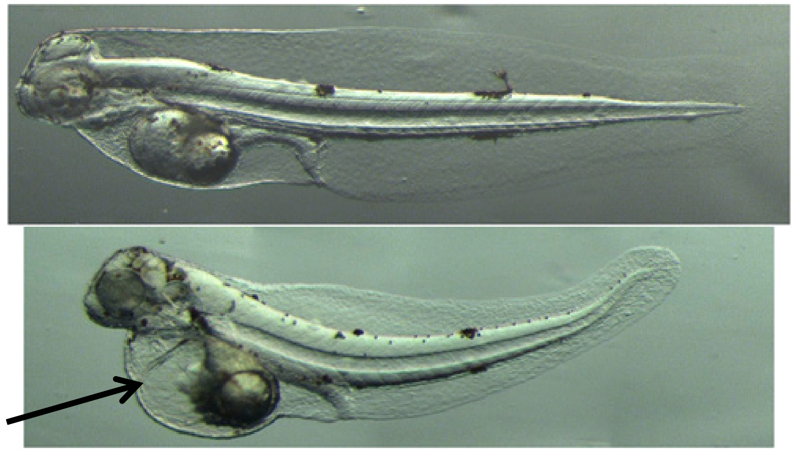
4.3.3.1.2 Other Toxic Effects of DWH Oil on Water Column Resources

In addition to lethality, the Trustees evaluated other toxic effects of DWH oil. Toxic responses documented in water column resources included cardiac (heart) toxicity and other developmental effects, reductions in growth rates, impaired immune function, reduced swimming performance, and other adverse physiological responses. Although not incorporated explicitly in the Trustees' quantification of injuries to water column resources, these other injuries—which can impair the health, fitness, and long-term survival of animals—provide additional context regarding the effects of the spill on the environment.

Cardiac Toxicity and Developmental Abnormalities

One of the more severe types of petroleum toxicity, only discovered relatively recently, is the adverse effect of low concentrations of PAH on heart development and function in fish embryos and larvae. Cardiac impacts on developing fish that are severe enough to impair survival occur at TPAH concentrations lower than those associated with lethality (Carls et al. 1999; de Soysa et al. 2012; Heintz et al. 2000; Incardona et al. 2014; Incardona et al. 2013).

The Trustees evaluated cardiotoxicity in a number of different water column species, including yellowtail amberjack, bluefin and yellowfin tuna, mahi-mahi, and red drum. Exposed fish demonstrated heart-related abnormalities, including decreased heart rates; abnormal heartbeat rhythms; abnormal heart development (e.g., tube heart); and edema (abnormal accumulation of fluid) near the heart, yolk sac, and abdominal areas (Incardona et al. 2014; Incardona & Scholz 2015; Incardona et al. 2013; Morris et al. 2015b; Morris et al. 2015d) (Figure 4.3-11).



Source: *NOAA.*

Figure 4.3-11. Cardiotoxic effects of oil on developing fish. Two images are shown of early life-stage red drum from Trustee studies (Morris et al. 2015b; Morris et al. 2015d). The top picture is a control fish that was not exposed to oil. The bottom fish was exposed to DWH oil (Slick A LEWAF) for 36 hours. This fish developed edema (excess fluid) around the heart (arrow). Other developmental deformities observed included curved spine and reduced growth (Incardona & Scholz 2015). Young fish in the wild with these types of effects are considerably less likely to survive.

Fish developed edema and decreased heart rates at very low oil concentrations, with some EC20 values less than 1 µg/L TPAH50 over 36 to 48 hours (Table 4.3-5) (Incardona et al. 2014; Incardona & Scholz 2015; Morris et al. 2015b; Morris et al. 2015d).

Table 4.3-5. Concentrations of DWH oil (as TPAH50) that resulted in 20 percent effects concentrations (EC20) for cardiotoxicity (e.g., Incardona et al. 2014; Incardona & Scholz 2015; Incardona et al. 2013; Morris et al. 2015b; Morris et al. 2015d).

Species	Life Stage	Duration (hours)	EC20 µg/L TPAH50
Mahi-mahi	Embryo	48	1.3–8.7
Red drum	Embryo	48	1.0–15.7
Southern bluefin tuna	Embryo	36	0.6–3.3
Yellowfin tuna	Embryo	48	0.5–4.1
Yellowtail amberjack	Embryo	48	2.8–8.3

Reduced Growth

Depending on the situation, smaller fish may be at a competitive disadvantage, as larger animals are less likely to be eaten by predators, are better able to catch prey, and have greater reproductive potential. Juvenile red drum and red snapper exposed to waterborne DWH oil for 1 to 2 weeks were smaller (shorter lengths and reduced weights) than control animals (Figure 4.3-12) (Ortell et al. 2015). These results are consistent with tests in fish and invertebrates exposed to DWH oil-contaminated sediment (see Sections 4.3.3.2 and 4.3.3.3, below), and field experiments in which shrimp were placed near heavily oiled marsh areas (Rozas et al. 2014).

Reduced Immune Function

Oil can compromise the immune system's ability to protect oil-exposed animals from disease (Kennedy & Farrell 2008; Khan 1990). The ocean is rich in pathogenic organisms, such as parasites, viruses, and bacteria. While healthy animals are typically able to fight off infections, or at least quickly recover from them, organisms that are exposed to oil would be more susceptible to infections and their negative repercussions.

To determine the potential adverse effects of exposure to DWH oil on the fish immune system, the Trustees conducted tests on red snapper, Atlantic croaker, and red drum in which fish were exposed to both DWH oil and an endemic bacterium from the Gulf of Mexico, *Vibrio anguillarum* (*V. anguillarum*). For example, juvenile red drum were exposed to waterborne oil for 4 days, followed by a 1-hour bacterial challenge. Survival and immune system effects of these fish were then compared with animals exposed to waterborne oil only, animals exposed to bacteria only (and never exposed to oil), and animals that were not exposed to oil or bacteria.

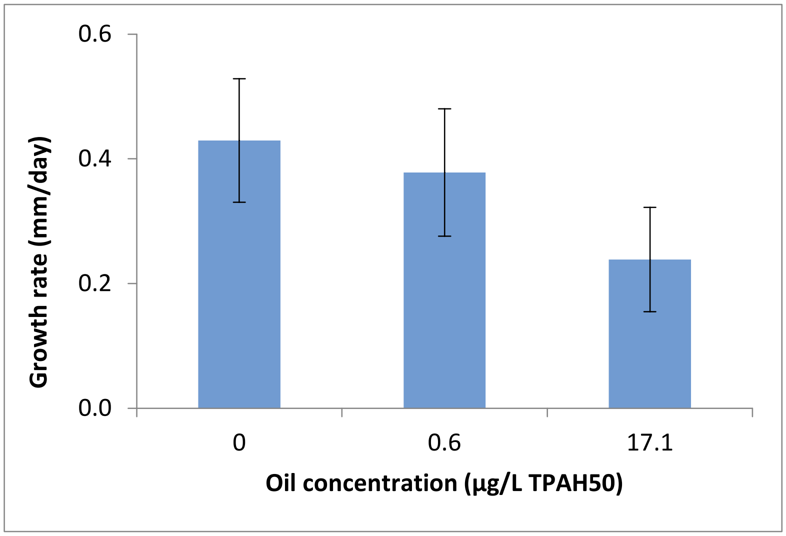


Figure 4.3-12. Effects of DWH oil (Slick A HEWAF) on growth rates of juvenile red snapper (75 days post-hatch) exposed to oil for 17 days (Morris et al. 2015b). Increasing concentrations of TPAH50 were associated with decreased growth rates. Bars show average growth rate over the 17-day period in millimeters per day. The error bars are one standard deviation of the mean.

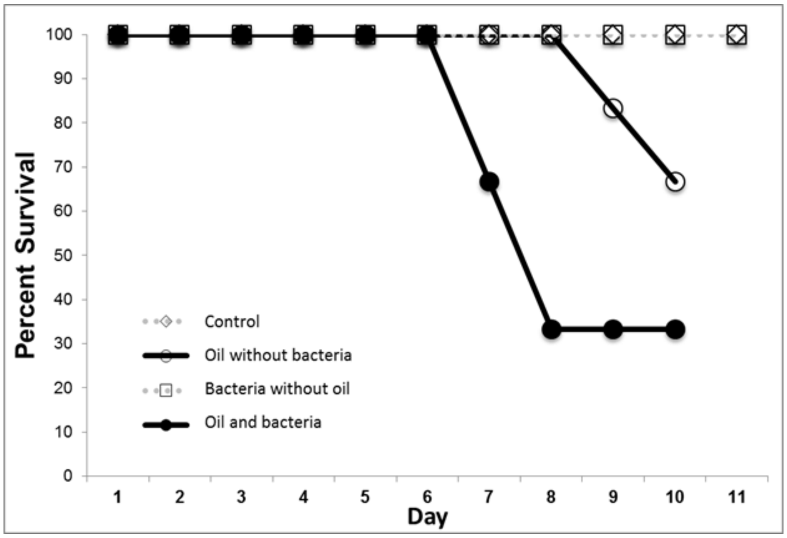


Figure 4.3-13. Percent survival of juvenile red drum exposed to one of four treatments: 1) neither oil nor bacteria (*Vibrio anguillarum*), 2) DWH oil without bacteria, 3) bacteria without oil, 4) DWH oil and bacteria. Exposure to oil and bacteria caused considerably more mortality than in the other treatments (Ortell et al. 2015).

Fish survival results are summarized below and in Figure 4.3-13 (Ortell et al. 2015):

- Greater than 60 percent of the fish exposed to oil and bacteria died.
- Thirty-five percent of fish exposed to just DWH oil (at the same concentration) died.
- All fish exposed to just bacteria without oil or those exposed to no bacteria or oil survived (Ortell et al. 2015) (Figure 4.3-13).

Tests on immune function also demonstrated that exposure to DWH oil resulted in reduced red blood cell counts and a negative effect on genes that are associated with the production of antibodies that help fight off infection.

Based on these data, the Trustees have concluded that exposure to DWH oil causes immunosuppression, which can lead to increased vulnerability to infectious diseases and the ability to recover from infections. Immunosuppressed animals will be at a severe disadvantage compared to unaffected animals.

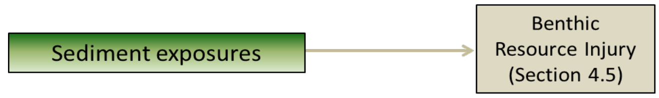
Reduced Swim Performance

The ability of fish to survive and thrive is dependent upon their ability to swim, whether to catch prey or escape from predators. To evaluate potential impacts on swim performance, the Trustees conducted laboratory studies with different life stages of mahi-mahi. The studies showed that the swim performance and associated metabolic and physiological status of mahi-mahi were adversely affected following short-term exposure to DWH oil, whether animals were exposed as embryos, juveniles, or adults. Embryos exposed to as little as 1.2 µg/L TPAH50 (Slick A HEWAF; based on average exposure concentrations) for 48 hours shortly after fertilization and then held and raised in clean water for 30 days experienced significantly reduced swim performance (Mager et al. 2014). Juvenile (30-day old) and adult mahi-mahi exposed to 30 µg/L or 8.4 µg/L TPAH50 (Slick A HEWAF), respectively, for 24 hours also experienced significantly reduced swim performance compared to controls (Mager et al. 2014; Morris et al. 2015b). Although this injury was not directly quantified, the Trustees concluded that fish exposed to DWH oil may have suffered from swim performance injuries that could have reduced their ability to escape predators or capture prey.

4.3.3.2 Toxicity of DWH Oil to Benthic Resources

Key Findings

- DWH oil was toxic to benthic resources in Gulf of Mexico sediments contaminated in the field during the spill and sediment spiked with DWH oil in the laboratory.
- Bottom-dwelling invertebrates experienced reduced survival, growth, and reproduction when exposed to DWH oil in Gulf of Mexico sediments from benthic nearshore and deep-sea environments.



The benthic resources toxicity testing focused on evaluating the toxicity of sediments contaminated with DWH oil to bottom-dwelling invertebrates (Table 4.3-1). The Trustees conducted bioassays using contaminated sediment collected from coastal areas of the northern Gulf of Mexico that experienced heavy oiling, or by adding varying amounts of DWH oil to clean sediments collected from the field (Krasnec et al. 2015a). The Trustees also evaluated the results of toxicity tests conducted with contaminated deep-sea sediments and amphipods performed by BP (Krasnec et al. 2015b).

In addition to being used for the benthic resources injury assessment (Section 4.5), sediment toxicity testing was also used to support the nearshore resources assessment. Those tests are discussed below in Section 4.3.3.3.

Benthic invertebrates, such as amphipods, are highly abundant, burrow into the sea floor, and are an important food source for many fish and invertebrate species. *Leptocheirus plumulosus* (*L. plumulosus*), a burrowing amphipod, was exposed to sediments contaminated with DWH oil to investigate effects on survival, reproduction, and growth. The amphipods were exposed for either 10 or 28 days to contaminated sediments collected from the northern Gulf of Mexico or sediments spiked in the laboratory using Slick A or B oils.

Exposure of the amphipods to oil-contaminated sediments resulted in mortality, with a calculated LC20 value of 7.2 mg/kg TPAH50 in 10-day tests over a range of sediment types (Figure 4.3-14) (Morris et al. 2015b). Adverse effects of contaminated sediments on growth and reproduction of the amphipods were also observed. This information is important in assessing injury in deep-sea sediments, in sediment adjacent to marsh and beach environments, and in marsh soils that have been contaminated by oil.

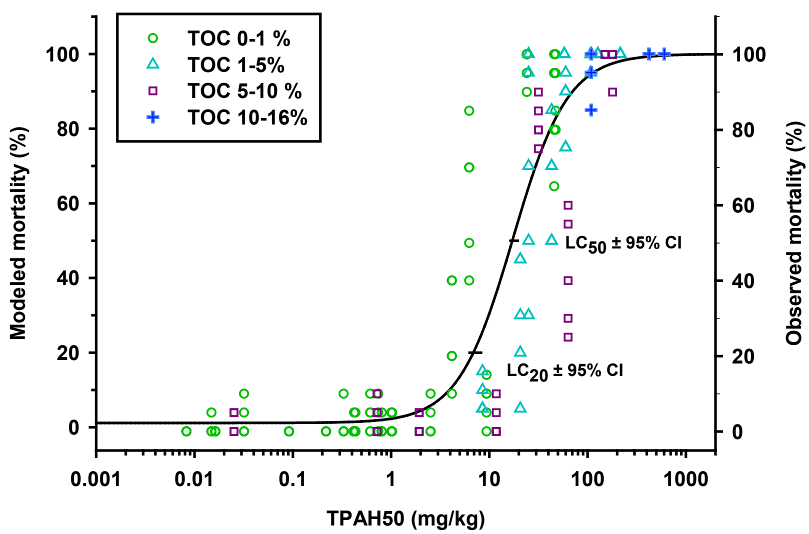
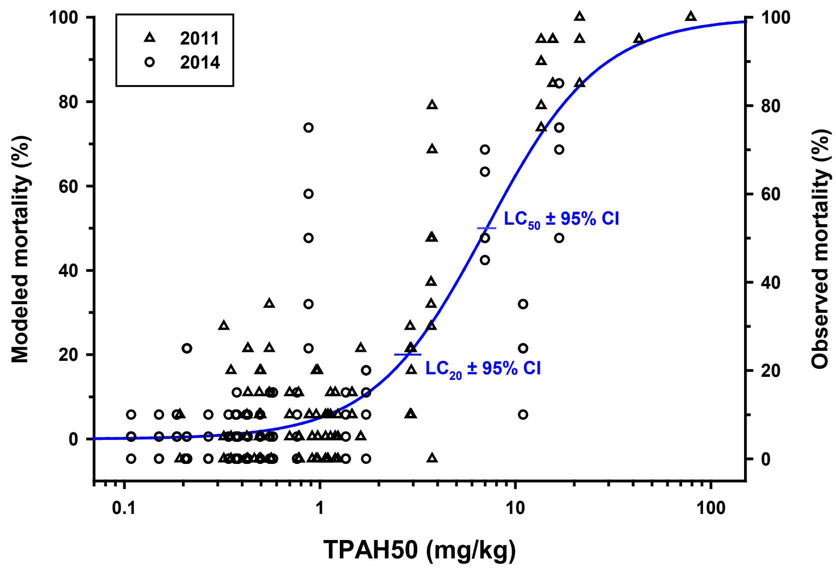


Figure 4.3-14. Juvenile amphipod (*L. plumulosus*) mortality after a 10-day exposure to contaminated sediments collected from the northern Gulf of Mexico or sediments spiked in the laboratory with DWH oil. The LC20 and LC50 values (95 percent CI) are 7.2 (6.3, 8.2) and 17.9 (16.4, 19.5) mg/kg TPAH50, respectively (Morris et al. 2015b). Data are binned according to total organic carbon (TOC) concentrations.

BP also performed sediment toxicity tests with amphipods (*L. plumulosus*) exposed to sediments collected at various times and locations in the deep sea, both near the wellhead and farther afield. The results of these tests were shared with the Trustees. The

Trustees' analysis of the BP data determined that the combined LC20 across the different assays was 2.8 mg/kg TPAH50, a value similar to the results of the Trustees' tests, particularly when considering the low organic carbon content (1.2 percent) of the deep-sea sediments (Figure 4.3-15).

Section 4.5 presents information on injuries to benthic resources, including a comparison of ECs to concentrations in sediments collected from the environment following the spill. Additional data collected by the Trustees on the toxicity of sediments to nearshore resources are described below in Section 4.3.3.3.



Source: *Krasnec et al. 2015*

Figure 4.3-15. Lethal toxicity (percent [%] mortality) of sediments collected from benthic deep-sea environments to amphipods. Data were generated by BP and shared with the Trustees who completed independent data analyses (Krasnec et al. 2015b). The average TOC content in these sediments was 1.2 percent (±0.3).

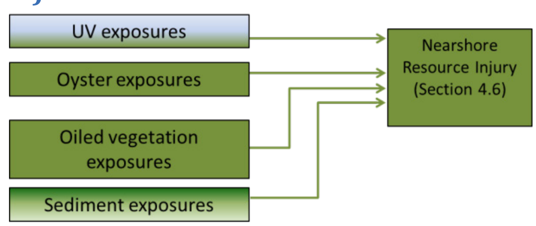
4.3.3.3 Toxicity of DWH Oil to Nearshore Resources

Key Findings

- The results of the Trustees' laboratory toxicity tests demonstrated that exposure of marsh organisms to sediments contaminated with DWH oil resulted in a series of adverse effects, including death, reduced growth, and reduced reproductive success.
- Higher concentrations of TPAH50 in sediments resulted in more adverse effects in more test species. Adverse effects were observed at concentrations as low as approximately 1 mg/kg TPAH in sediment.
- Southern flounder were adversely affected by exposure to oil-contaminated sediment. Toxic effects included damage to gill and liver tissues, reduced growth rates, and mortality.
- Exposure to oil-contaminated sediments caused growth inhibition in juvenile red drum and Pacific white shrimp.

- Gulf killifish embryos exposed to oil-contaminated suspended sediment were less likely to hatch or to survive after hatching.
- There was substantial mortality to fiddler crab offspring exposed to relatively low concentrations of oil in or on sediments as embryos, when followed by exposure to sunlight in clean water as hatched larvae.
- When marsh periwinkles were exposed to DWH oil on plants, they exhibited increased mortality and an impaired ability to move away from oil.
- Exposure to DWH oil caused adverse effects in all oyster life stages tested, at varying ECs.

4.3.3.3.1 Southern Flounder (*Paralichthys lethostigma*)



Flatfish, such as flounder, are particularly vulnerable to the toxic effects of contaminated sediments because they spend most of their lives partially covered by sediments on the sea floor. Southern flounder, a flatfish that lives in the northern Gulf of Mexico, were exposed to sediments spiked with weathered DWH oil (Slick B).

The tests showed a suite of toxic effects that ranged from tissue damage to lethality, depending on the concentration of TPAH50 in the sediment. Southern flounder exposed to contaminated sediment showed evidence of damage to gill tissues at lower concentrations (EC20 values from 0.3 mg/kg to 1.3 mg/kg TPAH50). Damage to gill tissue can have a detrimental effect on an animal's respiratory system (their ability to extract oxygen from water). Reduced growth rates were observed at higher concentrations (Figure 4.3-16; EC20 = 12.8 mg/kg TPAH50). Both of these impacts could have a negative impact on fish in the wild. Exposure to contaminated sediments also resulted in direct mortality, with an LC20 value of 36.3 mg/kg TPAH50 (Brown-Peterson et al. 2015; Morris et al. 2015b).

As discussed earlier, oil exposure can cause immunotoxicity, putting animals at a higher risk of infection and reducing their ability to survive infections (Kennedy & Farrell 2008; Khan 1990). Trustee testing involved exposing Southern flounder to sediment spiked with DWH oil (58 mg/kg TPAH50) for 7 days, followed by a 1-hour exposure to seawater containing *V.*

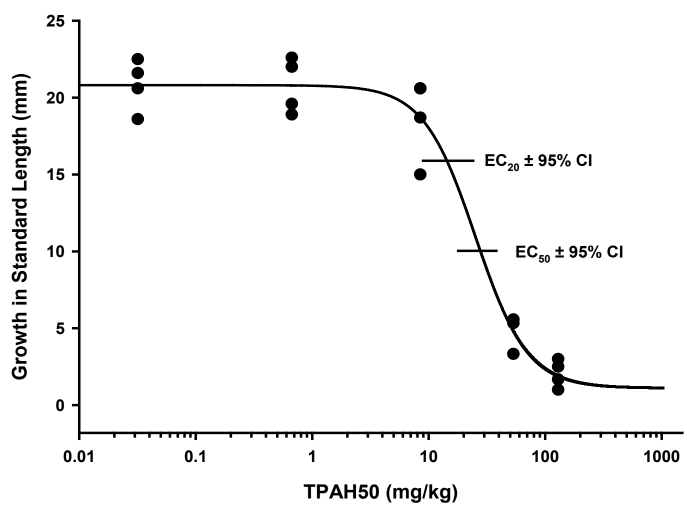


Figure 4.3-16. Growth in juvenile southern flounder after a 32-day exposure to sediments spiked with Slick B oil. The EC20 and EC50 values (with the 95 percent CIs in parentheses) are 12.8 (8.8–23.7) and 26.7 (17.6–37.4) mg/kg TPAH50, respectively (Brown-Peterson et al. 2015; Morris et al. 2015b).

anguillarum for 1 hour. Two days after the exposure to bacteria, 94 percent of the fish died, whereas there was no mortality at the same time period (7 days) when fish were exposed to oil alone (Morris et al. 2015b; Ortell et al. 2015). This large increase in oil toxicity that occurred following a short exposure to bacteria that is widespread in the Gulf of Mexico illustrates that the potential effects of oil in the environment—where animals are subject to natural stresses—may be greater than in carefully controlled laboratory conditions.

4.3.3.3.2 Red Drum (*Sciaenops ocellatus*)

Juvenile red drum forage in sediments and inundated marsh soils. The Trustees found that exposure to oil-contaminated sediments for 13 days caused growth inhibition in juvenile red drum (Figure 4.3-17). The growth of the exposed fish was reduced, with an EC20 value of 37 mg/kg TPAH50 (Morris et al. 2015b).

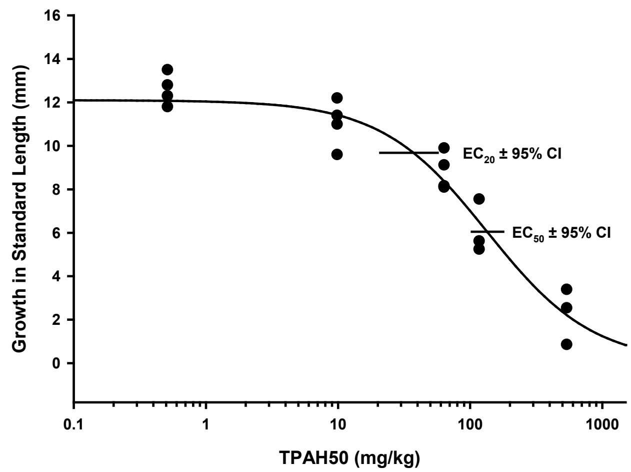
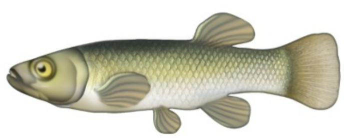


Figure 4.3-17. Growth in juvenile red drum after a 13-day exposure to sediments spiked with Slick B oil. The EC20 and EC50 values (with the 95 percent CIs in parentheses) are 37.1 (20.9–56.8) and 134 (103–178) mg/kg TPAH50, respectively (Morris et al. 2015b).

4.3.3.3.3 Gulf Killifish (*Fundulus grandis*)



In addition to direct contact with sediment on the seafloor, organisms in some nearshore environments were exposed to suspended sediments contaminated with oil. To investigate the toxic effects of contaminated suspended sediment, the Trustees exposed Gulf killifish embryos to fine-grained sediments spiked with Slick B oil. The sediments were suspended in the exposure water by gently moving the test chambers on an orbital shaker table. Exposure to contaminated suspended sediments impaired Gulf killifish embryo development and resulted in decreased hatch rates and increased mortality. These effects were combined into an “unviable embryo” endpoint. The LC20 for this unviable embryo response was 15.5 mg/kg TPAH50 in the underlying sediment that was suspended (Figure 4.3-18) (Morris et al. 2015b). These data demonstrate that exposure to oil-contaminated suspended sediments can cause toxic effects on reproduction and survival.

4.3.3.3.4 Shrimp

The Trustees relied on field evaluations to determine injury to white and brown shrimp. To augment field studies, a laboratory toxicity test was performed. Pacific white shrimp were used as a surrogate species for the brown and white shrimp that inhabit the Gulf of Mexico. Young shrimp (about 10-day old post-larvae) demonstrated reduced growth after only 6 days of exposure to sediment mixed with DWH oil, with an EC20 of 4.3 mg/kg TPAH50 (Morris et al. 2015b).

4.3.3.3.5 Fiddler Crabs (*Uca longisignalis*)

Fiddler crabs live in the intertidal zone of marshes, mudflats, and beaches. The burrows that they dig in the sediment are often inundated by seawater during high tides. In order to evaluate the UV photo-induced toxicity of sediment contaminated with DWH oil on early life-stage fiddler crabs, the Trustees designed an experiment to expose female adult crabs and their external egg masses to oiled sediment and UV light.

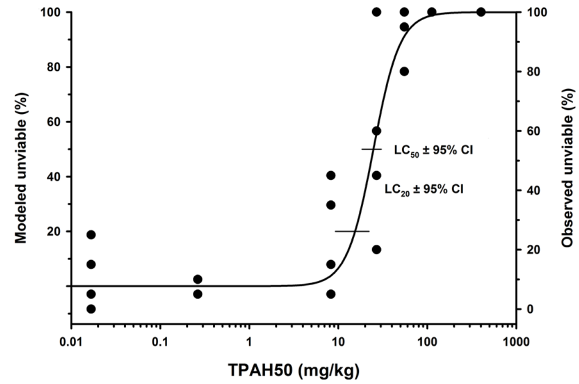


Figure 4.3-18. Relationship between TPAH50 (mg/kg) and viability of Gulf killifish embryos following a 20-day exposure to suspended sediments spiked with Slick B oil. The LC20 and LC50 values (with 95 percent CIs in parentheses) are 15.5 (9.3–22.1) mg/kg and 24.8 (18.5–30.4) mg/kg TPAH50, respectively. These values were calculated using the concentrations of TPAH50 in the sediment, which was suspended during the bioassay (Morris et al. 2015b).

In this study, adult fiddler crabs were placed on sediments coated with DWH oil for 10 days. The TPAH50 concentrations in the upper 2 centimeters (cm) of these sediments ranged from 0.07 (clean reference sediment) to 26 mg/kg. During the exposure period, several female crabs became gravid (i.e., produced fertilized egg masses that remain attached to the female; also known as "sponge crab"). The Trustees removed the gravid females from the oiled sediment exposures after 10 days and placed them in clean water for another 2 to 4 days until the embryos in their external egg masses hatched. After hatch, the zoea (i.e., larvae) were collected and either held indoors in clean water or exposed to varying levels of ambient sunlight in clean water for approximately 7 hours. Zoea from females exposed to contaminated sediment as embryos *and* subsequently exposed to sunlight in clean water experienced substantial mortality, with a calculated LC20 value of 0.51 mg/kg TPAH50 in the upper 2 cm of sediment (Figure 4.3-19) (Morris et al. 2015b). This study demonstrates substantial toxicity to fiddler crab offspring at relatively low concentrations of oil in or on sediments, even though the oiled sediments did not affect adult survival, fecundity, or behavior.

4.3.3.3.6 Marsh Periwinkle Snails (*Littoraria irrorata*)

As oil was deposited on shorelines, it covered sediment, soil, and vegetation that provide habitat for invertebrates such as snails and insects. The Trustees' assessment team determined that marsh periwinkles were injured by oiling: the densities of marsh periwinkles in oiled coastal habitats (e.g., marsh shoreline edge and marsh interior) were dramatically reduced compared with unoiled marsh (Section 4.6, Nearshore). To augment the field-based injury evaluation, the Trustees conducted a series of tests in which periwinkles were exposed to oiled marsh grass. These studies were designed to be similar to conditions on heavily oiled marsh platform sites where vegetation along the exposed marsh edge was most heavily oiled and often was lying flat on the marsh surface (Section 4.6). The tests were designed to assess whether exposure to oiled marsh platforms would affect periwinkle survival or their ability to move away from the oil toward cleaner upright plants farther back in the marsh.

In the first series of tests, periwinkles were placed on *Spartina* plant shoots coated with DWH oil at a thickness similar to that measured at heavily oiled marsh sites (about 1 cm) (Zengel et al. 2015). The amount of time it took for the periwinkles to move off of the oiled vegetation to non-oiled standing *Spartina* shoots about 23 cm away was recorded (Garner et al. 2015). Control animals in trays without oil reached the standing vegetation more quickly than the oil-exposed periwinkles. In the control group, 70 percent of the animals reached the standing vegetation in about 1 hour, and 85 percent of the periwinkles reached standing vegetation in about 2.5 hours. In contrast, only 18 percent of the oiled group reached the standing vegetation in about 4 hours, and only 22 percent reached the vegetation over the entire 72-hour duration of the test. Periwinkles in the oiled treatment that did not reach the standing vegetation in this test experienced very high mortality (77 percent). The results of this laboratory test suggest that periwinkles living in areas with heavily oiled marsh vegetation would likely have died.

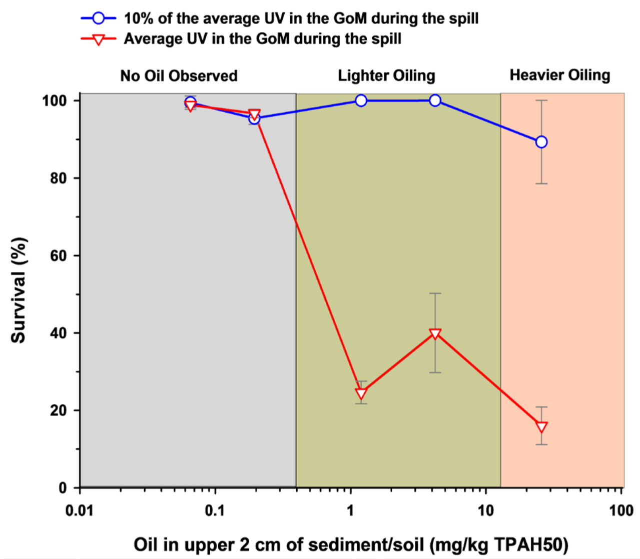


Figure 4.3-19. Survival of fiddler crab larvae exposed to contaminated sediments as embryos for up to 10 days and exposed to natural UV light in clean water for 1 day. Larvae exposed to full sunlight (100 percent of average incident UV in the northern Gulf of Mexico) suffered high mortality (triangles with red line). Little mortality was observed when crabs were only exposed to 10 percent of average incident UV light (circles with blue line). The colored bands represent the range of TPAH50 measured in different marsh shoreline oiling categories discussed in Section 4.6 (Morris et al. 2015b).

A follow-up test showed that periwinkle mortality was clearly related to the duration of oil exposure. After 16 hours, 35 percent of the periwinkles exposed to oil died. Mortality was 68 percent and 98 percent after 32 and 72 hours of oil exposure, respectively (Garner et al. 2015). Control periwinkles not exposed to oil had 0 percent mortality. Overall, the Trustees' laboratory test data support the findings and conclusions of the field studies in which heavily oiled sites had greatly reduced periwinkle abundance (see Section 4.6, Nearshore, for additional information).

4.3.3.3.7 Eastern Oysters (*Crassostrea virginica*)

The Trustees relied on field studies to evaluate injury to nearshore and subtidal eastern oysters. To augment these studies, a series of laboratory toxicity tests were performed with eastern oysters (native to the northern Gulf of Mexico). The DWH oil spill occurred during the peak of the oyster spawning season and contaminated oyster habitats that support spawning adults, developing embryos, and larvae (Section 4.6, Nearshore).

The Trustees tested oysters at all stages of development, including gametes (sperm and unfertilized eggs), embryos, planktonic veligers (i.e., larvae), spat (i.e., larvae attached to substrate), juveniles, and adults (Section 4.6 presents more information on the oyster life cycle). Oysters at various life stages were exposed to DWH oil in the water column and in sediment (both settled and suspended) to address different routes of exposure.

When oyster gametes (eggs and sperm) were exposed in the laboratory to DWH oil in water or suspended sediments, fertilization rates decreased. For example, when oyster gametes were exposed to oil (Slick A HEWAF) and UV light, EC50 values for fertilization ranged from 13 µg/L to 116 µg/L TPAH50 depending on the UV dose (Morris et al. 2015b). When gametes were exposed to suspended sediments that were collected during the response effort (Krasnec et al. 2015a), fertilization rates also decreased (EC20 = 40.6 µg/L TPAH50; Figure 4.3-20) (Morris et al. 2015b).

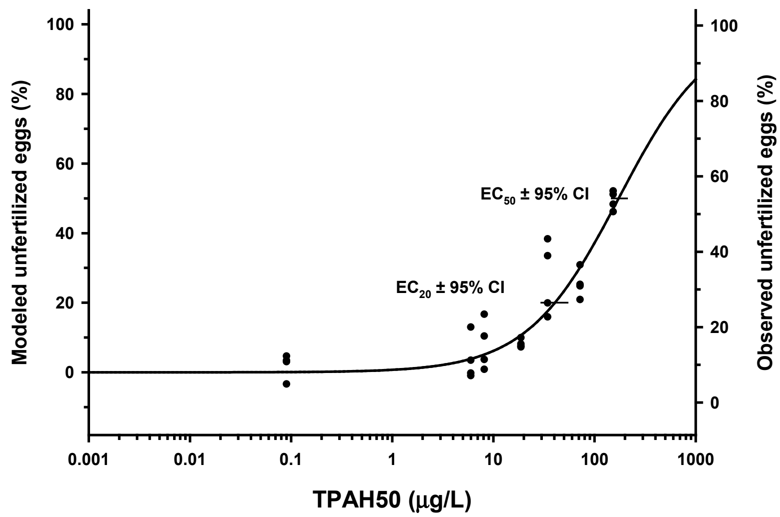


Figure 4.3-20. Effects of exposure to suspended fine-grain sediment contaminated with DWH oil on oyster fertilization. Gametes were exposed shortly after spawning and sampled after 1 hour of exposure to assess fertilization success. The EC20 on unfertilized eggs after 1 hour of exposure was 40.6 (95 percent CI 29.8–54.1) µg/L TPAH50 (Morris et al. 2015b).

Early life-stage oysters (embryos, veligers, pediveligers, and early spat) were also adversely affected by exposure to DWH oil. Waterborne weathered oil (Slick A HEWAFs) caused lethal and sublethal effects to early life -stage oysters (LC20 and EC20 [abnormality] values ranging from 33 µg/L to 645 µg/L TPAH50). Interestingly, the addition of dispersant (Slick A CEWAFs) resulted in lethal and sublethal effects at lower concentrations of oil (higher toxicity; LC20 and EC20 [abnormality] values ranging from 12 µg/L to 133 µg/L TPAH50) (Morris et al. 2015b).

In addition to WAF exposures, early life-stage oysters were also exposed to contaminated suspended sediments that were collected during the response effort (Krasnec et al. 2015a). Regardless of the exposure duration or the life stage tested, the veligers from those exposures were more likely to have developmental abnormalities than unexposed animals (Vignier et al. [In Press]). Veligers that were raised from contaminated sediment-exposed gametes were the most sensitive (abnormality EC20 = 1.1 µg/L TPAH50 for 24 hours of exposure), compared to veligers that were raised from exposed embryos (abnormality EC20 = 77.7 µg/L TPAH50 for 24 hours of exposure) or exposed as veligers themselves (abnormality EC20 = 95.9 µg/L TPAH50 for 48 hours of exposure) (Morris et al. 2015b). Pediveligers that were exposed to sediment spiked with Slick B oil had decreased settlement rates (EC20 = 6.5 mg/kg TPAH50).

4.3.3.3.8 Summary of Sediment Toxicity Testing Results

The Trustees' sediment toxicity tests revealed a series of different adverse effects over a range of sediment exposure concentrations. Overall, the higher the concentration of TPAH50 in the sediments, the more effects were observed in more test species and at a greater degree of severity (Figure 4.3-21). As described in Section 4.6 (Nearshore), these adverse ECs were exceeded along extensive lengths of marsh shorelines that were oiled as a consequence of the DWH spill.

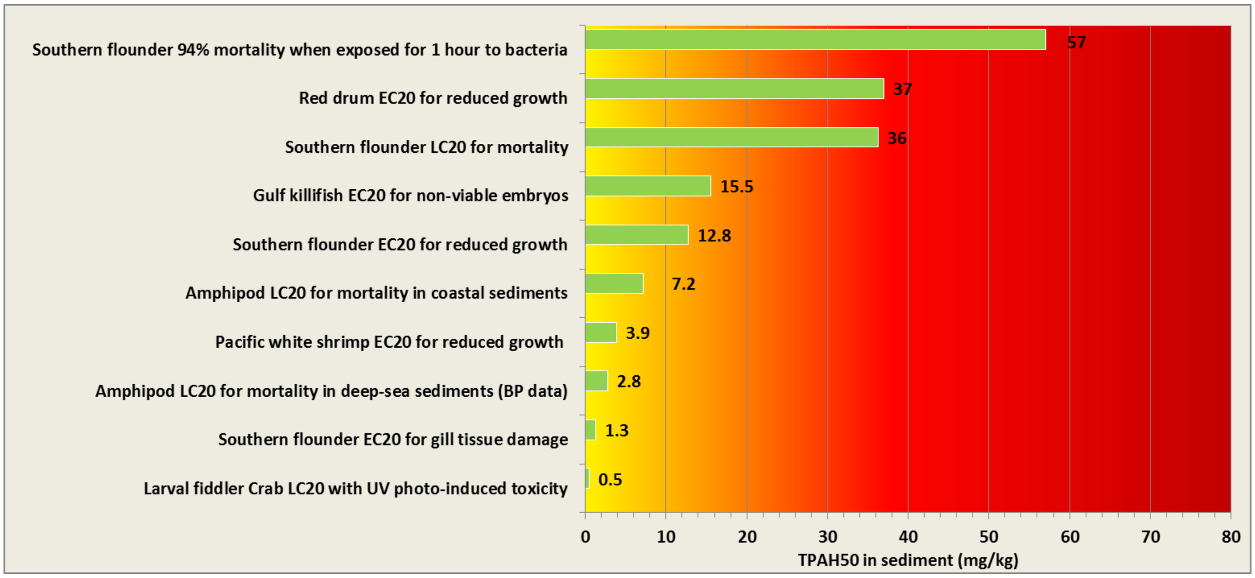


Figure 4.3-21. Toxicological ECs, shown in mg TPAH50 per kg of sediment, for organisms exposed to sediment contaminated with DWH oil. Higher sediment concentrations of TPAH50 resulted in more severe effects to more test species (Morris et al. 2015b).

4.3.3.3.9 Gulf Sturgeon

The Gulf sturgeon (*Acipenser oxyrinchus desotoi*) is an anadromous fish, which migrates from salt water into large coastal rivers to spawn. Data collected from the field by the Trustees indicated that these protected fish were potentially exposed to DWH oil.

The Trustees performed controlled exposures of shovelnose sturgeon as a surrogate for the Gulf sturgeon. Juvenile shovelnose sturgeon were exposed to HEWAFs at a concentration range of 5 µg/L to 10 µg/L TPAH33 for 7 or 28 days (FWS 2015). Investigators identified significant changes in cell process pathways related to immune system function (including neutrophils and T and B cell processes), wound healing, and DNA replication. Genetic analysis identified changes in cell processes related to DNA damage and repair, as compared to control fish (FWS 2015).

The results of this laboratory study suggest that Gulf sturgeon were likely experiencing differential gene expression alterations after both short-term and longer-term oil exposure. Overall, the laboratory study provided evidence of adverse health outcomes of DNA damage at the molecular and biochemical levels, and immune injury at the molecular, biochemical, cellular, and organ levels (FWS 2015).

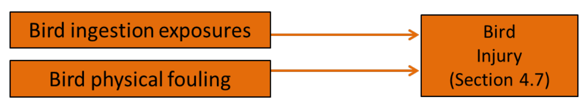
Section 4.6.7 presents additional information on injury determination for Gulf sturgeon.

4.3.3.4 Toxicity of DWH Oil to Birds

Key Findings

- When birds ingested food contaminated with DWH oil, they suffered from a variety of adverse health effects, including hemolytic anemia, liver dysfunction, kidney damage, hypothermia, weight loss, lethargy, abnormal feces, feather damage, moribundity (near death), and death.
- Ingestion of DWH oil caused several types of organ damage and dysfunction, including to liver, kidney, gastrointestinal tract, and cardiovascular systems. Ingestion of DWH oil disrupted digestive tract function, resulting in direct damage to tissues and poor absorption of fluids and nutrients.
- The Trustees' studies found previously undescribed alterations in heart function following oil ingestion, including heart tissue abnormalities, changes to heart function, and decreased blood pressure. Overall, disruption of organ physiology and function would have considerable negative consequences for a bird's fitness and survival.
- External oiling caused feather damage and reduced flight performance. Oiled birds demonstrated more erratic and less efficient flying, slower take-off speeds, shorter flight times, and higher energy costs.

4.3.3.4.1 Effects of Ingested Oil on Birds



In Trustee laboratory tests, when birds ingested oil-contaminated fish, they suffered from a variety of adverse health effects, including hemolytic anemia, liver dysfunction, kidney damage, hypothermia, weight loss, lethargy, abnormal feces, feather damage, moribundity (near death), and death.

Double-crested cormorants and laughing gulls were orally dosed daily with 0, 5, or 10 milliliters of oil per kg body weight for up to 21 days (cormorants) and 28 days (gulls) (Bursian et al. 2015b; Bursian et al. 2015c). The results of the oral toxicity studies indicate that oral dosing of Gulf-relevant species with DWH oil resulted in clinical signs and changes in a number of hematological, biochemical, and tissue endpoints consistent with oil exposures in previous field and laboratory studies (Ziccardi 2015), including:

- Clinical signs of anemia consistent with oxidative damage to red blood cells included decreased packed cell volume (PCV), increased incidence of red blood cells containing Heinz bodies (an indicator of impacts to hemoglobin), and changes in plasma clinical chemistries.
- Changes in oxidative stress endpoints that provide further evidence of systemic oxidative damage.
- Increased liver weights and decreased plasma cholesterol, glucose, and total protein concentrations, as well as concentrations of protein fractions found in dosed birds were indicative of liver dysfunction.

- Increased plasma urea and uric acid, along with pathological changes in the kidney were suggestive of kidney damage.
- During blood sample collections, many of the dosed birds were found to have diminished blood clotting abilities. Also during necropsies, some of the dosed birds were found to have hearts that were enlarged and flaccid. Both of these observations were new findings in birds exposed to oil.

Birds that ingested oil during preening of contaminated feathers also suffered adverse health effects, including lethargy, abnormal feces (i.e., watery feces with evidence of blood and tissue), feather damage, feather plucking, moribundity, signs of anemia, and heart defects (Dorr et al. 2015). Approximately 13 grams of oil were applied to the breast and back of the double crested cormorants every 3 days over a 2-week period (total of six applications). The level of oiling on the birds' plumage in these studies was consistent with a "moderate" degree of oiling (21 to 40 percent coverage of body), as described in Section 4.7 (Birds).

Exposed animals demonstrated a significant decrease in PCV and significant increases in Heinz bodies and reticulocyte counts. Similar effects were found in birds collected from oil-contaminated areas during the summer and fall of 2010 (Fallon et al. 2014). Like in the study with contaminated prey, many of the dosed birds had diminished blood clotting abilities. Liver, kidney, and gastrointestinal tract weight increased in response to oil application, and some exposed birds had cardiac abnormalities (as diagnosed by echocardiograms). In a similar study, birds that preened oil from their feathers had reduced body temperatures and a greater loss of body mass/body fat (Maggini et al. 2015).

A major consequence of oil ingestion, supported by Trustee laboratory studies, Trustee field studies (Fallon et al. 2014), and previously published work, is significant alterations to red blood cell presence and function. Oil exposure leads to the denaturation of hemoglobin, formation of Heinz bodies within cells, and reduction in the oxygen carrying capacity. This can have significant effects on bird performance, limiting their ability to fly, swim, and forage, with subsequent increased risk of death. Ingestion of DWH oil also decreased white blood cell counts, with related adverse effects on immune function. Immune impairment can reduce a bird's ability to combat bacterial, fungal, viral, or parasitic infections, increasing the risk of death.

Ingestion of DWH oil caused several types of organ damage and dysfunction, including liver, kidney, gastrointestinal tract, and cardiovascular systems. Ingestion of DWH oil also disrupted digestive tract function, resulting in direct damage to tissues and poor absorption of fluids and nutrients. Finally, the Trustee studies found previously undescribed alterations in cardiovascular function following oil ingestion, including heart tissue abnormalities (e.g., flaccid heart musculature), changes to heart function (e.g., increased ejection velocities and volumes), and decreased blood pressure. Overall, disruption of organ physiology and function would have considerable negative consequences for an animal's fitness and survival (Ziccardi 2015).

4.3.3.4.2 Physical Effects of External Oil

In addition to the toxic effects of oil, the viscous, sticky nature of oil negatively impacts birds' abilities to take off, fly, and follow efficient flight paths.

Using a wind tunnel, the Trustees measured how oil on the body (not the wings) affected flight energetics and flight ability for western sandpipers (Maggini et al. 2015). Trace oiling (less than 5 percent of body surface) and moderate oiling (21 to 40 percent) caused increases in the average energy cost of flight relative to baseline, with moderately oiled birds being the most affected. External oiling also caused more erratic flying (birds were more likely to run into the wall of the wind tunnel) and a preference for shorter flight times (Maggini et al. 2015). Moderately oiled birds had faster wingbeat frequencies and larger wing movements, leading to higher energy costs.

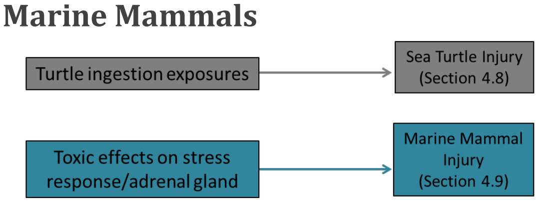
The flight performance of birds with oil on their wings and tail feathers was similarly affected. The Trustees used high-speed video to determine how trace levels of oil on wing and tail feathers affected the speed and angle of a bird's takeoff movements, and how that affected flight energy costs. Takeoff speed was slower in oiled birds, and they had to work harder to achieve flight when compared to unoiled birds (Maggini et al. 2015).

To investigate how oil affects a bird's ability to fly in a natural environment, the Trustees assessed the effects of externally applied DWH oil on the field-based flight performance of homing pigeons (Pritsos et al. 2015). The Trustees compared the flight performance of individual homing pigeons before and after oiling over trips of 50, 85, and 100 miles (from the release site to their home loft). After birds were exposed to oil, even in the light oiling category, they flew less efficiently (e.g., greater fluctuation in flight altitude over the course of the flight and altered flight paths) and took longer to return to their home loft. The studies also examined adverse health effects in oiled animals. Compared to their pre-oil flights, oiled animals weighed less after they returned to their home loft and took longer to recover the weight that is lost normally during the flight. Oiled birds also had indications of liver dysfunction and increased mortality.

The results of these flight studies indicate that even a trace amount of oil can cause a substantial increase in energy costs and can ultimately affect flight energetics and migratory performance. Mechanical effects (e.g., increased drag due to external oil) were partly responsible for increased energy demands for flight, implying that these results could be applied to all flying birds. In migratory birds, the added energetic costs of flight when oiled would result in additional time spent feeding, resting, and preening, which in turn would slow down their migration, affecting both their breeding performance and survival probability. These results imply that oiled birds will be less fit than non-oiled birds, leading to increased vulnerability to predation, reduced energy stores, and delayed arrival at breeding grounds (Ziccardi 2015).

4.3.3.5 Toxicity of DWH Oil to Sea Turtles and Marine Mammals

The Trustees' injury assessment of sea turtles and marine mammals included very limited toxicity testing because these organisms are federally protected.



4.3.3.5.1 Sea Turtles

During the spill, response workers collected many oiled sea turtles from the Gulf of Mexico. Sea turtles had DWH oil covering their bodies and coating their esophagi (Section 4.8). As a component of the sea turtle injury assessment, the Trustees conducted limited laboratory testing with two surrogate turtle species, red-eared sliders (*Trachemys scripta elegans*) and snapping turtles (*Chelydra serpentina*).

Exposures with surrogate species allowed us to obtain information on the toxicity of ingested oil that could not be measured directly in federally protected sea turtles.

Two oil exposure regimens were used (in addition to a control group) to approximate exposures estimated for minimally, lightly, and moderately oiled sea turtles observed in the field (see Section 4.8, Sea Turtles). Turtles were dosed daily for 14 days. Findings from the surrogate study, presented in Mitchelmore et al. (2015) and Mitchelmore and Rowe (2015) included the following:

- Most of the turtles (greater than 96 percent) exposed in the laboratory continued to voluntarily feed despite the oil exposure.
- Dose-dependent increases in the levels of PAH were measured, demonstrating uptake and metabolism of oil at levels similar to those measured in the limited number of field-collected sea turtles (Ylitalo et al. 2014).
- Oiled turtles exhibited alterations in multiple toxicity endpoints, including oxidative and DNA damage.
- Observed physiological abnormalities in oil-exposed turtles included evidence of dehydration, decreased digestive function, and decreased assimilation of nutrients.

4.3.3.5.2 Marine Mammals

Bottlenose dolphins living in habitats contaminated with DWH oil showed signs of adrenal dysfunction, and dead, stranded dolphins from areas contaminated with DWH oil had smaller adrenal glands (Schwacke et al. 2014; Venn-Watson et al. 2015). Endocrine systems, including the adrenal gland in mammals (and the kidney in fish), enable vertebrates to respond to changes in their environment. In response to disturbances or stressful situations, chemical signals from the brain trigger a cascade of hormone releases into the bloodstream.

To further investigate the effect of DWH oil on an exposed organism's ability to respond to stress, the Trustees conducted laboratory tests with the Gulf toadfish (*Opsanus beta*) and laboratory-cultured human adrenal cells. Preliminary studies demonstrate that kidney cells from fish exposed to DWH oil exhibit an inhibition in their ability to secrete important stress hormones in response to a stimulant. Similarly, DWH oil caused dysregulation of stress hormone production in preliminary studies with human adrenal cells (the H295R cell line) (Takeshita et al. 2015).

4.3.4 Conclusions

The Trustees conducted a comprehensive program to evaluate the toxic effects of DWH oil on natural resources. The testing program consisted of studies designed to evaluate toxicity for the resource categories and exposure pathways in the injury assessment, discussed below.

Overall, the Trustees found that exposure to DWH oil causes a wide range of toxic effects, including death, impaired reproduction, disease, and other physiological malfunctions that reduce the ability of organisms to survive and thrive. Measured and modeled concentrations of DWH oil in surface water and sediments in the Gulf of Mexico at a number of locations and times during and following the spill were greater than the range of concentrations shown to cause these toxic effects in the Trustees' laboratory studies.

Specific findings from the Trustees' toxicity testing program are presented below.

4.3.4.1 Water Column Resources

- The embryos and larvae (i.e., early life-stage) of fish (ichthyoplankton) and various life stages of pelagic invertebrates (zooplankton) are particularly susceptible to the toxic effects of DWH oil, both when mixed with water and when present in the form of a surface slick. Measured and modeled concentrations of DWH oil in the Gulf of Mexico exceeded lethal levels in a number of locations and times during and following the spill.
- Thin, rainbow sheens of surface slick oil were extremely lethal to fish embryos and invertebrates.
- DWH oil is roughly 10 to 100 times more toxic to semi-transparent invertebrates and early life-stage fish in the presence of natural sunlight.
- In addition to lethality, exposure to DWH oil causes developmental abnormalities, including heart and spinal defects. Many of these developmental abnormalities are severe enough to kill early life-stage fish.
- Older fish (juveniles or adults) are less susceptible than embryos and larvae to the short-term lethal effects of DWH oil exposure. At most of the oil concentrations that occurred after the spill, the toxic effects of oil on older fish are more likely to have manifested as sublethal injuries, including growth inhibition, immunosuppression, decreased swim performance, or an abnormal stress response.
- The lethal effect of DWH oil on fish embryos and larvae and invertebrates has important ecological implications. In addition to sustaining fish and invertebrate populations, these small, planktonic organisms are an important base of the marine food web.

4.3.4.2 Benthic Resources

- Exposure of amphipods, a bottom-dwelling invertebrate, to sediments contaminated with DWH oil resulted in mortality at concentrations observed in deep-sea sediments and nearshore sediment and/or marsh soils following the spill.

4.3.4.3 Nearshore Resources

- Exposure of marsh organisms to sediments contaminated with DWH oil resulted in a series of adverse effects, including death, reduced growth, and reduced reproductive success.
- Higher concentrations of TPAH50 in sediments resulted in more adverse effects in more test species. Adverse effects were observed at concentrations as low as approximately 1 mg/kg TPAH50.

- Southern flounder were adversely affected by exposure to oil-contaminated sediment. Toxic effects included damage to gill and liver tissues, reduced growth rates, and mortality.
- Exposure to oil-contaminated sediments caused growth inhibition in juvenile red drum and Pacific white shrimp.
- Gulf killifish embryos exposed to oil-contaminated suspended sediments were less likely to hatch or to survive after hatching.
- There was substantial mortality to fiddler crab offspring exposed to relatively low concentrations of oil in or on sediments, when followed by exposure to sunlight.
- When marsh periwinkles were exposed to DWH oil on plants, they exhibited increased mortality and an impaired ability to move away from oil.
- Exposure to DWH oil caused adverse effects in all oyster life stages tested, at varying effects concentrations.

4.3.4.4 Birds

- When birds ingested food contaminated with DWH oil, they suffered from a variety of adverse health effects, including hemolytic anemia, liver dysfunction, kidney damage, hypothermia, weight loss, lethargy, abnormal feces, feather damage, moribundity (near death), and death.
- Ingestion of DWH oil caused several types of organ damage and dysfunction, including to liver, kidney, gastrointestinal tract, and cardiovascular systems. Ingestion of DWH oil disrupted digestive tract function, resulting in direct damage to tissues and poor absorption of fluids and nutrients.
- The Trustee studies found previously undescribed alterations in heart function following oil ingestion, including heart tissue abnormalities, changes to heart function, and decreased blood pressure. Overall, disruption of organ physiology and function would have considerable negative consequences for a bird's fitness and survival.
- External oiling caused feather damage and reduced flight performance. Oiled birds demonstrated more erratic and less efficient flying, shorter flight times, and higher energetic costs.

4.3.4.5 Sea Turtles and Marine Mammals

- Surrogate species of freshwater turtles that ingested DWH oil exhibited statistically significant alterations in multiple toxicity endpoints, oxidative damage, and DNA damage.
- Observed physiological abnormalities in oil-exposed turtles included evidence of dehydration, decreased digestive function and assimilation of nutrients.

- Exposure to DWH oil causes dysregulation of stress hormone secretion from adrenal cells (human cell line) and kidney cells (Gulf toadfish). Impacts on the endocrine system will affect an animal's ability to maintain homeostasis and respond appropriately to stressful situations and will lead to reduced fitness.

4.3.4.6 Common Observations Across the DWH Toxicity Testing Program

When organisms are exposed to chemical contaminants, the resulting toxic effects can manifest in a variety of manners depending on the species, individual life history, and the nature and concentration of the exposure. Despite this variation, the Trustees found a considerable degree of consistency among the types of toxic responses observed across the different organisms tested. For example, the Trustees observed cardiac effects in both fish and birds (Dorr et al. 2015; Incardona et al. 2014; Incardona et al. 2013; Morris et al. 2015b). Disruption of blood cells and function were also observed in fish and birds, both in the laboratory and in the field (Bursian et al. 2015b; Bursian et al. 2015c; Dorr et al. 2015; Morris et al. 2015b). Evidence of oxidative damage following exposure to oil was observed in fish, birds, and turtles (Bursian et al. 2015a; Mitchelmore et al. 2015; Morris et al. 2015b). Impairment of immune system function following exposure to oil was also observed in fish and birds, and was observed in field studies of dolphin health (Bursian et al. 2015b; Bursian et al. 2015c; Dorr et al. 2015; Morris et al. 2015b; Ortell et al. 2015; Venn-Watson et al. 2015). Evidence of impairment to stress responses and adrenal function was observed in fish, birds, and dolphins (Morris et al. 2015b; Takeshita et al. 2015; Venn-Watson et al. 2015). Evidence of impaired swim performance was observed in fish (Mager et al. 2014), and impaired flight performance was observed in birds (Maggini et al. 2015; Pritsos et al. 2015).

Figure 4.3-22 illustrates the range of potential toxicological effects associated with exposure to DWH oil. Not every organism exposed to oil will experience all of the adverse health effects presented, and there is a very wide range in sensitivities between species and between individuals of the same species. However, all of the organisms that were exposed to elevated concentrations of DWH oil were forced to use energy to deal with the toxic insult. Many of those organisms would eventually have recovered fully, but others would have suffered from irreversible physiological effects that resulted in death, reduced life expectancy, or reduced reproduction.

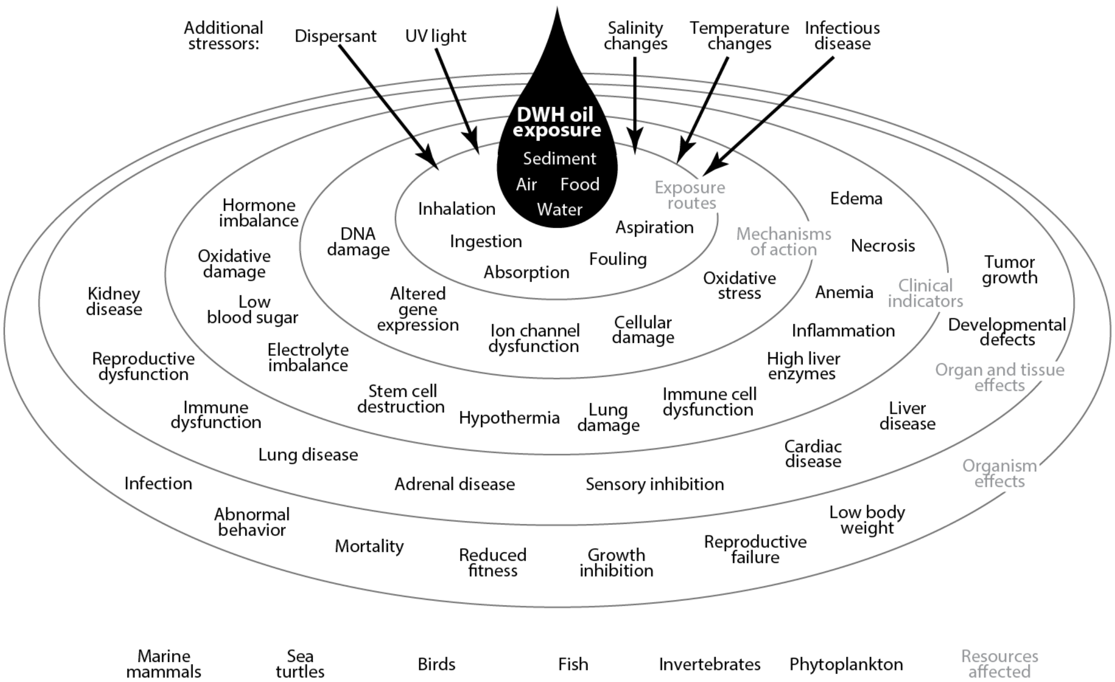


Figure 4.3-22. Conceptual illustration of the constellation of relationships between oil exposure and toxicological effects in organisms that were exposed to DWH oil during and after the spill. All of the exposure scenarios and resulting effects shown are supported by information in the literature and/or data generated through the Trustees' toxicity testing program.

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4.4 Water Column

What Is in This Section?

- **Executive Summary**
- **Introduction and Importance of the Resource (Section 4.4.1):** Why do we care about the water column and the biological resources in the water column?
- **Approach to the Assessment (Section 4.4.2):** How did the Trustees assess injury to the water column?
- **Exposure (Section 4.4.3):** How, and to what extent, were water column organisms exposed to *Deepwater Horizon* (DWH) oil?
- **Injury Determination (Section 4.4.4):** How did exposure to DWH oil affect the water column?
- **Injury Quantification (Section 4.4.5):** What was the magnitude of injury to the water column?
- **Conclusions and Key Aspects of the Injury for Restoration Planning (Section 4.4.6):** What are the Trustees' conclusions about injury to water column organisms, ecosystem effects, and restoration considerations?
- **References (Section 4.4.7)**

Executive Summary

The DWH incident resulted in a large, continuous release of oil at a depth of 1,500 meters in the northern Gulf of Mexico over a period of 87 days before the well was capped. The spill exposed many different and highly diverse biological communities throughout the water column to oil. Prior to this spill, many of the biota in this area had not been well studied. After the spill began, the Trustees conducted a large, sustained, and multifaceted oceanographic field program, including more than 40 cooperative studies with BP that involved multiple oceanographic research vessels, remotely operated underwater vehicles, aircraft, satellite resources, and other specialized equipment. This effort produced a large inventory of physical, biological, and chemical data.

However, because the impacted area is vast and empirical data were ephemeral, the Trustees could not fully characterize the contamination in space and time. As such, the Trustees quantified injury to water column biota by combining available empirical data with several different modeling analyses.

The area of oil observed floating on the ocean surface for the duration of the spill was quantified using remote sensing imagery. The volume of water in the subsurface mixed zone was quantified using empirical chemistry data collected under the footprint of the floating oil. The number of biological organisms killed due to direct exposure to the slick or lethal concentrations of polycyclic aromatic hydrocarbons (PAHs) in the upper water column was calculated using data synthesized from Natural

Resource Damage Assessment (NRDA)-specific field studies, historical collections, NRDA toxicity testing studies, and the published literature. The spill resulted in a surface slick that covered a cumulative area of at least 112,100 square kilometers (43,300 square miles) for 113 days in 2010. Via mixing due to winds and waves, the average daily volume of water affected by surface oil slicks was 57 billion cubic meters (15 trillion gallons). This occurred in an area of high species diversity during a time of year (spring and summer) when seasonal productivity peaks in the northern Gulf of Mexico. The Trustees quantified the direct kill and foregone production of fish and invertebrates exposed to DWH oil in the surface slick and the subsurface mixed zone. The exposure resulted in the death of between 2 and 5 trillion fish larvae and between 37 and 68 trillion planktonic invertebrates.

The Trustees used a combination of modeling and empirical data to quantify the volume of contaminated subsurface water in the cone of rising oil, the volume of contaminated subsurface water in the deep water plume, the amount of small oil droplets found in subsurface "clouds," and the amount of dissolved contaminants. The NRDA sampling in the deep water highlights the diversity and abundance of animals exposed to oil in the deep pelagic waters of the Gulf of Mexico. The Trustees quantified the direct kill of fish and invertebrates exposed to DWH oil both in the rising cone of oil and in the deep water plume. They also investigated foregone production for a critical subset of these species. The exposure resulted in the death of between 86 million and 26 billion fish larvae and between 10 million and 7 billion planktonic invertebrates.

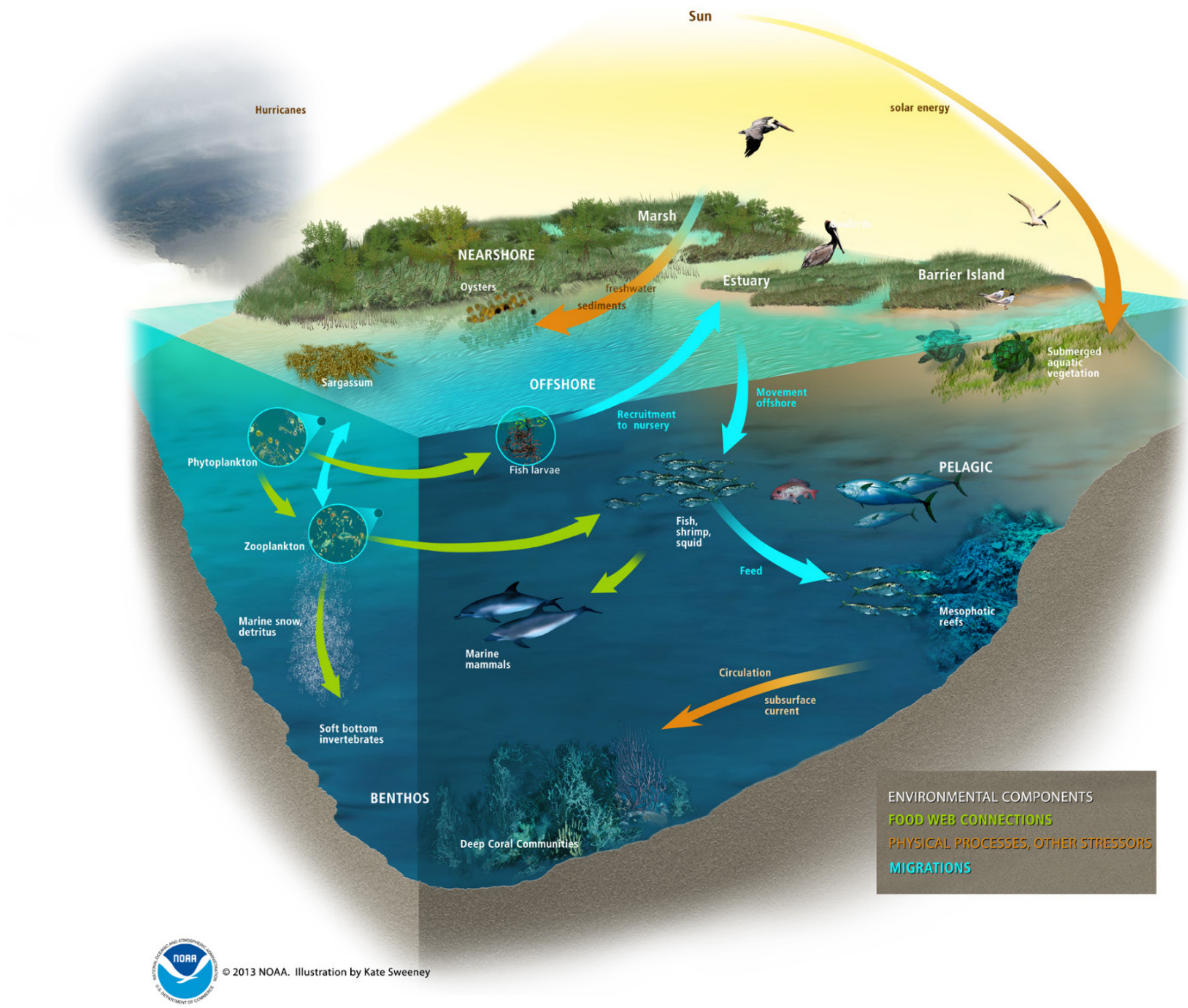
The Trustees also quantified injury to *Sargassum,* a brown marine algae that creates essential habitat for invertebrates, fish, birds, and sea turtles. The Trustees quantified both the lost area of *Sargassum* that resulted from direct oiling and the foregone growth that resulted from this exposure. Heavy oil (greater than 5 percent coverage) affected 23 percent of the *Sargassum* (873 to 1,749 square kilometers) in the northern Gulf of Mexico, resulting in a range of lost *Sargassum* area from foregone growth between 4,524 and 9,392 square kilometers. The Trustees did not quantify lethal and sublethal effects to *Sargassum*-dependent fish and decapods, but include in this section a qualitative discussion about the effects of habitat loss and direct oil exposure to these animals.

4.4.1 Introduction and Importance of the Resource

The Gulf of Mexico waters support a wide variety of organisms, including plankton, more than a thousand known fish species at different life stages (Felder & Camp 2009), mobile invertebrates (such as shrimp, crabs, and squid), sea turtles, seabirds, and marine mammals. These organisms, among others, play important ecological roles. For instance, they serve as prey or predators in the food web, and they cycle and transport nutrients both horizontally (between nearshore and offshore areas) and vertically (between the surface and deep water). Many fish and crustaceans support robust commercial and recreational fisheries. *Sargassum* is an important offshore habitat at the surface for juvenile fish and turtles, providing both shelter and food. *Sargassum* is a key habitat of the ecosystem in the northern Gulf of Mexico, providing the only naturally occurring floating structure in an otherwise featureless open ocean. Figure 4.4-1 illustrates biological communities included in the water column section and indicates key ecosystem processes; Figure 4.4-2 illustrates types of fauna that depend on and use *Sargassum,* including fish, sea turtles, birds, and marine mammals.

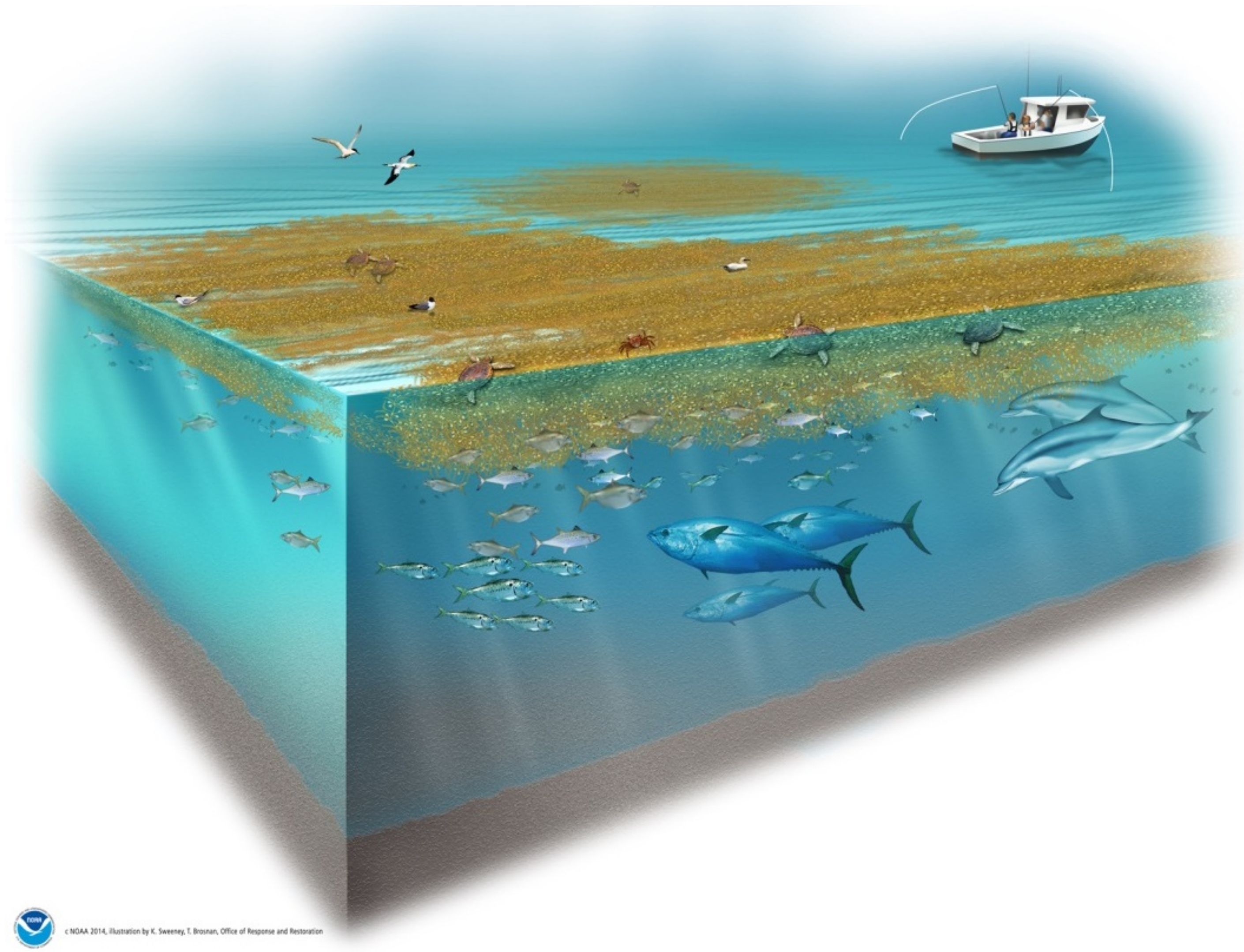
Following the DWH blowout, the oil and dispersants that spread throughout the water column in the deep sea, offshore regions, and nearshore regions impacted these productive and diverse environments (Section 4.2, Natural Resources Exposure). Animals were bathed in a fluid environment that contained surface oil, oil droplets, dissolved oil, dispersants, and elevated concentrations of PAHs. These organisms may have ingested contaminated water, food, and particles; had contaminated water flow over gills; and come into direct contact with extensive oil slicks. Physical processes, such as convergent currents and fronts that play a role in transporting, retaining, and concentrating organisms and *Sargassum*, are the same processes that act to concentrate oil, thus increasing the exposure of organisms to oil. Sunlight, essential for fueling photosynthesis that results in highly productive surface waters, also acts synergistically with PAHs to increase oil toxicity (Section 4.3, Toxicity).

This section addresses injuries to water column resources during and after the spill. The working definition of "water column" here includes virtually everything aquatic or dependent on aquatic systems extending from the shoreline out to deep waters. Not included are sediment-associated benthic communities, birds, sea turtles, and marine mammals, which are addressed in other sections of this document.



Source: *Kate Sweeney for NOAA.*

Figure 4.4-1. Illustration depicting the biological communities included in this section and indicating key ecosystem processes. Depicted here are the various areas of the water column, including estuary and offshore/oceanic areas. Green arrows indicate foodweb connections, blue arrows show migrations of biota from one zone to another, and orange arrows show physical processes that influence the biological communities.



Source: *Kate Sweeney for NOAA.*

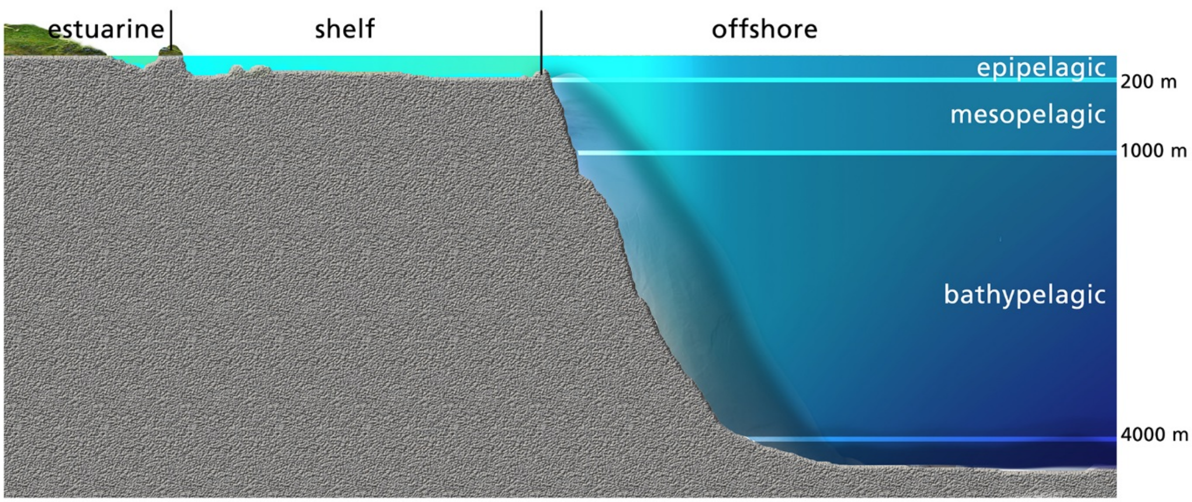
Figure 4.4-2. Illustration of *Sargassum* and associated fauna, including fish, sea turtles, birds, and marine mammals. *Sargassum* is a brown algae that forms a unique and highly productive floating ecosystem on the surface of the open ocean.

4.4.1.1 Water Column Areas and Zones

The northern Gulf of Mexico water column is composed of various habitats, ranging from shallow estuarine waters to dark, deep water environments. Many physical and chemical features (or characteristics) govern these habitats. Examples include light; depth; temperature; pressure; salinity; currents; freshwater inputs; and transport of organic matter, nutrients, and sediments. Horizontal and vertical zones of the water column are outlined below and illustrated in Figure 4.4-3.

Horizontally, the water column can be divided into three main areas: 1) the ***estuarine area*** extending from the barrier islands inward; 2) the ***shelf/neritic area*** over the continental shelf, extending from the barrier islands to the continental shelf break; and 3) the ***offshore/oceanic area*** extending from the shelf break outward.

Vertically, the water column is governed by light, depth, temperature, and pressure. The three main depth zones are: 1) the ***epipelagic zone***, in the upper 200 meters of the water column, where there is enough light for photosynthesis to occur; 2) the ***mesopelagic zone*** from the bottom of the epipelagic zone to approximately 1,000 meters beneath the ocean surface, where some light penetrates, but not enough to fuel photosynthesis; and 3) the ***bathypelagic zone*** from approximately 1,000 to 4,000 meters in depth. Without any sunlight, the bathypelagic zone is dark and cold and is under high pressure due to its depth.



Source: *Kate Sweeney for NOAA.*

Figure 4.4-3. The horizontal and vertical zones of the water column in the northern Gulf of Mexico. Horizontally, the water column can be described in three main areas: the estuarine area (barrier islands inward), the shelf/neritic area (barrier islands to shelf break), and the offshore/oceanic area (shelf break outward). Vertically, the water column includes three main depth zones: the epipelagic zone (0–200 meters beneath the ocean surface), the mesopelagic zone (200–1,000 meters deep), and the bathypelagic zone (1,000–4,000 meters deep).

4.4.1.2 Water Column Species

The water column in the northern Gulf of Mexico provides a large and expansive habitat for a diverse community of species, all of which make up an interconnected and complex food web (Figure 4.4-4;

Chapter 3, Ecosystem Setting). At the bottom of the food web are phytoplankton and zooplankton, which are important food sources for many species of fish and crustaceans (i.e., planktivores). In turn, these planktivores are food for larger predatory species such as tuna and sharks. Table 4.4-1 lists different living marine resources found in the water column, from microscopic bacteria to large predatory fish. The table also describes these resources' importance in the ecosystem and their connection to different Gulf habitats.

Table 4.4-1. Description of selected water column resource groups found in the Gulf of Mexico and their importance to the Gulf ecosystem.

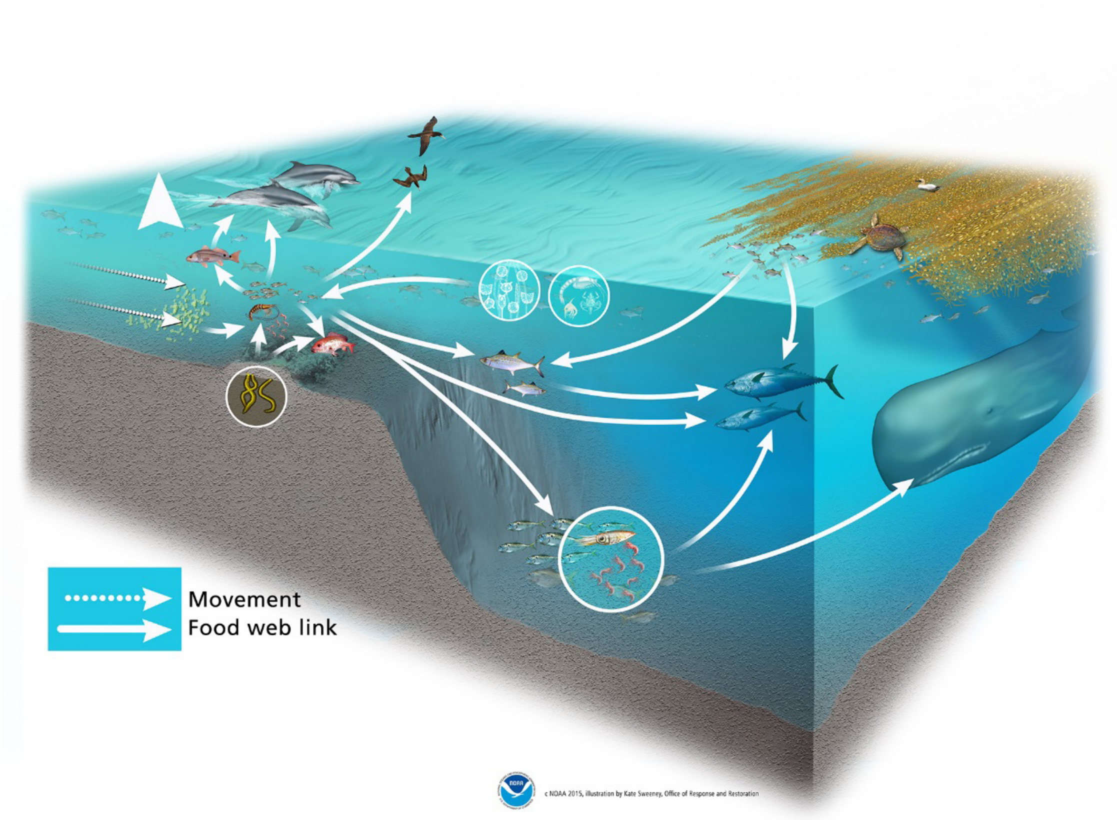
Water Column Resources	Description of Resource
Bacteria	Bacteria are single cell organisms without cell nuclei. Abundant throughout the water column, they serve as important components of the microbial food web (Miller 2004).
Phytoplankton	Phytoplankton are small single cell algae found in the photic zone of the water column, requiring sunlight and nutrients to grow. Phytoplankton abundance typically varies seasonally. Common types of phytoplankton include diatoms and dinoflagellates (Miller 2004). Phytoplankton are the chief "primary" producers in the water column and are an important food source at the base of the marine food web. Phytoplankton contribute to "marine snow," a term used to describe dead and decaying organic detritus falling through the water column (Miller 2004).
Zooplankton	Zooplankton are small, free-swimming animals found within all zones of the water column. Common types of zooplankton include single-celled protozoans, such as foraminifera; gelatinous zooplankton, such as jellyfish; annelids, such as polychaetes; molluscs, such as pteropods; crustaceans, such as copepods; and vertebrates, such as larval fish (Miller 2004). Zooplankton are considered "secondary" producers, feeding on phytoplankton and smaller zooplankton, and they are an important food source for fish and invertebrates. Zooplankton serve as conveyors of energy vertically in the water column, transferring organic carbon and nutrients from the surface waters to the deep water environments. This downward transfer occurs both by active transport (e.g., daily vertical migration) and passive transport (e.g., the sinking of fecal pellets) (Ducklow et al. 2001).
Estuarine-dependent water column species	Estuarine-dependent species include more than 250 species. Representative species ranging from invertebrate secondary producers (e.g., various shrimp and crabs) to low trophic level consumers (e.g., menhaden, anchovies, and striped mullet) to higher trophic level predators (e.g., Atlantic croaker, spotted or speckled seatrout, red drum, striped mullet, sand seatrout, black drum, sheepshead, southern flounder, and some species of shark) (O'Connell et al. 2005). These species are found in estuaries, over the shelf, and in the open ocean, with different life stages typically using different habitats. Estuarine-dependent species may be obligate (i.e., without the estuarine habitat, the species would be unable to survive and/or reproduce) or facultative (i.e., the species may derive a benefit from use of the estuarine habitat, but do not require such use for survival or reproduction). Estuarine-dependent species connect the estuarine and oceanic systems (Able 2005; Able & Fahay 1998; Day Jr. et al. 2013), and are an important food source for the pelagic food web.
Coastal and oceanic epipelagic water column species	Coastal and oceanic epipelagic water column species are those that spend their entire lives on the continental shelf or in the offshore environment, and typically within the epipelagic zone (less than 200 meters below the surface). Species include smaller forage fish (e.g., anchovies, herrings, and sardines) and large predatory fish (e.g., mackerels, tunas, jacks, and sharks). Some large oceanic species, such as tuna, occupy both the surface and mid-water portions of the

Water Column Resources	Description of Resource
	water column (Block et al. 2001; Teo et al. 2007), providing a link between these areas in the food web. All life stages, including eggs and larvae, are important food sources for higher trophic-level organisms.
Mid and deep water column species	Mid-water and deep water column species, found within the mesopelagic zone of the water column (200 to 1,000 meters below the surface), are adapted to little or no light and low food availability. Mesopelagic fishes include lanternfish, bristlemouths, and hatchetfish (Hopkins & Sutton 1998; Quintana-Rizzo et al. 2015). Mesopelagic invertebrates include shrimp, mysids, and squid (Hopkins & Sutton 1998; Passarella & Hopkins 1991; Quintana-Rizzo et al. 2015). The mesopelagic community typically exhibits diel vertical migration, feeding on zooplankton in the uppermost 200 to 300 meters of water at night (Hopkins et al. 1994; Hopkins & Sutton 1998; Lancraft et al. 1988). This migration contributes to the vertical transport of organic matter between the epipelagic zone and the mesopelagic zone. These species are prey items for larger pelagic species such as tunas and billfishes.
Continental shelf reef fish	Shelf reef fish, found on both natural and artificial reefs on the continental shelf, include larger species (e.g., triggerfish, amberjacks, groupers, and snappers) and small cryptic fish (e.g., blennies) (Addis et al. 2013; Dance et al. 2011). Reef fish are recreationally, commercially, and ecologically important and many species are considered overfished stocks (NOAA 2015).
Sargassum	*Sargassum* is a brown alga that floats on the ocean surface. It is a source of primary production and provides habitat and food for sea turtles, marine birds, fish, and invertebrates. It also fills a critical role in nutrient cycling and sedimentation for nearby ecosystems. Designated as Essential Fish Habitat, it provides fish larvae and juveniles protection from predators. It also provides nursery habitat for many important fishery species (e.g., dolphinfish, triggerfishes, tripletail, billfishes, tunas, and amberjacks) and for ecologically important forage fish species (e.g., butterfishes and flyingfishes) (Powers 2012).

4.4.1.3 Ecological Relationships and Processes

Although expansive, the species and habitats of the northern Gulf of Mexico are linked through chemical and physical processes and biological relationships (Chapter 3, Ecosystem Setting). Foodweb dynamics; the movement of organisms between habitats; and the transport of nutrients, sediments, and other materials vertically and horizontally all play a role in the structure and function of the Gulf ecosystem (Chapter 3, Ecosystem Setting).

Predator-prey relationships are dynamic and create an interconnected web of organisms, with energy flowing from primary producers, such as phytoplankton, through a number of trophic linkages to top predators, such as tuna (Figure 4.4-4; (e.g., Althauser 2003; de Mutsert et al. 2012; Masi et al. 2014; Tarnecki et al. 2015). Figure 4.4-4 shows a highly simplified food web, depicting a couple dozen of the thousands of species in the Gulf of Mexico. The figure's simplified depiction does not illustrate how many species occupy different positions in the food web as they grow. For example, a given species of fish may be consumed by certain animals when it is small, but then consume those same animals when it grows to become an adult. The diversity of communities in the water column, the sometimes shifting trophic linkages, and the wide variety of interactions mean that perturbations—such as an injury to one or more components of the food web—may have broader direct, indirect, and sometimes non-intuitive ecological consequences (Fleeger et al. 2003; Fodrie et al. 2014; Peterson et al. 2003; Pimm et al. 1991; Tarnecki et al. 2015).



Source: Kate Sweeney for NOAA.

Figure 4.4-4. Simplified foodweb diagram of the shelf and offshore Gulf of Mexico water column.

The active movement of species between habitats is another important ecological characteristic of the Gulf ecosystem (Chapter 3, Ecosystem Setting). As discussed above, some estuarine-dependent water column species move from nearshore to offshore during their life cycle, linking these two areas and their respective food webs. In addition, some species of zooplankton, fish, and other invertebrates migrate vertically in the water column, transporting energy and materials between the surface and deep water zones.

Nutrients, sediment, and organic matter are also transported horizontally and vertically through water movement (Chapter 3, Ecosystem Setting). Currents and winds move water horizontally, connecting the highly productive and nutrient-rich waters of the coastal areas with the more oligotrophic (i.e., lower nutrient) offshore waters; sinking detritus transports organic matter from the surface to deep waters. This detritus includes plant and animal material, marine snow, and fecal pellets.

4.4.2 Approach to the Assessment

Key Points

- A wide diversity of water column species was exposed over a large area and through many pathways. These interactions' complexity necessitated an assessment approach that employed an array of datasets and analyses to characterize DWH oil exposure and subsequent injuries to water column organisms.
- The Trustees applied a combination of field, laboratory, remote sensing, and numerical modeling approaches.
- A fish-health field survey and analysis of fisheries-independent datasets were conducted to determine community level and physiological effects in the water column that were caused by the DWH oil spill.

As discussed elsewhere in this report (Section 4.3, Toxicity), natural resources may be adversely affected via different exposure pathways: either directly (e.g., toxic effects of oil on an exposed species) or indirectly (e.g., through loss of spawning habitat or reductions in prey availability caused by the spill) (Fleeger et al. 2003; Fodrie et al. 2014; Peterson et al. 2003). When natural resources are injured, cascading ecological effects can result (Fleeger et al. 2003; Fodrie et al. 2014; Peterson et al. 2003). These effects include changes in ecological structure (such as altering the abundance or presence of organisms that comprise the community in an area) and ecological functions (such as altering the flow of nutrients and energy). This document's water column injury assessment takes fundamental ecosystem relationships and processes into consideration.

To characterize the oil exposure and subsequent injury to water column organisms, the Trustees used both field and laboratory data. The Trustees collected data on the fate and transport of the oil and on the abundance and distribution of organisms in the water column. Field studies were conducted to document environmental conditions, evaluate exposure, and assess the condition of biological resources. Numerous toxicity tests were conducted in laboratories to determine the toxicity of MC252 oil to various life stages of Gulf water column species (Section 4.3, Toxicity). All of this information—combined with hydrodynamic, biological, and toxicological modeling—was used to estimate the nature and extent of injuries to water column species.

Because of the diversity and complexity of the Gulf ecosystem, the vast area of the northern Gulf of Mexico affected by the oil spill, and the practical challenges of performing scientific studies in logistically challenging habitats (e.g., deep waters with safety concerns), it was not possible to study every species, habitat, and ecological process. Therefore, the Trustees applied an understanding of fundamental ecological relationships and processes to focus on representative species and habitats, using study results to make reasonable scientific inferences about natural resources and services that were not explicitly studied. As described in Ecosystem Settings (Chapter 3), the Trustees relied on this understanding of ecological relationships to develop potential restoration approaches.

This section presents the Trustees' approach to the water column injury assessment. Section 4.4.2.1 presents the conceptual model for the pathway and exposure of water column resources to DWH oil.

Section 4.4.2.2 presents the integrated water column resource assessment approach, including the methods for injury quantification of fish and invertebrate species. Section 4.4.2.3 presents the *Sargassum* assessment approach, and Section 4.4.2.4 documents additional biological assessment approaches not covered in the earlier sections.

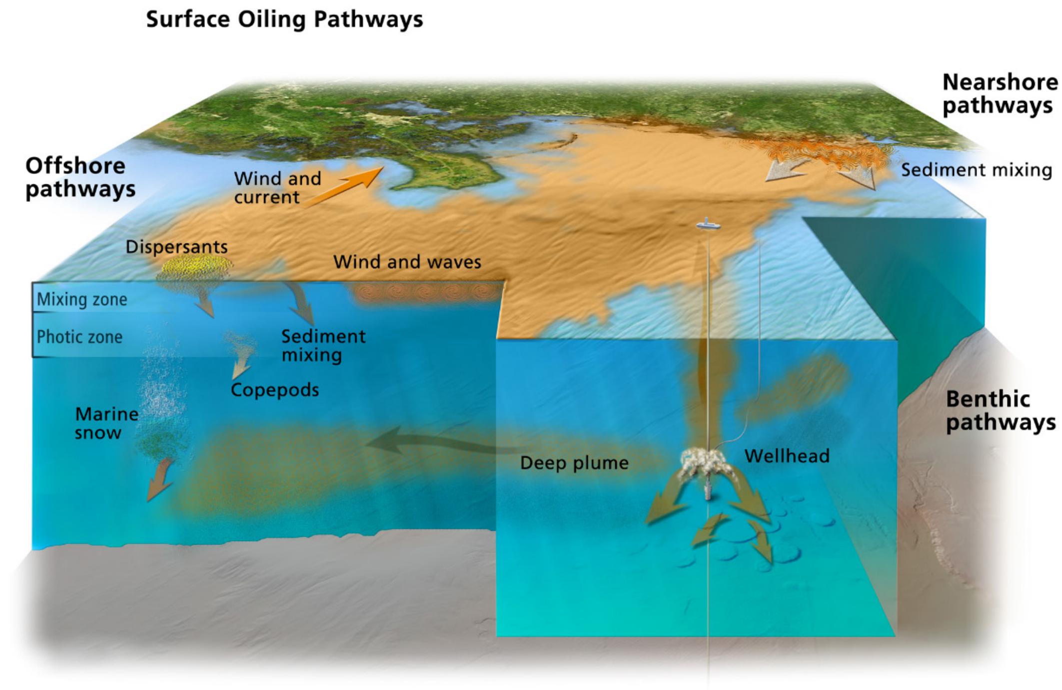
4.4.2.1 Pathways for Exposure

The DWH oil spill impacted a large expanse of the northern Gulf water column, extending from the biologically diverse deep-sea environment to the highly productive coastal waters (Section 4.2, Natural Resources Exposure). As Figure 4.4-5 shows, oil that discharged from the wellhead transported through the water column via five main pathways:

- Oil droplets released from the wellhead rose up through the water column, resulting in the **rising oil plume.**
- Oil was dispersed, both physically and chemically, near the wellhead, and a layer of dissolved oil and oil droplets was trapped at depth and moved with deep-sea currents, resulting in a **deep water oil plume.**
- Oil that reached the surface waters was transported horizontally by winds and currents over great distances, resulting in a large **surface slick** that eventually reached shorelines.
- Oil within the surface slick became mixed in the upper portions of the water column due to natural physical processes and cleanup response actions, resulting in a **subsurface entrained layer.**
- Oil droplets within the water column became attached to particulates, such as detritus or marine snow, and were transported to the benthos and sometimes resuspended, resulting in a **downward flux of particulates**.

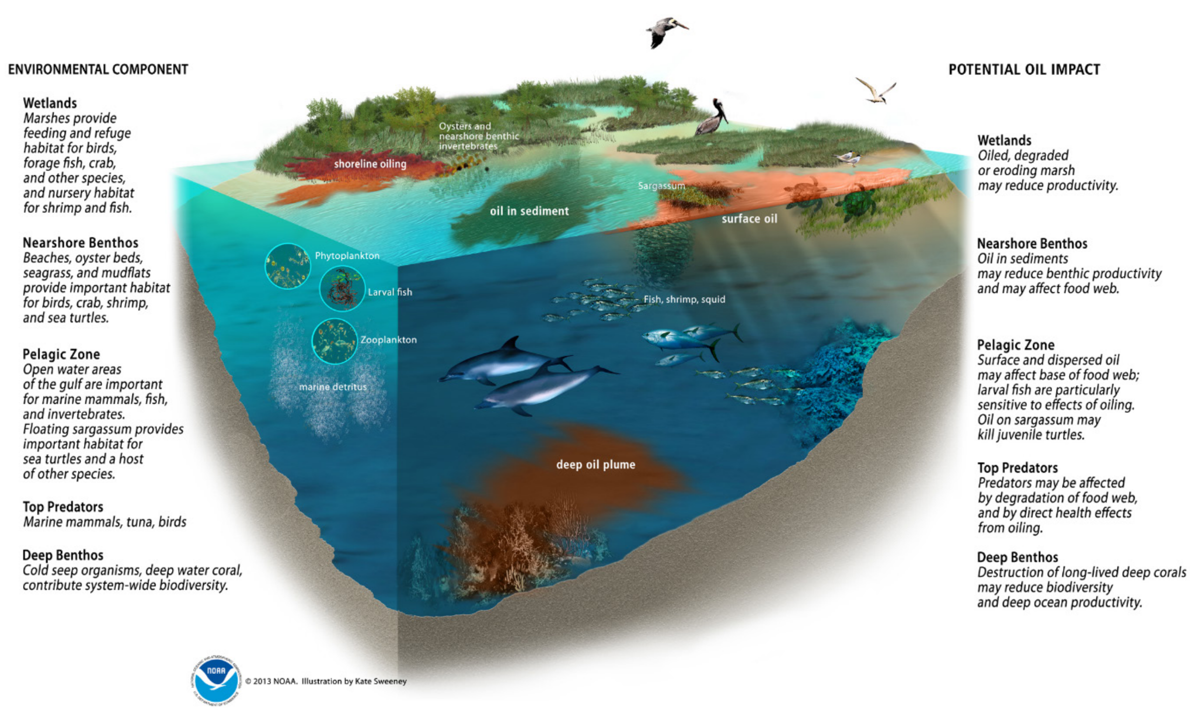
The injury quantification focuses on the first four pathways. The last pathway exposes both water column and benthic resources and is examined in Section 4.5 (Benthic Resources).

Water column resources were exposed to oil in various forms, including oil droplets; dissolved hydrocarbons; oil attached to particulates, such as marine snow; oil-contaminated food; and weathered oil in the surface slick (Section 4.2, Natural Resources Exposure). Figure 4.4-6 illustrates the distribution of key water column resources in relation to their likely oil exposure pathway and potential oil impact. Toxic effects of oil to phytoplankton, zooplankton, and many species of fish and crustaceans have been extensively documented in the literature and NRDA-funded studies (Section 4.3, Toxicity). Ultraviolet (UV) light from the sun is known to increase the toxicity of oil for many species in the upper water column (Section 4.3, Toxicity). Physiological endpoints, such as reduced growth, impaired reproduction, and adverse health effects, have also been observed in the field (Section 4.4.2.2) and supported by laboratory experiments (Section 4.3, Toxicity). Lethal and sublethal impacts at the organismal level could result in larger, population- or community-level effects, such as shifts in abundance, trophic structure, and community structure (Fleeger et al. 2003; Fodrie et al. 2014; Peterson et al. 2003).



Source: *Kate Sweeney for NOAA.*

Figure 4.4-5. Illustration of the DWH oil release pathways.



Source: *Kate Sweeney for NOAA.*

Figure 4.4-6. Water column resources and potential oil impacts.

4.4.2.2 Integrated Water Column Resource Assessment Approach

The Trustees conducted an integrated water column resource analysis to determine and quantify injuries to northern Gulf water column resources. Section 4.4.2.2.1 describes the set of related methods the Trustees used to address the surface slick and subsurface mixed zone; Section 4.4.2.2.2 describes the different set of related methods used to evaluate the rising plume and the deep water plume.

The Trustees' quantification of injury to water column biota is focused on larval fish and planktonic invertebrates because these early life stages are more sensitive to oil exposure (Section 4.3, Toxicity). Injury to adult life stages was evaluated but not quantified. Figure 4.4-7 presents an overview of the Trustees' water column assessment approach, and the following paragraphs describe specific approaches to evaluating empirical data and modeling results. See Section 4.4.2.4 for additional studies the Trustees conducted to evaluate community/population level and physiological effects of the DWH oil spill on water column resources.

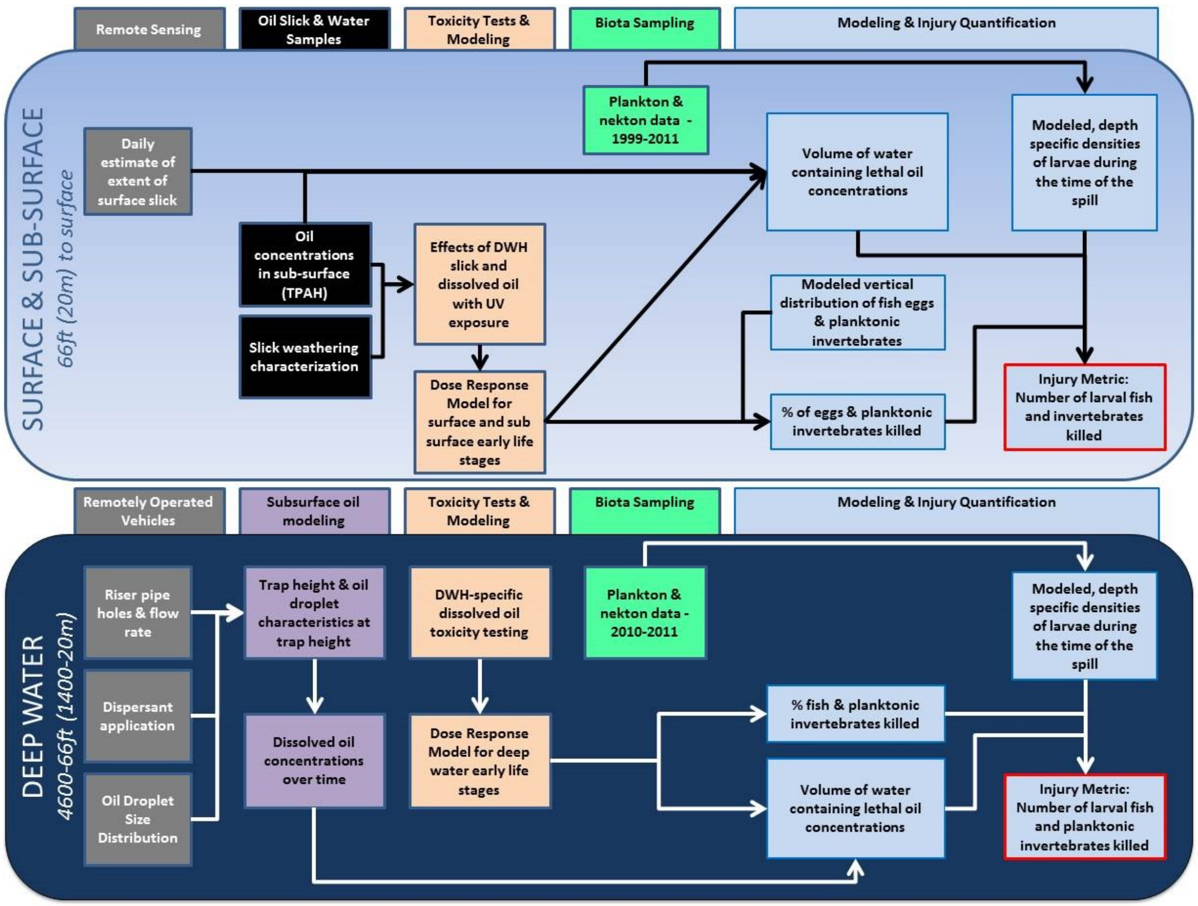


Figure 4.4-7. Approach taken to assess injury to water column habitat and biological resources.

4.4.2.2.1 Surface Slick and Subsurface Mixed Zone

Data from remote sensing, combined with empirical chemistry, were used to quantify, for the duration of the spill, both the area of surface floating oil and the volume of water under the slick that was toxic to water column organisms. To determine the impact of the oil on the biological community in the upper water column, the Trustees used biological datasets derived from the following sources (French McCay et al. 2015c):

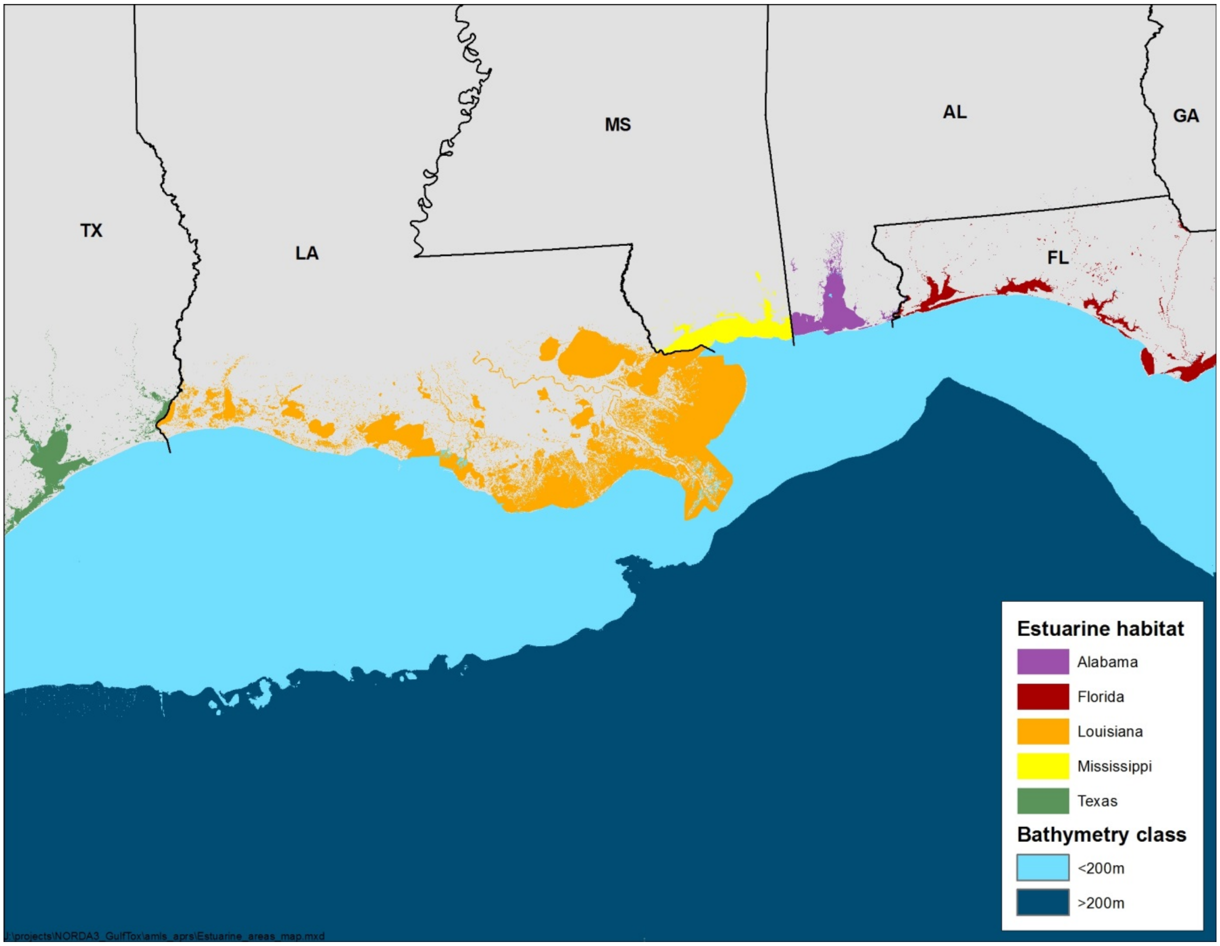
- Historical surveys by the National Marine Fisheries Service (NMFS) Southeast Area Monitoring and Assessment Program (SEAMAP).
- A plankton survey by the Dauphin Island Sea Laboratory's Fisheries Oceanography of Coastal Alabama (FOCAL) program.
- The DWH NRDA plankton program (French McCay et al. 2015c).

The historical data and DWH NRDA data were used to predict the density of eggs and larvae present in surface waters containing floating oil. NRDA toxicity program data (Section 4.3, Toxicity) were then used to generate the percent of eggs and larvae killed by this oil.

The Trustees quantified injury in the surface slick for three distinct zones, which are defined below and shown in Figure 4.4-8:

- An offshore zone, defined as areas where the water is more than 200 meters deep.
- A shelf zone, defined as areas seaward of barrier islands where the water is less than 200 meters deep.
- An estuarine zone, defined as all waters landward of barrier islands.

Distinct offshore, shelf, and estuarine assessments were performed because these areas have different species distributions, toxicity studies, and satellite imagery. However, the approach to quantify injury was generally the same across all three zones.



Source: *Travers et al. (2015a); (2015b). Estuarine waters from Cowardin et al. (1979); land modified in Louisiana using Couvillion et al. (2011). Bathymetry adapted from NOAA (2006) and NOAA (2010).*

Figure 4.4-8. Map of the north central Gulf of Mexico, distinguishing the offshore zone (depth greater than 200 meters), the shelf zone (areas seaward of barrier islands with water depth less than 200 meters), and estuarine waters (shallow waters inside the nearshore barrier islands).

Surface Oil Observations and Mapping

As discussed in Section 4.2 (Natural Resources Exposure), the Trustees analyzed remote sensing data to delineate the extent of DWH oil slicks. For the water column analyses, the Trustees relied heavily on synthetic aperture radar (SAR) images for estimating the daily spatial extent of surface oil, because SAR has the greatest spatial and temporal coverage of the available remote sensing instruments (Garcia-Pineda et al. 2009; Graettinger et al. 2015). For days when SAR images were unavailable, the areal extent of surface oil was assumed to be the average of the slick area from the previous day and the following day. One limitation in the SAR data is that some days have only a single SAR image that covers just a portion of the Gulf oil slicks (Graettinger et al. 2015). When the SAR data had limited spatial coverage, the Trustees only used available images to estimate the areal extent of the slick—an approach that underestimated areas on these days.

In addition to SAR imagery, the Trustees analyzed imagery from several other airplane- and satellite-mounted sensors, including data from the National Aeronautics and Space Administration's (NASA's) Moderate Resolution Imaging Spectroradiometer (MODIS) and NASA's/the U.S. Geological Survey's (USGS's) Landsat Thematic Mapper. These sensors collect data at wavelengths that include visible spectra (similar to a camera) and infrared (including thermal infrared that detects when an oil slick appears warmer or colder than the surrounding sea). The Trustees integrated the data from multiple sensors into a model that not only estimated the oil slick areal extent, but also estimated coverage of thicker oil (or emulsions) and thinner oil (Garcia-Pineda et al. 2009; Graettinger et al. 2015). Some analyses of injury in the upper water column relied on this integrated model of remote sensing data.

Empirical Chemistry Data

As described in greater detail in Section 4.2 (Natural Resources Exposure), the Trustees collected and analyzed numerous samples of the oil floating throughout the northern Gulf of Mexico to characterize the surface slick chemistry. Floating oil varied in age, from relatively fresh oil that had recently risen from the wellhead to the surface to DWH oil that had remained in the water column for weeks or longer and eventually transported to the surface offshore of marshes and beaches. The chemistry of the floating oil is detailed in Section 4.2 (Natural Resources Exposure).

Multiple sampling studies collected water samples at different depths to assess water column oil concentrations. The subsurface water column injury quantification used a dataset compiled from sampling data documented in multiple sources, including the Trustees' NRDA, the BP NRDA, the Response (cleanup), and the BP Public website (Travers et al. 2015a). The Trustees used this dataset to estimate the distribution of oil in the upper mixed zone of the water column in the offshore, shelf, and estuarine areas. Section 4.4.3.2, below, describes the results of this analysis.

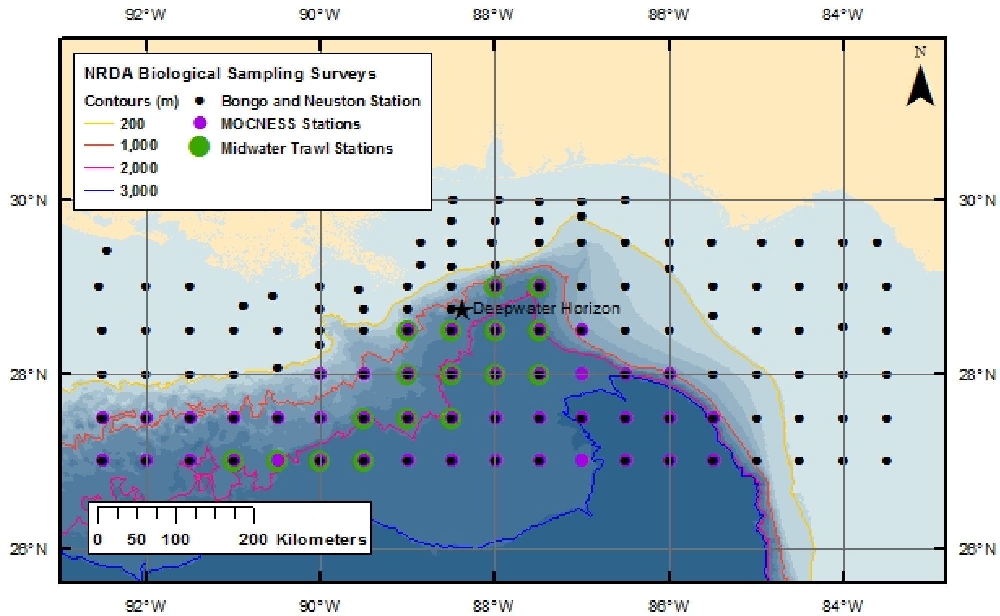
Oil is a complex mixture made up of thousands of organic compounds (NRC 2003). Oil concentrations in the environment are often described in terms of the concentrations of a limited set of compounds found in the oil. Typically, when assessing the effects of oil, researchers focus on the concentrations of PAHs, which are the set of compounds thought to be the most toxic (NRC 2003). The DWH NRDA toxicity testing program generally reported effect concentrations in terms of the sum of 50 PAHs (TPAH50) (Forth et al. 2015; Morris et al. 2015b). Consequently, to assess injuries in the water column resulting from oil and for comparison of toxicity test results, we used TPAH50 to describe oil concentrations.

Empirical Biological Data

To estimate the number of fish and invertebrate species exposed to oil, the Trustees reviewed and analyzed numerous pre-spill data sources. For example, the Trustees reviewed and analyzed 10 years of pre-spill SEAMAP data and technical reports from NOAA and the Bureau of Ocean Energy Management. The data and information from these sources were then used to calculate densities of taxa. In cases where pre-spill data were not available for a given habitat or community, post-spill data were considered. The metadata for the datasets can be found in Table 4.4-2. Maps showing the location of NRDA and SEAMAP sampling are included in Figure 4.4-9, Figure 4.4-10, Figure 4.4-11, and Figure 4.4-12.

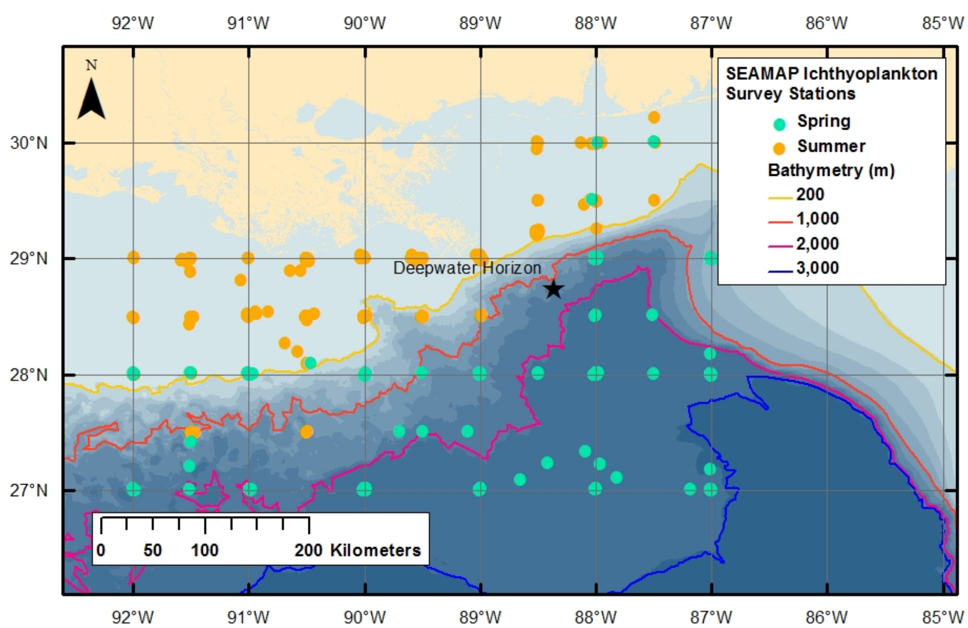
Table 4.4-2. Description of empirical biological datasets used to determine biological densities.

Dataset	Time Period Covered	Description of Dataset
1. SEAMAP Ichthyoplankton Survey	1999–2009	Ichthyoplankton and small juvenile fish densities in the upper 200 meters in shelf and offshore waters
2. SEAMAP Invertebrate Zooplankton Survey	1999–2009	Invertebrate microzooplankton densities (other than decapods) in the upper 200 meters in shelf and offshore waters
3. NRDA Plankton bongo sample data	2011	Decapod larval densities in the upper 200 meters in shelf and offshore waters
4. NRDA Plankton 1 m^2 MOCNESS sample data	2011	Fish and decapod larval densities below 200 meters in offshore waters
5. SEAMAP Shrimp/Groundfish Survey	1999–2009	Juvenile and adult fish and invertebrate densities in the upper 200 meters in shelf waters
6. NRDA 10 m^2 MOCNESS sample data	2011	Micro-nektonic pelagic fish and planktonic invertebrate densities in offshore waters, depths greater than 200 meters
7. NRDA Pisces Midwater Trawl data	2011	Nektonic pelagic fish and invertebrate densities in offshore waters, depths greater than 200 meters
8. Deep Gulf of Mexico Benthos (DGoMB) Survey (Powell et al. 2003; Rowe & Kennicutt II 2009)	2003; 2009	Demersal fish and invertebrate megafauna densities in offshore waters, depths greater than 200 meters
9. NRDA Flying Fish Observations	2011	Juvenile and adult fish in surface waters of shelf and offshore waters
10. Stock Assessment-Based Estimates	1999–2009	Juvenile and adult fish in shelf and offshore waters
11. Estuarine fish and invertebrate densities (Brown et al. 2013)	2013	Estuarine fish and invertebrate densities applicable to waters inside the barrier islands
12. Dauphin Island Sea Lab FOCAL plankton survey	2007–2009	Nearshore (estuarine) larval fish and planktonic invertebrate densities applicable to waters inside the barrier islands



Source: *French McCay et al. (2015c).*

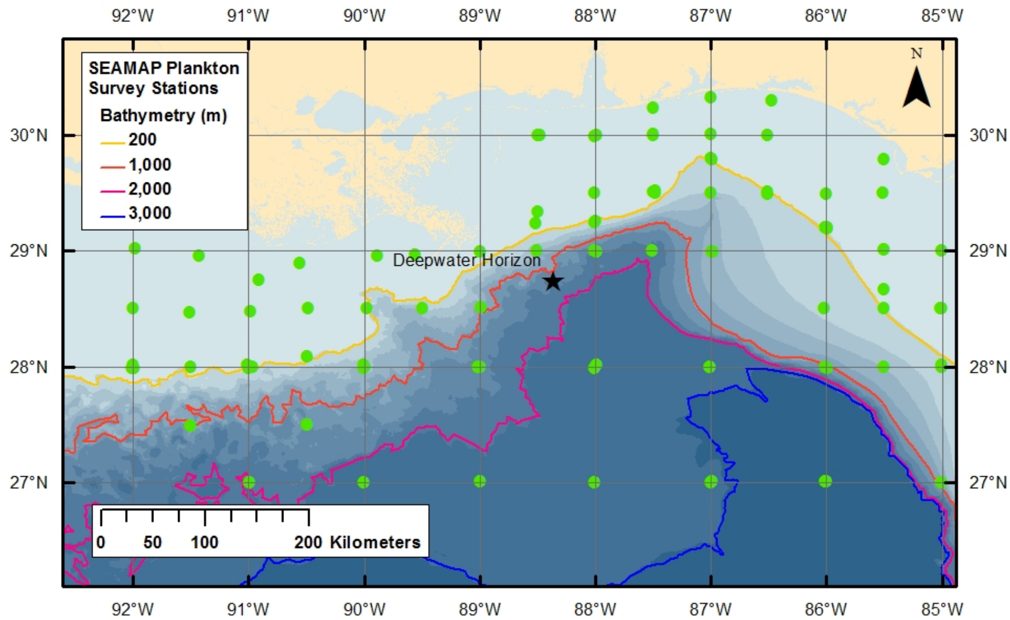
Figure 4.4-9. Location of NRDA biological sampling survey stations. Sampling was primarily conducted using three methods: bongo and/or neuston net tows (black dots), both 1- and 10-square meter MOCNESS sampling nets (purple dots), and midwater trawls (green dots).



Source: *French McCay et al. (2015c).*

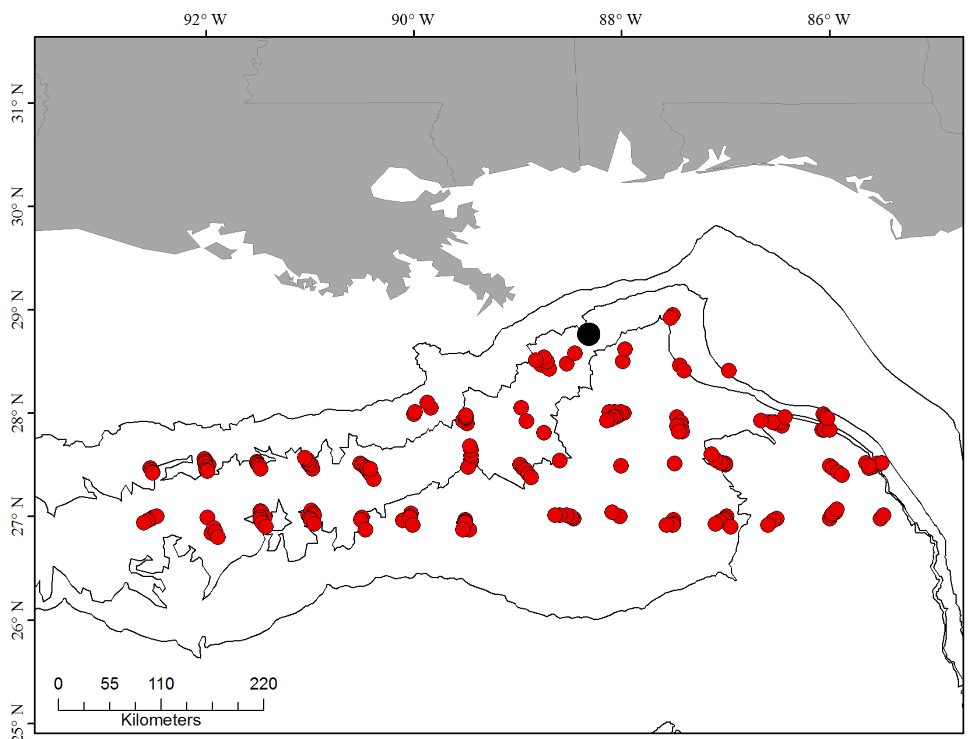
Figure 4.4-10. A portion of the geographic extent and survey station locations of SEAMAP Plankton Survey data used to derive ichthyoplankton densities.

4.4.2 Approach **to the Assessment**



Source: *French McCay et al. (2015c).*

Figure 4.4-11. A portion of the geographic extent and survey station locations used by SEAMAP to calculate invertebrate zooplankton densities.



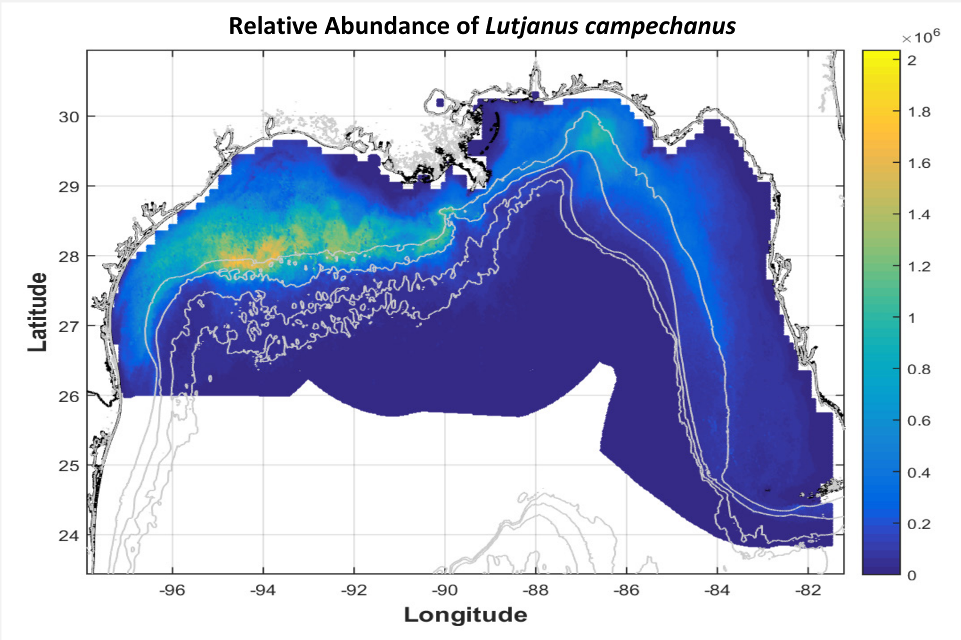
Source: *French McCay et al. (2015c).*

Figure 4.4-12. Locations of samples (red dots) used from the NRDA 10 m^2 MOCNESS surveys, cruise MS7. MC252 Wellhead indicated with the black dot. Black lines represent 200-meter, 1,000-meter, 2,000-meter, and 3,000-meter bathymetric contours.

While these datasets cover a wide range of organisms, data for some groups remain incomplete due to sampling limitations. For example, fast-swimming pelagic species are rarely, if ever, caught in trawls and other sampling gears. Also, most studies only sample smaller fish, typically from age-0 and age-1 year classes. Due to these and other factors, the derived species densities described below and used to calculate injury to water column data should be considered an underestimate of actual species densities.

Generalized Additive Models

Generalized additive models (GAMs) (Hastie & Tipshirani 1990; Wood 2006) are a well-established statistical modeling technique. Using data from historical bongo net sampling across the Gulf of Mexico and a suite of environmental data, GAMs were developed to provide predictions of the relative abundance and distribution of larval fishes in the U.S. Economic Exclusion Zone in the Gulf of Mexico for any day between April 23 and August 11, 2010 (Christman & Keller 2015; Quinlan et al. 2015). Figure 4.4-13 shows the relative abundance of larval red snapper expected in each of more than 400,000 grid cells with a nominal size of about 1.7 square kilometers. The GAMs for red snapper well-represented the seasonal changes in the abundance of red snapper larvae as well as the distributional patterns and how those distributions changed through time. Maps like these, and the data behind them, were used to estimate the abundance and distributional patterns for a variety of larval fishes and invertebrates, and to explore the overlap between surface oil slicks and these organisms.



Source: *John Quinlan, NOAA (2015).*

Figure 4.4-13. Estimated relative abundance of larval red snapper on June 20, 2010, in the U.S. Economic Exclusion Zone. Relative abundance is related to the expected number of larval red snapper per 1,000 square meters based on bongo net sampling. The density values (number per 1,000 square meters) were scaled to the area of the grid cell in this figure. Grey contours depict the 100-, 400-, 1,000-, and 1,500-meter isobaths.

Determining Distribution and Abundance of Taxa and Life Stages

Using SEAMAP ichthyoplankton data from 1999 to 2009, statistical techniques were applied to predict larval densities for the period of the spill. For a subset of species present in the Gulf that were abundant, represented different life history characteristics, or were of particular economic or ecological concern, generalized additive models (GAMs) were developed using the SEAMAP Ichthyoplankton Survey bongo net catch data and spatially and temporally correlated environmental characteristics (e.g., location, depth, temperature). The daily density maps derived from the GAMs were used as baseline densities present during the spill from April to July 2010 (Christman & Keller 2015). For other species or taxonomic groups, the average abundance of that taxon was used. Average abundances represent the daily densities and were generated either seasonally offshore (i.e., spring and summer) or monthly in the nearshore estuarine zone. Though average density estimates were established for each taxon individually, confidence ranges around these averages were large due to the patchy distributions of many planktonic organisms. Thus, where estimates are provided that sum across species, the confidence interval was generated using the pooled data for those taxa.

Many pelagic spawning fish have positively buoyant eggs—some of which were found in the upper mixed layer and interacted with both the surface slick (Gearon et al. 2015) and contaminated water under the slick. The Trustees used models to estimate the vertical distribution of eggs and developing embryos in the water column (Wobus et al. 2015) and to assess eggs' and embryos' exposure to the oil. The exposure calculations were made using the distribution of TPAH50 concentrations and toxicity testing results, as described below.

The vertical distribution of eggs in the upper water column depends on the wind speed at the surface, the diameter of the eggs, and the eggs' density (in grams per cubic centimeter) compared to the water density. When eggs are larger and less dense, their relative concentrations near the surface increase. On the other hand, relative egg concentrations near the surface are reduced with smaller, denser eggs and higher wind speeds, which increase dispersion and push eggs deeper into the water column. To quantify the vertical distribution of eggs as a function of wind speed, egg diameter, and egg density in grams per cubic centimeter, the Trustees used VertEgg—a model that estimates the static distribution of eggs in the water column, but does not simulate the movement of eggs over time (Ådlandsvik 2000; Wobus et al. 2015).

Modeling of Toxicological Effect

To calculate a range of potential toxicity to ichthyoplankton and zooplankton exposed to DWH oil, three species of fish and two species of invertebrates, all indigenous to the Gulf of Mexico, were selected. The sensitivity of these species to DWH oil represented the range of sensitivity (with and without UV light) observed across a wide range of taxa that were tested for the DWH NRDA. For purposes of this analysis, waters not subject to UV-PAH phototoxicity were those at least 20 meters below the surface or turbid enough to preclude significant UV light penetration based on field data (Lay et al. 2015a) for early life stage fish. See Section 4.3 for a more detailed explanation of the Trustees' toxicity program, including explanations of toxicity endpoints and associated acronyms used in this section.

Toxicity in the Absence of Sunlight

The selected species that represent the low and high end of the range of sensitivity in the *absence* of UV light were the more sensitive bay anchovy (*Anchoa mitchilli*) and the less sensitive red drum (*Sciaenops ocellatus*) (Morris et al. 2015c). The concentration that kills 20 percent of the test organisms (Section 4.3, Toxicity)—known as the LC_{20}—for bay anchovy (based on a 48-hour test) and red drum (based on a 72-hour test) are 1.3 and 21.9 µg/L TPAH50, respectively. For invertebrates, the low and high sensitivity species and their corresponding LC_{20} values are copepod (*Arcartia tonsa*; LC_{20} = 33.5 µg/L TPAH50 based on a 96-hour test) and blue crab (*Callinectes sapidus*; LC_{20} = 79.0 µg/L TPAH50 based on a 48-hour test), respectively. These ranges were used to evaluate TPAH50 water column concentrations in waters that do not receive appreciable UV light.

Toxicity in the Presence of Sunlight

Biota near the ocean surface are exposed to sunlight and DWH oil. The Trustees investigated photo-induced toxicity on Gulf early life stage fish and invertebrates and determined that, consistent with the literature, UV light can greatly enhance the toxicity of DWH oil on early life stage organisms (Section 4.3, Toxicity). In fact, the average amount of UV light measured in the Gulf of Mexico during the spill can increase the toxicity of DWH oil by approximately 10 to 100 times over 1 day (Lay et al. 2015b; Morris et al. 2015b). Therefore, a UV adjustment factor was derived to apply to dose-response curves for several fish and invertebrate species. Adjustments to dose-response relationships were made using the average daily integrated UV intensity for the Gulf of Mexico during the spill (1,550 mW-s/cm^2, UV-A, 380 nm; Lay et al. (2015a).

For both fish and invertebrates, two species were selected to represent the low and high end of the range of sensitivity, similar to the approach for habitats without UV light, described above. For the UV-adjusted toxicity, the low and high sensitivity fish and invertebrate species and the magnitude of the increased toxicity based on each adjustment relative to their sensitivity in the absence of UV light are: bay anchovy (14-fold increase), mahi-mahi (*Coryphaena hippurus*; 15-fold increase), copepod (27-fold increase), and blue crab (27-fold increase). UV-adjusted dose-response curves were used to estimate the percent mortality for these species.

In addition to exposure to oil entrained in the water, organisms may also have been exposed to floating oil slicks or sheens through direct contact. As described in Section 4.3, the Trustees determined the toxicity of very thin surface sheens of oil in the presence of varying levels of UV light (Morris et al. 2015a). For purposes of this analysis, a very thin sheen is approximately 1 micron (µm) thick, approximately 40 times thinner than a single human hair. When exposed to the integrated average dose of UV light in the Gulf of Mexico over the course of the spill, the toxicity (percent mortality) of thin surface sheens to red snapper (embryo), bay anchovy (embryo), spotted seatrout (embryo), and mysid shrimp (juvenile) is 85, 89, 100, and 100 percent mortality, respectively. Based on these results, percent mortalities were developed for organisms exposed to the surface slick zone:

- 91 percent mortality—the average across the three fish species—was used for the two UV-exposed representative fish species (bay anchovy and mahi-mahi).

- 100 percent mortality—the result for mysid shrimp—was used for the two representative invertebrate species (copepod and blue crab).

Estimated Mortality from Oil Exposure

To estimate the mortality to fish embryos and invertebrates caused by oil exposure, the Trustees assessed oil concentrations and UV doses encountered during the spill. Applying a Monte Carlo approach (i.e., repeated random sampling) (Robert & Casella 1999) using the egg and TPAH50 concentration distributions described above, the Trustees generated probabilistic estimates of TPAH50 concentrations at different depths that any given egg beneath a surface slick might have encountered over the course of the spill.

For each randomly selected egg, a UV dose was calculated by applying a UV extinction coefficient to the average incident UV at the water surface. Extinction coefficients were based on an average of offshore measurements in the Gulf of Mexico (Lay et al. 2015a). The Trustees used the combination of TPAH50 and UV to calculate the percent mortality using the UV-adjusted dose-response curves for the "sensitive" and "less sensitive" species, described above (Morris et al. 2015b; Morris et al. 2015c). The Trustees used the calculated mortality for each of the thousands of simulated scenarios to estimate the percentages of total mortality over the water column. The Trustees used a similar approach for invertebrates, except that invertebrates were assumed to be evenly distributed vertically in the water column (Travers et al. 2015b).

For the estuarine waters, the Trustees evaluated exposure only to floating oil. The estuarine waters generally contain high concentrations of sediment, and UV light does not penetrate deep in these turbid waters. In this analysis, oil slick toxicity was estimated only to the depth where 10 percent of incident UV light remains, which is approximately 0.2 meters beneath the surface, based on measurements made in Barataria Bay (Lay et al. 2015a). The average mortality was therefore estimated from the water surface to a depth of 0.2 meters beneath the surface (Section 4.3, Toxicity).

Estimated Production Foregone

The production foregone model estimates the lost future growth (i.e., production) that the killed organisms would have produced had they otherwise lived their normal lifespan. It does not include losses that would have occurred from reproduction or additional generations. The biomass of organisms directly killed as the result of the DWH spill represents the weight of the organisms at their death; the production foregone model determines the biomass (additional weight) these organisms would have accrued as they grew from their early life stages into adults or until they died naturally or were harvested. Production foregone uses information on mean growth and survival for each species. Assessing production foregone allows for a more thorough representation of spill-related injuries to water column organisms than would be captured by calculating what is lost by the direct kill alone. Results of the production foregone model are measured in biomass, which can be used to address biological concerns and can be informative when considering restoration needs.

Production foregone was calculated for larvae of 29 fish species that have growth models reported in state, federal, or international stock assessments, including snappers, tunas, mackerels, seatrout, croakers, and billfish. As part of their stock assessment development, the needed growth and mortality rates have been well studied and reviewed by fisheries managers, such that the production models based on these inputs are robust. Invertebrate production foregone was calculated using the growth models of two species: blue crab (*Callinectes sapidus*) and white shrimp (*Litopenaeus setiferus*). Each of these invertebrate species has an available federal stock assessment or has been extensively studied. The white shrimp growth model was applied to shrimp where adults grow to similar size (i.e., approximately 100 to 200 millimeters in length and maturing in approximately 1 year). For crabs, the blue crab growth model was applied to all crabs in that family (i.e., to the family Portunidae). Development of the production foregone model and estimations of production foregone per individual killed are described in French McCay et al. (2015a).

4.4.2.2.2 Rising Oil and Deep Plume

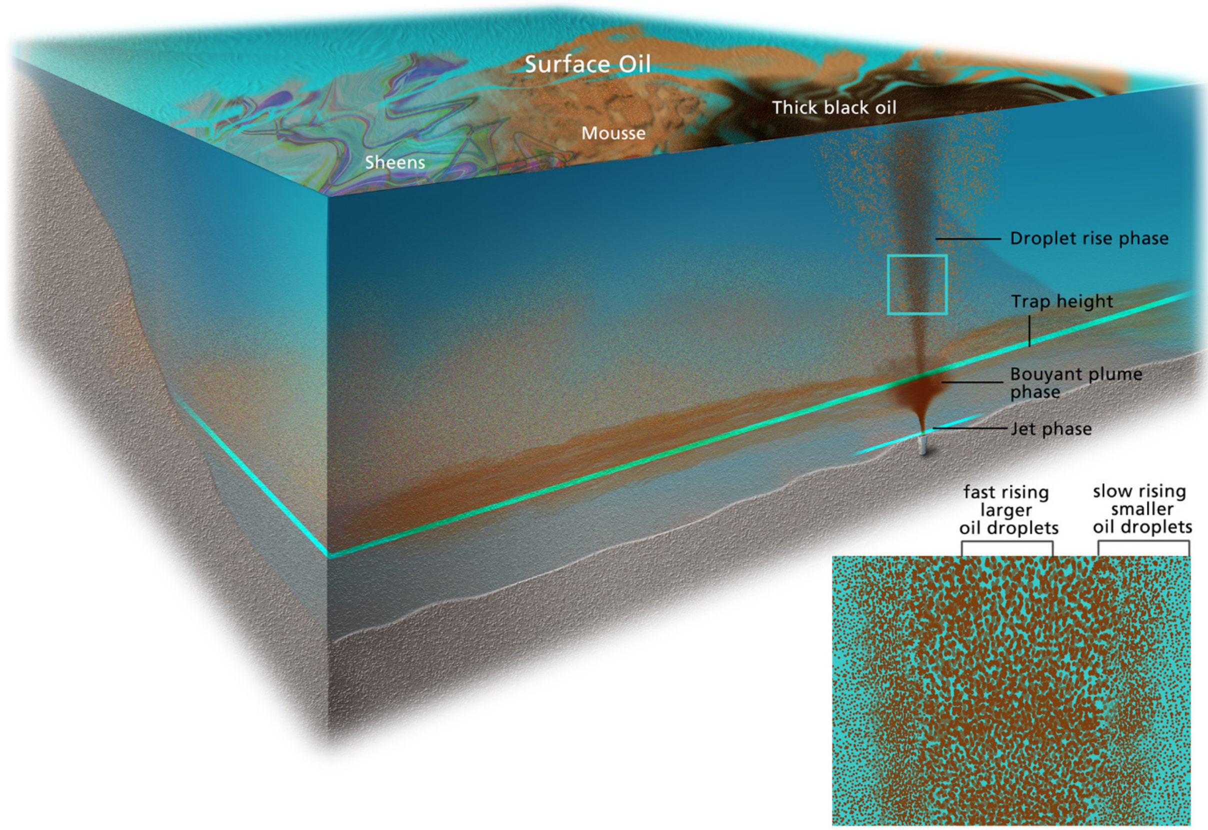
Empirical chemistry data and a highly developed modeling approach were used to quantify the volume of water contaminated with PAHs for the duration of the spill. To determine the impact of the contaminated oil on the biological community in the rising cone of oil and the deep water plume, the Trustees used various biological datasets to predict the density of ichthyoplankton and invertebrate larvae present in the water column. Data derived from the NRDA toxicity program (Section 4.3) were used to generate the percent of larvae killed through contact with the rising oil.

Modeling of Oil at Depth

The Trustees modeled the fate and transport of the rising "cone" of oil from the blowout through the deep water column.

Figure 4.4-14 is a conceptual model of the blowout and rising oil droplet phases whereby oil droplets of various sizes moved upward through the water column. The OILMAPDeep blowout model (Spaulding et al. 2015) evaluates the jet and buoyant plume of the release from the broken riser. The model determines the neutral-buoyancy depth, also known as the "trap height," which is where oil droplets separate and are subsequently transported horizontally by currents and vertically by their individual buoyancies. Models were used to estimate the oil masses and droplet sizes of the released oil droplets. This analysis was based on the U.S. v. BP et al. (2015) findings of 4.0 million barrels of oil released from the reservoir and 3.19 million barrels of oil discharged to the Gulf of Mexico. The amount discharged each day between April 20 and July 15, 2010, was assumed proportional to the daily release volumes estimated by the Flow Rate Technical Group (McNutt et al. 2011).

Oil droplet mass, size, and location estimates from the Spaulding et al. (2015) analysis were used as input to the Spill Impact Model Application Package (SIMAP) oil fate model (French McCay 2003, 2004). SIMAP then evaluated weathering (i.e., dissolution and degradation), movements, and concentrations of oil and components (e.g., PAHs) from the trap height to the ocean surface (French McCay et al. 2015b). SIMAP also predicted TPAH50 concentrations over time in a three-dimensional spatial grid extending from water column depths of 1,400 meters beneath the surface up to 20 meters beneath the surface. The uppermost 20 meters were evaluated as part of the surface layer analysis.



Source: *Kate Sweeney for NOAA.*

Figure 4.4-14. Conceptual model of the blowout and rising oil droplet phases and a depiction of surface oil features.

SIMAP results were synthesized into daily TPAH50 concentration distributions that were used to evaluate toxicity. Specifically, the estimated TPAH50 concentrations were used with the dose-response curves developed for more and less sensitive fish and invertebrates in the absence of UV (see "Modeling of Toxicological Effect" section, above) to estimate the percent mortality in each concentration grid cell, assuming a daily exposure. The effect of UV on toxicity was not considered for the SIMAP-modeled TPAH50 concentrations, because UV does not appreciably penetrate to the depths considered in the SIMAP model (i.e., 20 to 1,400 meters beneath the surface). The estimates of percent mortality multiplied by volume affected were summed daily, and by depth layers at 20-meter intervals, to estimate volumes of water where plankton were killed. These numbers were multiplied by the numbers of organisms per volume (see "Empirical Biological Data" section, below) to calculate the numbers killed.

Empirical Chemistry Data

As oil continued to be released from the wellhead, scientists on both Response and NRDA cruises collected information regarding water and components of the water column to determine where the deep oil was going. However, it was difficult to sample the rising cone due to restrictions on vessels near the wellhead. Nonetheless, 47 water samples that were collected from May to August 2010, within

5 kilometers of the well and from a depth of 40 to 1,000 meters beneath the surface (below the upper mixed zone and to the top of the deep plume) were identified by forensic analysis as MC252 oil. The maximum TPAH50 concentration in these forensically matched samples was 19 µg/L (Payne & Driskell 2015a).

Later in the spill, concern over the deep oil plume grew and the deep plume was sampled more thoroughly. The concentrations of oil-derived chemicals were highest nearer the well and generally decreased with distance from the well. Particulate oil was present in the plume 155 kilometers from the well, and dissolved hydrocarbons from the oil could be detected up to 267 kilometers from the well. Forensic analysis indicated that more than 800 samples collected at depths of at least 1,000 meters contained MC252 oil. The highest TPAH50 concentration among these samples was 68 µg/L. Other indicators of the deep plume (e.g., presence of dispersant-derived chemicals, fluorescence, and decreased dissolved oxygen) were measured as far as 412 kilometers southwest of the well (Payne & Driskell 2015a).

Empirical Biological Data

Long-term biological data like those for surface waters (e.g., SEAMAP) do not exist for the deep water pelagic zone. Many deep water species occupy specific depth ranges in the mesopelagic and bathypelagic zones. The deep mid-water trawl nets and depth-stratified MOCNESS nets used during the NRDA were the most comprehensive sampling conducted to date for pelagic animals of the deep Gulf of Mexico waters. These data were used to describe the distribution and abundance of deep water fish and invertebrates exposed to oil (Sutton et al. 2015). Acoustic data collected for the NRDA were used to examine both the depths and locations of the deep layers of fish and invertebrates and their daily vertical migrations in the water column (Boswell et al. 2015).

Modeling of Toxicological Effect

The toxicological approach for deeper water is the same as previously described for surface waters (in Section 4.4.2.2.1), with two exceptions. First, UV effects are not considered for deeper water because UV does not penetrate to the depths considered (Lay et al. 2015a). Second, different species were selected (see Section 4.4.2.2.1) to bracket a range of sensitivity in the absence of UV.

4.4.2.3 *Sargassum* Assessment

The Trustees assessed exposure and injury to *Sargassum* and associated fauna. The Trustees documented direct oiling of *Sargassum* and then determined the following: the areal extent of surface oiling from the DWH spill, the density (i.e., percent cover) and area of *Sargassum* in the northern Gulf of Mexico, the area of *Sargassum* exposed to oil, and the amount of *Sargassum* area foregone due to lost growth caused by exposure to oil. The major inputs for the *Sargassum* assessment are described below and summarized in Table 4.4-3.

Table 4.4-3. Description of *Sargassum* assessment inputs.

Dataset or Model	Time Period Covered	Description of Dataset
Extent of Surface Oiling		
1. NOAA oil-on-water product	2010	Daily polygons of oiling from April to July 2010
***Sargassum* density calculations**		
2. NSF aerial surveys	2010, 2011	Photographs of *Sargassum* from low altitude aerial surveys of the northern Gulf of Mexico
3. USGS/NASA Landsat data	2010, 2011	Landsat data from 2010 and 2011

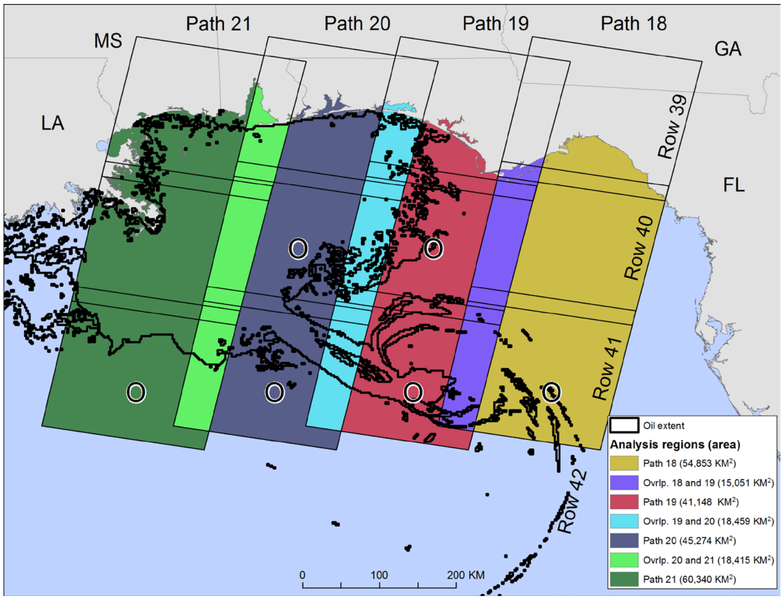
Extent of Surface Oiling

To estimate the area and extent of surface oil to which *Sargassum* was exposed, the Trustees relied on an intensive analysis of daily surface oil coverage based on multiple satellite sensors from April to August 2010 (Graettinger et al. 2015). To assess the upper and lower bounds of exposure, the Trustees developed two sets of cumulative oiling footprints based on two levels of percent cover of surface oil and two time periods. The surface oiling dataset provides information on the area of the ocean where there was oil with a given percent being covered by "thick oil." These polygons include ocean area that had surface oiling for at least 1 day. Based on expert opinion informed by field observation of both surface oil and *Sargassum*, the Trustees selected two cutoffs of the percent area covered by thick oil to bound the likely exposure of *Sargassum* to oil: areas with greater than 5 percent thick oil and areas with greater than 10 percent thick oil.

Sargassum moves across the northern Gulf of Mexico with currents and winds, taking approximately 6 weeks (S. Powers, personal communication, September 16, 2015) to move across the full range of the area affected by the spill. As a result, all the *Sargassum* in the area affected by oil is replaced after 6 weeks. Accordingly, exposure of *Sargassum* to surface oil was assessed in two separate 6-week timeframes: the early part of the spill (April 25 to June 5, 2010) and the latter part of the spill (June 6 to July 17, 2010). *Sargassum* present at the beginning of the spill was assumed to be oiled and injured as it moved through the spill area for 6 weeks and then replaced by additional *Sargassum* over the following 6 weeks. The total amount of *Sargassum* injured by the spill is the sum of quantities in areas of thick oil for these two time periods (Doiron et al. 2014).

Density of *Sargassum* and *Sargassum* Growth

The Trustees estimated *Sargassum* density (i.e., percent cover) by combining Landsat satellite images and low-altitude aerial photography. Landsat provides broad spatial coverage of the northern Gulf of Mexico (see Figure 4.4-15), but lacks sufficiently fine resolution to identify all *Sargassum* on the ocean surface. While low altitude aerial photographs have limited spatial coverage, they provide superior resolution in identifying *Sargassum*. Through a statistical analysis of satellite and low-altitude images matched by date and location, the Trustees developed a mathematical formula to estimate *Sargassum* percent cover (Hu 2015; McDonald 2015).



Source: *Hu (2015).*

Figure 4.4-15. Landsat paths used in the analysis of *Sargassum* percent cover estimates. "O" denotes areas that were only recorded in 2010. Landsat is a satellite run by USGS and NASA to collect land-surface data. In 2010, USGS increased Landsat coverage over the Gulf of Mexico to capture more images of the DWH oil spill.

An additional measure of *Sargassum* injury is the surface area foregone due to lost growth caused by oil exposure. As *Sargassum* moves across the northern Gulf of Mexico, it grows at a rate of 4 percent per day (LaPointe 1986; as cited in Powers 2012). The Trustees used this growth rate, combined with information on *Sargassum* in oil-contaminated surface waters, to calculate a range of surface area foregone.

4.4.2.4 Additional Biological Assessment

The Trustees conducted several additional studies to evaluate community/population level and physiological effects of the DWH oil spill on water column resources other than those described above. To investigate fish health, the Trustees worked with researchers to implement a field-based fisheries survey. The Trustees also used long-term fisheries-independent datasets to investigate population level impacts to commercially and recreationally important species. These investigations were used to evaluate injury to adult life stages. However, for reasons described below, these injuries were not quantified.

Fish Health Study

In the wake of the DWH oil spill, an increasing number of anecdotal reports were received of red snapper with skin lesions found in northern Gulf of Mexico waters. In response, the Trustees collaborated with academic researchers to quantify the prevalence and persistence of fish with lesions and to collect information about the health of fish beyond their external wounds. This research was conducted in a series of cruises along the continental shelf and within the Louisiana coastal estuary and marsh habitats. The scientists recorded observations of external abnormalities, measured and weighed whole fish and selected organs, and conducted necropsies. Additionally, fish and tissues samples were sent to analytical laboratories to determine many health endpoints, such as tissue damage (histopathology), age (otolith analysis), blood condition (blood chemistry and blood cell counts), and presence of pathogens.

Fisheries-Independent Data Analysis

To compare fish and invertebrate populations before and after the DWH oil spill, researchers at NOAA's Northwest Fisheries Science Center (NWFSC) analyzed fisheries-independent survey data (Ward et al. 2015). In this analysis, SEAMAP trawl survey data were used to assess population changes and detect shifts in catches across 51 species of fish and invertebrates. SEAMAP data series were expressed as standardized catch-per-unit effort (CPUE) and assessed using multivariate state-space models (Ward et al. 2015) and by intervention analysis (Scheuerell et al. 2015; Ward et al. 2015). The state-space models focused on detecting changes in standardized residuals of abundance, while the intervention analysis focused directly on CPUE. Both modeling approaches incorporated relevant environmental conditions (e.g., temperature, salinity, and dissolved oxygen) as covariates, as these factors may affect fisheries' abundance over time. In addition, data from trawl, seine, and gillnet surveys conducted by the Louisiana Department of Wildlife and Fisheries (LDWF) were used to assess changes in the population density of 12 fish and invertebrate species following the DWH oil spill. Catch records were expressed as standardized monthly CPUE and modeled with a delta-generalized linear modeling approach (Ward et al. 2015). The analyses incorporated environmental conditions (e.g., temperature, salinity, and turbidity) as independent variables in the models.

The Trustees used SEAMAP data and conducted population modeling to evaluate the potential effects of the oil spill on abundance and recruitment of red snapper (Tetzlaff & Gwinn 2015). The SEAMAP data used in these analyses included the early fall plankton survey, the fall groundfish survey (predominantly age-0 fish), and the summer groundfish survey (predominantly age-1 fish). Using these data, the catch of age-0 and age-1 red snapper were modeled for the eastern and western Gulf of Mexico through 2012 (Tetzlaff & Gwinn 2015). In addition, a before-after-control-impact analysis was performed to evaluate red snapper abundance before, during, and after the DWH oil spill in the impacted area (i.e., within the area of the Gulf of Mexico that was impacted by the DWH oil spill) and in a control area (i.e., outside the area of the Gulf of Mexico that was impacted by DWH oil). Depth, dissolved oxygen, and salinity were included as covariates in the model (Tetzlaff & Gwinn 2015). This analysis extends work described in the SEDAR *Gulf of Mexico Red Snapper Stock Assessment Report* (A.G Pollack et al. 2012; A.G. Pollack et al. 2012; SEDAR 2013).

Lastly, the Trustees investigated changes in recruitment and growth rates of red snapper on reefs in the northern Gulf of Mexico following the DWH oil spill (Patterson III 2015). Fisheries-independent data were collected using two methods: hook-and-line gear (i.e., bottom and vertical long lines) on or near artificial reefs in coastal Alabama and near petroleum platforms in coastal Texas, and remotely operated vehicles (ROVs) deployed on natural and artificial reefs sites in Alabama and northwest Florida. Age estimates in these datasets were derived from otolith analysis or age-length relationships. Various analyses and statistical tests were conducted using these fisheries-independent data to examine changes in red snapper recruitment and growth rates following the DWH oil spill.

4.4.3 Exposure

Key Points

- Following the blowout, DWH oil spread throughout the Gulf of Mexico water column, resulting in the deep-sea oil plume, rising oil plume, surface slick, and subsurface oil entrainment.
- Water column resources were exposed to DWH oil, and PAH accumulation was documented in the marine food web.
- *Sargassum* and surface oil are both transported by the same physical processes. Thus, as one would predict, *Sargassum* and oil were observed accumulating together in convergence zones (i.e., where surface waters come together).
- Understanding species distributions and life history patterns over time and space allowed the Trustees to determine that exposure occurred to different species and life stages both spatially and temporally.

The DWH blowout resulted in an unprecedented volume of oil transported throughout the water column, exposing an array of diverse and productive habitats and species. The Trustees used NRDA empirical data, modeling, and studies from the scientific community to document the spread of DWH oil through the water column and exposure to water column resources. The pathway of oil and exposure to water column resources is described for the surface slick and subsurface mixed zone (Section 4.4.3.1) and the rising oil and deep plume (Section 4.4.3.2). Evidence of oil exposure to *Sargassum* and the associated community is also described (Section 4.4.3.3).

4.4.3.1 Surface Slick and Subsurface Mixed Zone

Surface waters are the most biologically productive areas of the ocean as plankton (including the larvae of many economically important marine species) growth is enhanced due to the presence of nutrients in sunlight. The result is a congregation of life that forms the foundation for the marine food web.

Following the DWH blowout, a portion of the oil rose through the water column to the surface. Once at the surface, the oil slick spread across thousands of square kilometers and was transported by winds and currents over great distances, eventually reaching the northern Gulf of Mexico shorelines. Various physical factors influenced the persistence of oil at the surface (see Section 4.2). Some oil volatilized into the air and some was removed mechanically or by in situ burning. Wind and wave action and application of dispersants resulted in some oil within the surface slick becoming mixed in the upper water column. Entrainment of smaller oil droplets can result in dissolved hydrocarbon compounds in the water column.

Oil in the Subsurface Mixed Zone of Offshore and Shelf Waters

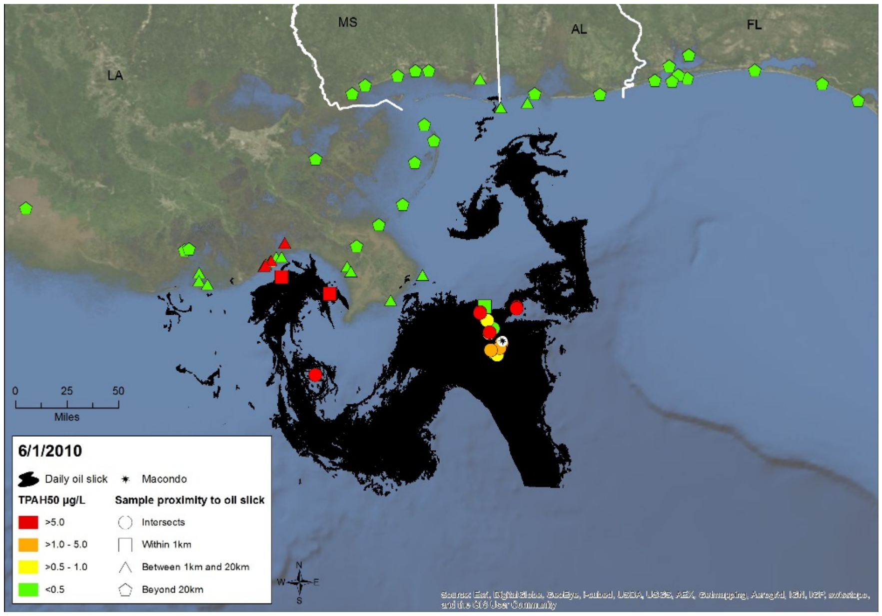
Floating oil was observed on the surface throughout the 87 days that oil flowed from the wellhead, and persisted for more than 3 weeks after the wellhead was capped (Section 4.2, Natural Resources

Exposure). The Trustees used satellite images to determine the areal extent and cumulative number of days where oil was observed on the sea surface (Section 4.4.2.2.1; Section 4.2, Natural Resources Exposure). This analysis has shown that 112,100 square kilometers (43,300 square miles) of open ocean surface water were exposed to oil—an area roughly the size of the state of Virginia (Graettinger et al. 2015). At its peak, oil covered more than 39,600 square kilometers (15,300 square miles) of the sea surface on a single day—an area about 10 times the size of Rhode Island (Graettinger et al. 2015).

Field-collected water samples were used to define the concentrations of oil resulting from entrainment. Oil concentrations were described as the sum of 50 PAHs (TPAH50) for comparison to toxicity test results (Section 4.2, Natural Resource Exposure; Section 4.3, Toxicity). Since most oil in the upper mixed zone was from surface slicks, the Trustees used the surface oil footprint to focus their analysis of subsurface waters. Samples used to evaluate subsurface mixed zone contamination were those collected both in the top 50 meters of the water column and at locations under or proximate to surface oil (defined by the daily SAR imagery) on the day collected. Samples were designated as "intersects SAR" if they were collected under the footprint of floating oil.

Samples located outside the floating oil footprint were grouped into three additional proximity groups: less than 1 kilometer from the floating oil, 1–20 kilometers from the floating oil, and more than 20 kilometers from the floating oil. Samples that were collected on dates that did not have a SAR image available were not assigned to an oil slick proximity group. Water samples collected within surface oil slicks were most likely to contain oil, and these samples generally had higher TPAH50 concentrations than samples collected at some distance from floating oil. The frequency of measured TPAH50 concentrations greater than 0.5 µg/L illustrates this point. Although we used dose-response curves rather than a single threshold to estimate mortality of biota, this TPAH50 concentration (0.5 µg/L) was selected because it is sufficiently high to harm sensitive life stages of biota in the presence of UV light (Section 4.3, Toxicity) (Morris et al. 2015b). Among samples in the uppermost 2 meters of the water column, 54 percent of samples that intersected the oil slick footprint had TPAH50 concentrations above this concentration; however, only 15 percent of samples within 1 kilometer of floating oil, 5 percent of samples within 20 kilometers of floating oil, and 2 percent of samples beyond 20 kilometers of floating oil exceeded a concentration of 0.5 µg/L. Ultimately, given the strong relationship between proximity to floating oil and elevated PAHs in the underlying water column, the Trustees focused subsequent evaluations on only those samples that were collected within the footprint of the floating oil.

Figure 4.4-16 further illustrates how TPAH50 concentrations in surface waters varied with distance from floating oil on 1 day of the spill: June 1, 2010.



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

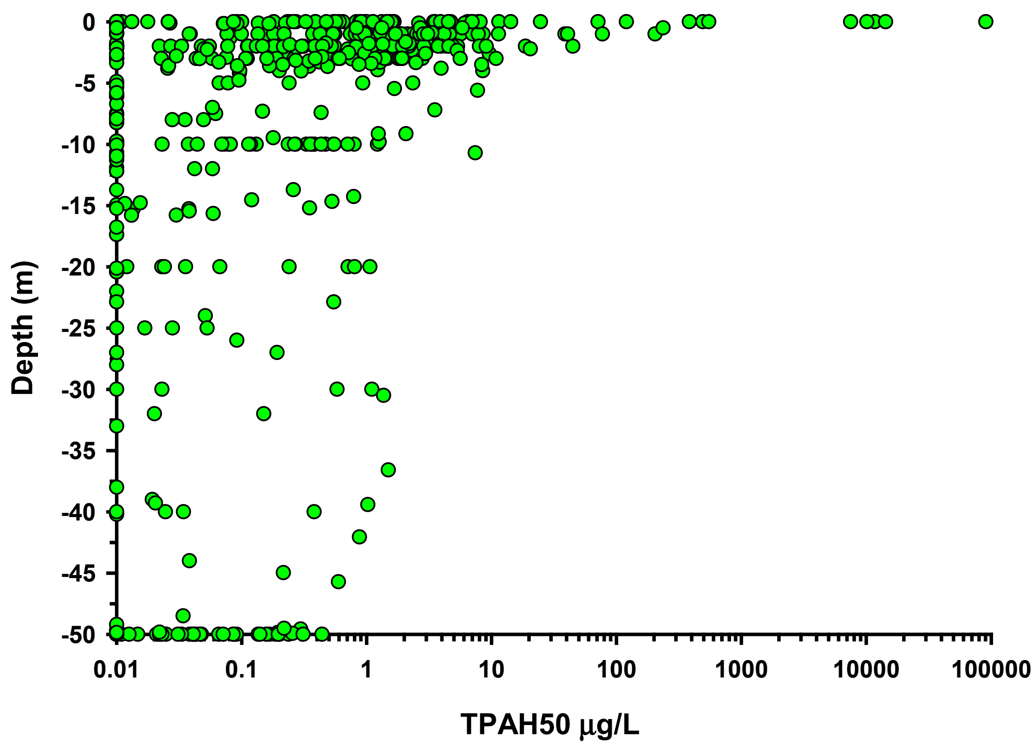
Figure 4.4-16. Example comparing the extent of surface oil (shaded black area) as determined from SAR imagery and water samples collected in the upper mixed zone on 1 day of the spill: June 1, 2010. The various colors indicate ranges of TPAH50 concentrations in water column samples collected from 0–20 meters depth, with red indicating the highest concentrations and green indicating the lowest concentrations. The symbol shape indicates the distance between the sampling location and floating oil on the sampling date.

The Trustees examined TPAH50 concentration as a function of depth beneath the surface using samples collected within or beneath the oil slick footprint, as determined from SAR analyses. These data were used to derive distributions of TPAH50 concentrations for different depth intervals (Figure 4.4-17). Those distributions, in turn, allowed for determining the range of PAHs to which an organism at that depth would have been exposed.

Below depths of 20 meters, PAHs were not detectable or were only detectable at low concentrations; the exception was for samples near BP's Macondo well, which may have been collected in the rapidly surfacing oil. An upper mixed layer depth of 20 meters is also consistent with conductivity, temperature, and depth (CTD) data collected during many of the cruises over the course of the DWH oil spill. Grennan et al. (2015) found that the average depth of the upper, mixed layer in the vicinity of the DWH spill site was approximately 16 meters, but extended to depths of 29 meters at some times. Based on the available data, we focused our analysis of surface oil effects on the upper 20 meters of the water column.

Some samples were collected at the surface, or a depth of 0 meters. Although oil concentrations in these samples would represent what an organism at the surface of the ocean might encounter, they would not necessarily represent concentrations in water beneath the surface slick. To evaluate water beneath the slick, these surface samples were excluded.

Approximately 380 water samples were collected at depths of 0.1 to 20 meters beneath the surface and under oil slicks. Most of these samples contained detectable TPAH50 concentrations, but 19 percent contained no detectable PAHs. The Trustees used the measured water concentrations to develop a statistical distribution describing the range and vertical distribution of TPAH50 concentrations under floating oil (Travers et al. 2015a).This is a limited dataset, particularly given the great spatial extent of the oil covering the Gulf of Mexico. Therefore, estimates of TPAH50 concentrations used in the analyses have inherent uncertainties.



Source: *Data obtained from DIVER; figure from Travers et al. (2015a).*

Figure 4.4-17. The figure shows TPAH50 levels in water column samples collected at the surface or beneath surface slicks over the course of the spill (green dots). Depth refers to the depth below the water surface at which the sample was collected from 0 to 50 meters. The 19 percent of samples with TPAH50 concentrations less than 0.01 μg/L are plotted at 0.01 μg/L. The Trustees used data to assess injuries in the upper water column from 0 to 20 meters below the surface.

The Trustees also forensically evaluated selected water samples collected in the upper 20 meters (65 feet) of the water column. For this analysis, care was taken to exclude samples that may have included the surface slick oil as a part of the sample. The Trustees confirmed that DWH oil was present in near-surface mixed zone water samples as far as 96 kilometers in most directions from the wellhead (Payne & Driskell 2015a). Through extensive sampling, the Trustees verified that samples of ambient water in the Gulf of Mexico (i.e., waters not affected by the DWH spill) have almost undetectable concentrations of PAHs. Specifically, NRDA water samples collected in the upper 20 meters (65 feet) of the water column in areas unaffected by the DWH spill had an average concentration of less than 0.06 μg/L (Payne & Driskell 2015a).

Oil in the Estuarine Waters

In estuarine waters, the Trustees evaluated water chemistry data in areas where oil was floating. These areas included Terrebonne, Barataria, and Mobile Bays, and Chandeleur and Mississippi Sounds. Consistent with methods used for the offshore areas, the Trustees considered the sample locations relative to oil slicks detectable in SAR imagery collected on the same day. Of the more than 3,700 nearshore/estuarine water samples collected between April and August, 2010, most were collected prior to the arrival of floating oil or in places away from floating oil. Only 120 of these samples were collected within 1 kilometer of an oil slick detectable in a SAR image on the same day. A 1-kilometer buffer was used in the estuarine waters because of the highly patchy nature of floating oil in these areas.

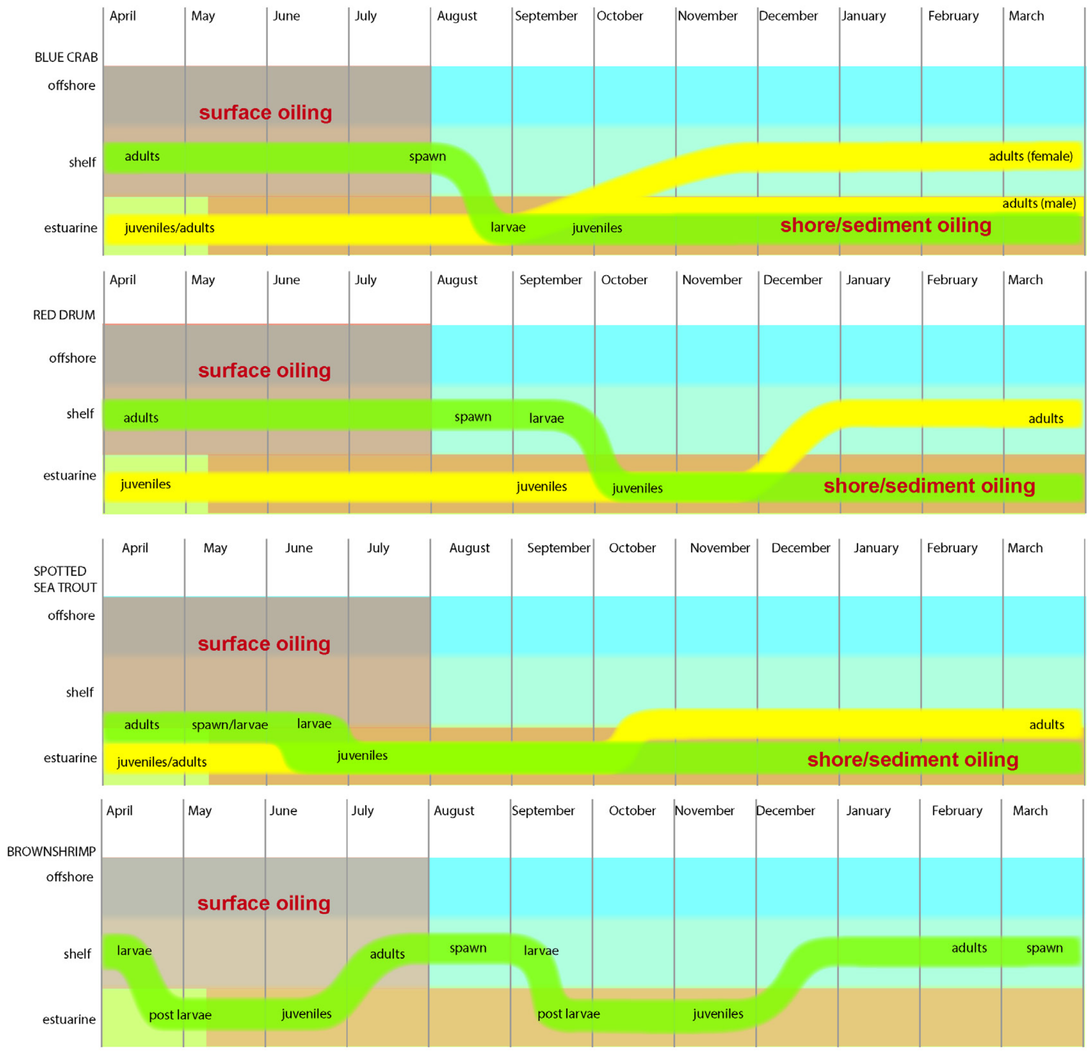
Oil concentrations in the estuarine or nearshore water samples collected near SAR-detected oil slicks were further evaluated based on whether they were collected at the water surface or at depth. Most of the samples were either collected at the water surface or the sample depth was not reported. Within this group of surface samples, TPAH50 concentrations varied from trace levels to 29 µg/L. Of the 24 samples that researchers collected below the water surface and that were associated with surface slicks, 10 had non-detectable PAHs; the TPAH50 concentrations found among the remaining 14 subsurface samples were 0.05 to 0.7 µg/L (Travers et al. 2015a).

Ultimately, the number of water analyses that researchers collected in estuarine waters near surface oil slicks during the spill was quite limited, resulting in considerable uncertainty about estuarine water concentrations associated with floating oil. However, the available data suggest that concentrations of TPAH50 were relatively low in estuarine waters below surface oil slicks. Therefore, estimates of natural resource injuries in estuarine waters relied on other lines of evidence such as the adverse effects of oil slicks (rather than PAHs) on biota (Morris et al. 2015a). The Trustees also conducted a forensic assessment of nearshore water samples collected during the year subsequent to the spill. DWH oil was forensically identified in 361 samples demonstrating that DWH oil was present in the nearshore or estuarine water column during the DWH spill or in the months that followed. Furthermore, DWH oil persisted in subsurface waters in some areas into 2011 (Payne & Driskell 2015b), long after floating oil was visible on the waters. The Trustees' analysis indicated that these samples contain dissolved components possibly leaching from previously deposited oil sources in the nearshore environment (Payne & Driskell 2015b).

Biological Distributions in Relation to Oil in the Upper Water Column

Many organisms are found in the subsurface mixed zone and regularly come in contact with the surface of the ocean. The persistence of floating oil on the surface of the Gulf of Mexico for nearly 4 months was a route of exposure for water column organisms, either through direct exposure to the oil sheen or to water contaminated by oil, dissolved hydrocarbons, and dispersants under the surface slick (Section 4.2, Natural Resources Exposure).

As Figure 4.4-18 illustrates, many fish and invertebrates species spawn in the estuarine, shelf, and offshore waters of the Gulf of Mexico during the spring and summer, releasing eggs and larvae into the upper water column. Thus, the potential for impact to early life stages is great.



Source: *Kate Sweeney for NOAA.*

Figure 4.4-18. Timing and location of fish and invertebrate life stages in relation to surface and shoreline oiling. Many estuarine-dependent species at different life stages are present in estuarine and shelf waters of the Gulf of Mexico during the spring and summer, overlapping with the timing of the DWH oil spill. Adults typically move to shelf waters or tidal passes to spawn. Following spawning, eggs and larvae are released into the water column, resulting in the potential exposure of these early life stages to the surface slick and subsurface oil entrainment. The larvae are eventually transported back into estuaries via winds and currents, exposing these same organisms to shoreline and sediment oiling as post-larvae, juveniles, or adults. The green and yellow bars in the top three figures represent different cohorts.

For example, using fisheries datasets and modeling, researchers estimated that the DWH oil spill overlapped 15 to 19 percent of high quality early life stage habitat for blackfin tuna during June and July 2010, 11 to 14 percent for mahi-mahi (dolphinfish), and 5 to 7 percent for sailfish (Rooker et al. 2013). Similarly, Muhling et al. (2012) reported that, on a weekly basis, up to 5 percent of bluefin tuna spawning habitat was likely impacted by the surface oil. Two tagged adult bluefin tuna were in close proximity to BP's Macondo well on the day of the accident and remained nearby for several weeks. Both fish putatively spawned in late April or early May in waters affected by the incident (Wilson et al. [In Press]).

Eggs of many pelagic fish species, such as mahi-mahi, are positively buoyant and are suspended in the water near the surface of the ocean until they hatch (Gearon et al. 2015), which could lead to high exposure to oil in the presence of the surface slick. Larval life stages of fish and crustaceans are transported by tides and currents, which could result in exposure to contaminated water. The SEAMAP ichthyoplankton dataset indicates that fish eggs were the taxonomic group found in the greatest density both on and off the shelf in both the spring and summer (French McCay et al. 2015c).

Several field studies reported by the scientific community suggest DWH oil exposure and PAH accumulation occurred via the marine food web:

- Researchers reported that mesozooplankton collected from the northern Gulf of Mexico showed evidence of exposure to PAHs and a PAH distribution similar to DWH oil (Mitra et al. 2012). The authors concluded that the DWH oil spill may have contributed to contamination in the northern Gulf ecosystem (Mitra et al. 2012).
- In a study conducted along the Mississippi Gulf Coast, Xia et al. (2012) measured significantly higher PAH concentrations in Mississippi seafood (i.e., fishes, shrimps, crabs, and oysters) in the months directly following the DWH oil spill, compared to samples measured at the later part of the study. Although PAHs were detected, all tested samples were below public health levels of concern (Xia et al. 2012).
- In an offshore fish survey, researchers documented relatively high concentrations of PAH metabolites in the bile of red snapper collected in 2011 in the northern Gulf of Mexico (Murawski et al. 2014). Consistent with the decreasing DWH oil exposure footprint over time, significant declines in PAH metabolites in red snapper bile were observed from 2011 to 2012 (Murawski et al. 2014; Snyder et al. 2015).
- Researchers also reported that PAH concentrations in the liver of reef fishes increased up to 20-fold between summer 2010 and fall 2010 and 2011 (Romero et al. 2014).

In response to human health concerns, a large effort was also conducted to test Gulf seafood for contaminants (Fitzgerald & Gohlke 2014; Ylitalo et al. 2012). For example, the federal and Gulf state agencies analyzed more than 8,000 seafood samples, including fish, shrimp, crabs, and oysters, collected in federal waters of the Gulf of Mexico (Ylitalo et al. 2012). Seafood samples consisted of edible muscle tissue (fillets) of whole fish or groups of small shellfish. Overall, PAHs and dispersants were found in low concentrations or below the limits of detection (Ylitalo et al. 2012). When detected, the concentrations were at least two orders of magnitude lower than the U.S. Food and Drug Administration (FDA) level of concern for human health risk (Ylitalo et al. 2012). Similar results were found from a study by Fitzgerald and Gohlke (2014), which tested the edible muscle tissue (fillets) of seven species of reef fish, including red snapper, red grouper, and tilefish, for PAHs, metals, and dispersants. Of the 92 samples analyzed, dispersants were not detected, and only two had detectable levels of PAHs and all were below the FDA level of concern (Fitzgerald & Gohlke 2014). Although these results may appear contrary to the ones discussed above, differences in the types of tissue sampled likely explain the discrepancies between studies. Fish are known to efficiently metabolize and eliminate PAHs (Stein 2010; Varanasi et al. 1989), typically resulting in relatively low or undetectable concentrations of PAHs in their muscle tissue just days after exposure (Stein 2010; Varanasi et al. 1989); conversely, PAH concentrations are typically highest in the bile and liver of fish after exposure (Varanasi et al. 1989).

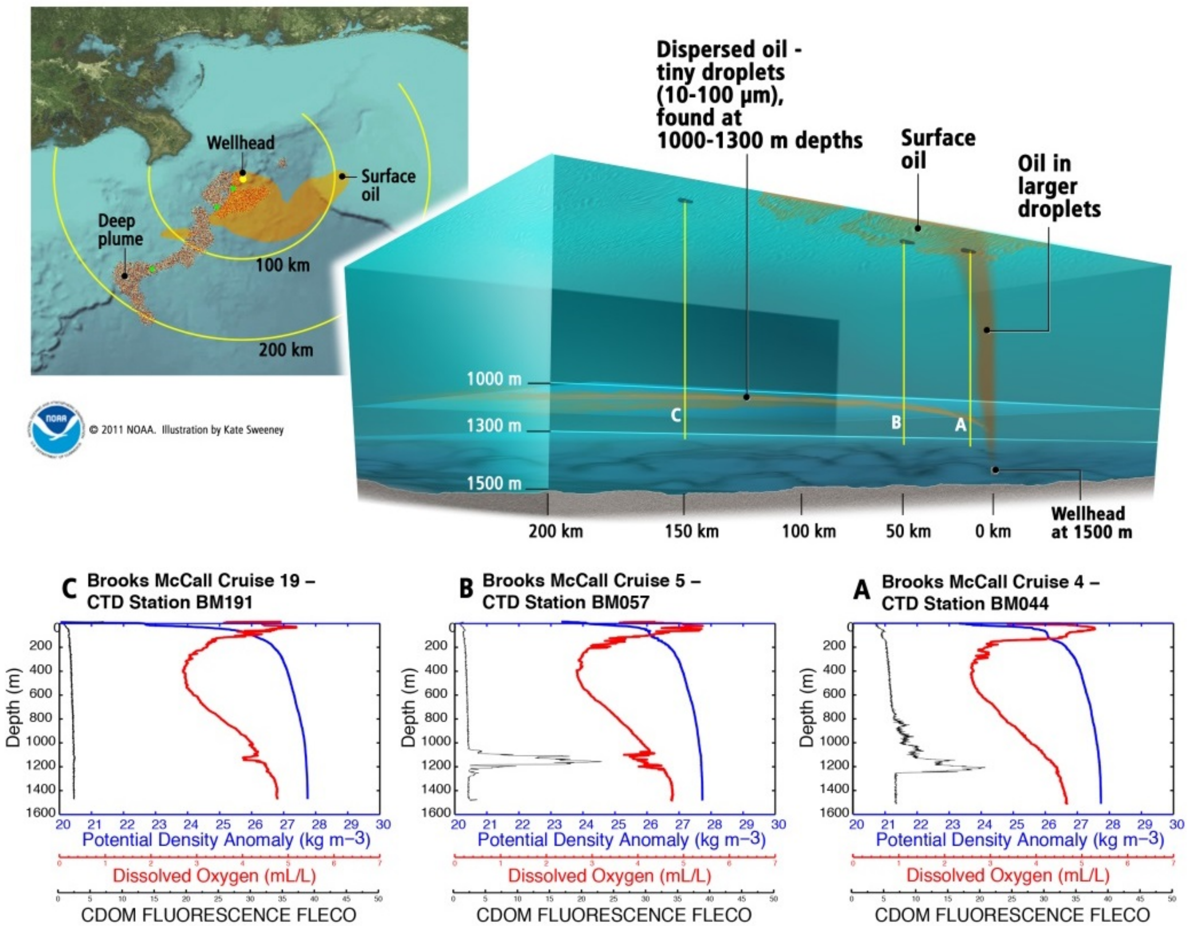
In addition to evaluating PAH concentrations in organisms, other analytical approaches were used to evaluate exposure. For example, based on stable isotope ($\delta^{13}C$) and radioisotope (^{14}C) analysis of plankton and fish, researchers concluded that petroleum-based carbon may have entered the planktonic (Chanton et al. 2012; Cherrier et al. 2013; Graham et al. 2010) and shelf reef fish food webs (Tarnecki & Patterson 2015). Additionally, researchers at the University of South Florida analyzed trace elements in otoliths of offshore fish species, including red snapper, red grouper, and southern hake, all collected following the DWH oil spill (Granneman et al. 2014a, 2014b). The researchers observed trace element anomalies in otolith profiles that occurred during the timeframe of the DWH oil spill event (Granneman et al. 2014a, 2014b). However, the authors note that some of the elements that changed during this time period were closely associated with salinity (Granneman et al. 2014a).

4.4.3.2 Rising Oil and Deep Plume

Oil released from the broken wellhead both dispersed at depth and rose through nearly a mile of water (Section 4.2, Natural Resources Exposure). Large oil droplets (greater than 1 millimeter in diameter) rose quickly—within a few hours to a day—to the ocean surface. Medium-sized oil droplets (between 100 microns and 1 millimeter in diameter) rose to the surface over the course of several days, during which time they were transported by currents away from the wellhead. High turbulence and the injection of dispersants at the source caused oil to be dispersed at the wellhead, which created small oil droplets and particles that remained near the release depth. These smaller oil droplets and dissolved hydrocarbons moved with the deep-sea currents, resulting in a deep-sea oil plume. The composition of the released gas-liquid mixture changed over time and space as the result of dilution, changes in pressure, dissolution, and addition of other constituents such as dispersants, methanol, and anti-foaming additives. Microbial consumption of gas and lighter fractions of oil were also documented (Valentine et al. 2010).

Research conducted for the NRDA during the response effort and research conducted by the academic community successfully located, tracked, and measured hydrocarbon concentrations in the rising oil and deep plume (see Figure 4.4-19). Although sampling was often excluded in the area nearest the wellhead due to safety concerns, water samples collected from the rising cone of oil were forensically matched to Macondo oil; these samples had a maximum TPAH50 concentration of 19 µg/L (Payne & Driskell 2015a). Sampling results indicated the presence of a plume approximately 1,000 meters beneath the surface and extending 10–35 kilometers from the wellhead (Camilli et al. 2010; Diercks et al. 2010; Hazen et al. 2010; Reddy et al. 2012), and the plume contained elevated petroleum hydrocarbon concentrations (Camilli et al. 2010; Diercks et al. 2010; Reddy et al. 2012) with TPAH concentrations reaching 189 µg/L (Diercks et al. 2010). Researchers also observed reduced dissolved oxygen concentrations in the deep plume (Du & Kessler 2012; Hazen et al. 2010; Kessler et al. 2011).

As described in Section 4.4.2.2.2 above, the Trustees conducted simulations of the rising cone of oil and deep plume (French McCay et al. 2015b). Over the course of the release, the model-estimated maximum TPAH50 concentrations in the cone and deep plume just above the source were 218 µg/L and 872 µg/L, respectively. The Trustees also estimated the volume of water with TPAH50 concentrations greater than 0.5 µg/L in these two areas. The maximum volume in the cone was estimated at 33 billion cubic meters (8.7 trillion gallons) and within the deep plume was 3.5 billion cubic meters (930 billion gallons). The modeling approach and results are described in more detail in (French McCay et al. 2015b).



Source: *Kate Sweeney for NOAA.*

Figure 4.4-19. Oil transport through the Gulf of Mexico water column, illustrating the deep-sea plume and rising oil. The lower figures A–C show CTD vertical water profiles measured on the Brooks McCall cruises. The depth below the ocean surface is shown on the left side of the profiles; the profiles extend from the bottom of the ocean (at 1,500 meters depth) to the surface. The CTD profiles provide information about the density of the water (blue), the dissolved oxygen content (red), and fluorescence that can indicate the presence of oil (gray). A decrease in dissolved oxygen and an increase in fluorescence at depth indicates the presence of the deep plume.

The oil released at the wellhead yielded a complex pathway of oil contamination that affected the Gulf of Mexico's bathypelagic and mesopelagic waters. NRDA surveys at these depths documented an enormous diversity of mesopelagic and bathypelagic fish, with more than 450 species identified. Some of these fish, including lanternfish, bristlemouths, and hatchetfish, were in high abundance (Sutton et al. 2015). These species could have been exposed to spill-related chemicals in various ways: directly to oil droplets or dissolved hydrocarbon compounds; through ingestion of oil droplets, contaminated water (e.g., oil droplets, dissolved hydrocarbons), or contaminated particles (e.g., detritus, marine snow); or through feeding on contaminated food, such as smaller fish, smaller invertebrates, phytoplankton, and zooplankton.

Field studies reported by the scientific community suggest oil exposure and PAH accumulation in the mesopelagic food web. For example, researchers reported post-spill muscle PAH concentrations observed in 2010 and 2011 in mesopelagic fishes were up to 10-fold higher than pre-spill levels observed in 2007 (Romero et al. 2014). In addition, researchers measured depletions in ^{13}C stable isotopes in two mesopelagic fishes and one shrimp species collected following the spill; these researchers concluded their results suggest carbon from the DWH oil spill was incorporated into the mesopelagic food web (Quintana-Rizzo et al. 2015).

4.4.3.3 Exposure of *Sargassum*

Sargassum is typically present in offshore waters throughout the northern Gulf of Mexico, including the area from Louisiana to the Florida Panhandle. In 2010, it was present in the area of surface oiling resulting from the spill. Both surface oil and *Sargassum* are subject to the same physical processes, leading to their accumulation in convergence zones. Thus, *Sargassum* located within the surface oiling footprint was likely co-located with areas of surface oiling, especially areas where thicker amounts of oil accumulate. This *Sargassum* along with any associated fauna was subject to injury.

During the months following the spill, Trustees documented direct exposure of *Sargassum* to oil throughout the time surface oil was present (Figure 4.4-20). This evidence comes from observations of direct oiling during the spill response and observations of *Sargassum* within the oiling footprint (Powers 2011; Powers et al. 2013).

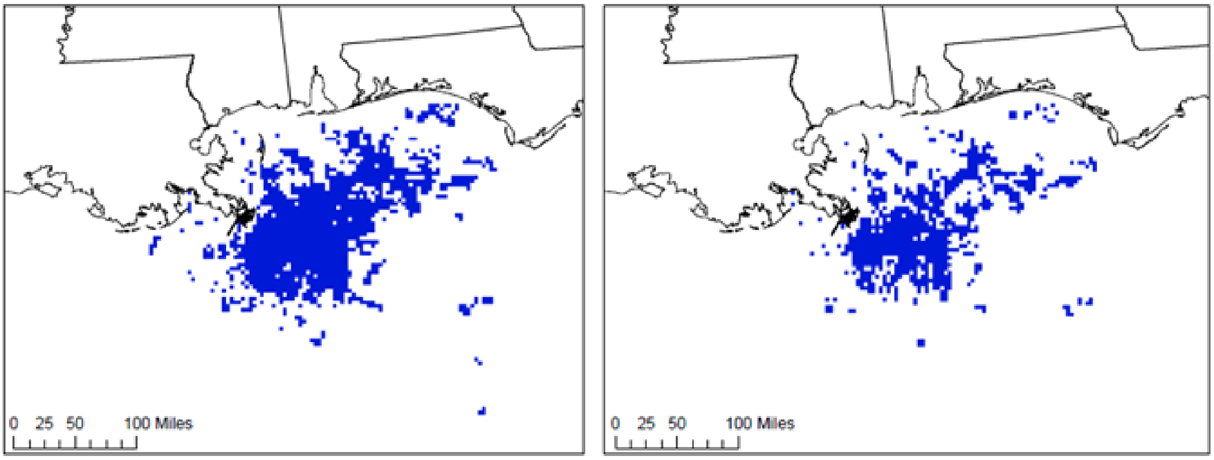
Source: *Powers and Hernandez, NSF supported project, 2010.*

Figure 4.4-20. Example of oiled *Sargassum*.

Using the approach described in Section 4.4.2.3, the Trustees estimated a range for the area over which *Sargassum* was exposed to surface oil from the DWH spill of 26,025 to 45,825 square kilometers (Table 4.4-4). The lower and upper ends of this range are based on the area of the cumulative oil footprint with greater than 10 percent thick oil and greater than 5 percent thick oil, respectively. Figure 4.4-21 shows the lower and upper ranges of the areal extent of the cumulative surface oil footprint.

Table 4.4-4. Cumulative surface oil footprint by percent of area with thick oil.

Percent of Area with Thick Oil	Total (Square Kilometers)
> 5% thick oil	45,825
> 10% thick oil	26,025



Source: *Doiron et al. (2014).*

Figure 4.4-21. Cumulative areas of surface oiling displaying the total area in the northern Gulf of Mexico with at least 1 day of >5% thick oil (45,825 km^2; left) and >10% thick oil (26,025 km^2; right).

4.4.4 Injury Determination

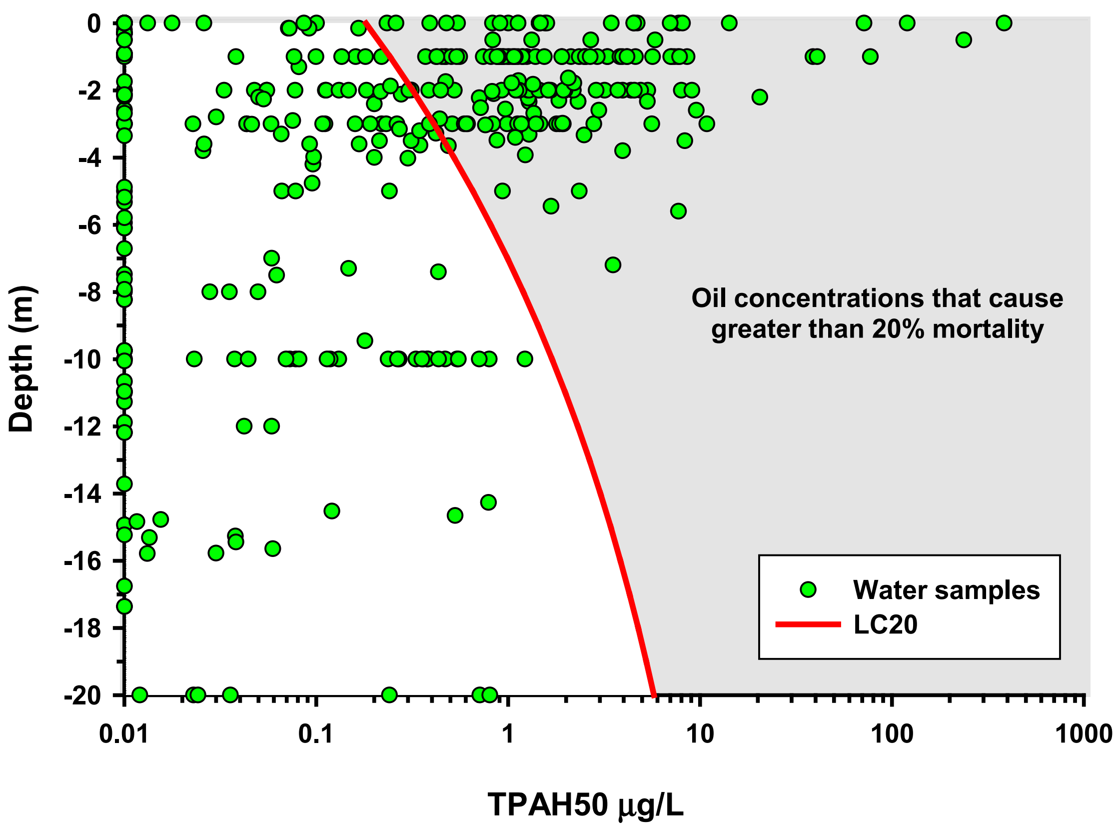
Key Points

- Water column concentrations measured in the Gulf of Mexico following the DWH oil spill exceeded levels known to cause lethal and sublethal effects to water column organisms.
- Early life stages of fish and invertebrates are particularly sensitive to oil exposure. Sunlight has been documented to magnify this effect.
- Field studies documented community-level and physiological injuries to water column resources, including trophic shifts, community structure shifts, reduced growth, impaired reproduction, and adverse health effects.

As presented earlier, the Trustees used NRDA empirical data, modeling, and additional studies from the scientific community to document DWH oil contamination throughout the Gulf water column and exposure of DWH oil to water column resources. As a next step, the Trustees used laboratory studies and field observations to understand potential effects of oil on water column organisms. The Trustees used representative species as key indicators of oil effects and applied an understanding of fundamental ecological relationships and processes to make reasonable scientific inferences about natural resources and services that were not explicitly studied. As discussed below, post-spill PAH concentrations measured in the northern Gulf water column exceeded levels known to cause lethal and sublethal effects among selected organisms. Field studies conducted in the Gulf of Mexico following the DWH oil spill also provide strong evidence of injuries to water column resources at both the species and community levels. The following sections describe the effects of oil on water column resources as documented in both laboratory studies (Section 4.4.4.1) and field observations (Section 4.4.4.2). These sections review studies funded through the NRDA and those reported by the scientific community.

4.4.4.1 Laboratory Studies

Toxicity studies conducted by the Trustees demonstrated that DWH oil mixed into the water and floating on the surface is toxic to early life stages of Gulf fish and invertebrates (Section 4.3, Toxicity). Additionally, the Trustees determined that exposure to UV light during or after exposure to DWH oil increases the toxicity by 10 to 100 times (Lay et al. 2015b). With exposure to an average amount of UV light present in the Gulf of Mexico during the spill, these toxic effects manifest over a relatively short timeframe (i.e., on the order of 24 hours). Concentrations of TPAH50 and exposure to UV light in the water column during the spill were sufficient to cause acute mortality to ichthyoplankton and zooplankton species. For example, based on the UV-adjusted estimate of toxicity for spotted seatrout at varying UV-levels with depth in the water column, many samples collected below or near the floating oil exceeded LC_{20} concentrations—or levels estimated to kill at least 20 percent of the population (Figure 4.4-22).



__Source:__ Abt Associates; TPAH50 data obtained from DIVER.

Figure 4.4-22. This figure shows that many water samples collected during the spill were toxic to early life stage spotted seatrout. The figure compares TPAH50 concentrations in water samples collected at different depths during the spill (green dots) to the estimated LC_{20} values for spotted (speckled) seatrout with UV light attenuation values (red line). Non-detect samples were set to 0.01 µg/L TPAH50 in this illustration so that they would be visible on the log scale. All field samples in the gray shaded area represent acutely toxic concentrations of TPAH50 to ichthyoplankton. For more information, see Section 4.3 (Toxicity), Travers et al. (2015b), and Lay et al. (2015b).

In addition to acute mortality, laboratory studies have documented a range of adverse toxicological effects due to oil on fish and invertebrates across numerous biological endpoints, including reduced growth (Brewton et al. 2013; Brown-Peterson et al. 2011; Brown-Peterson et al. 2015b; Griffitt et al. 2012; Griffitt et al. 2013; Morris et al. 2015b), immune suppression (Ortell et al. 2015), reduced swim performance (Mager et al. 2014; Morris et al. 2015b), and impaired cardiovascular development (Incardona et al. 2014). Sublethal toxic effects can reduce an organism's health, fitness, and ability to reproduce and survive. See Section 4.3 (Toxicity) for a detailed discussion on the range of effects caused by exposure to DWH oil.

4.4.4.2 Field Observations

4.4.4.2.1 Fish and Crustaceans

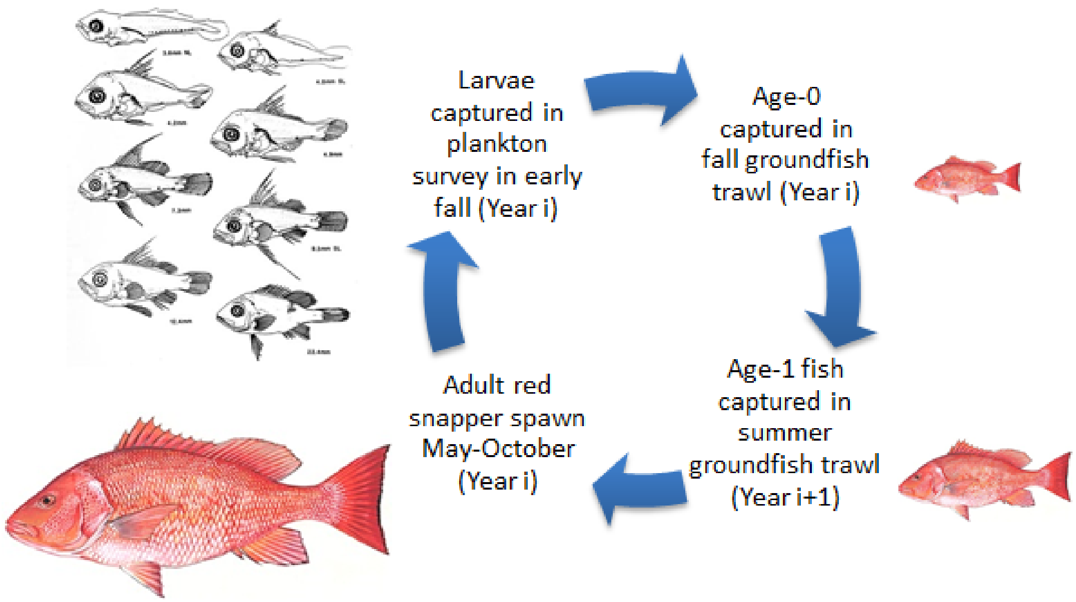
Reduced Recruitment

> **Red Snapper Life History**
>
> Adult red snapper spawn during the spring, summer, and fall months in the Gulf of Mexico (Moran 1988), releasing eggs into the water column. Larvae are then present in the water column for 15 to 30 days post-hatching. Age-0 red snapper typically are found in low-relief shell rubble and sand habitats (Patterson III et al. 2007; Patterson III et al. 2015). Some juvenile red snapper recruit to reefs with greater vertical relief at age 1, and the majority recruit to reefs by age 2 (Patterson III 2015; Patterson III et al. 2007).

Red snapper is a commercially and recreationally important species found in the Gulf of Mexico, with early life stages present in the water column and older individuals recruiting to natural and artificial reefs on the continental shelf (Patterson III et al. 2007). Due to its behavior to congregate around reef structures, red snapper are highly susceptible to fishing and are on the NMFS list of overfished stocks (NOAA 2015). The Gulf of Mexico red snapper stock assessment uses fisheries-independent and fisheries-dependent data to estimate the status of the stock relative to fishing and biomass benchmarks via stock assessment modeling; this stock assessment was updated through 2011 (SEDAR 2013). Although the stock assessment showed that the Gulf of Mexico red snapper population was rebuilding since the mid-2000s, the model predicted the lowest recruitment of age-0 red snapper in the eastern Gulf of Mexico in 2010 and 2011, compared to the past 20 years (A.G Pollack et al. 2012; SEDAR 2013; Tetzlaff & Gwinn 2015). To further investigate this finding and determine if the diminished recruitment was a result of the DWH oil spill, the Trustees analyzed fisheries-independent datasets across multiple gear types, habitat types, and size classes to examine red snapper abundance in the years following the DWH oil spill (Patterson III 2015; Tetzlaff & Gwinn 2015). Other field observations on the red snapper population, such as changes in trophic structure and reduced growth, are discussed in the sections below.

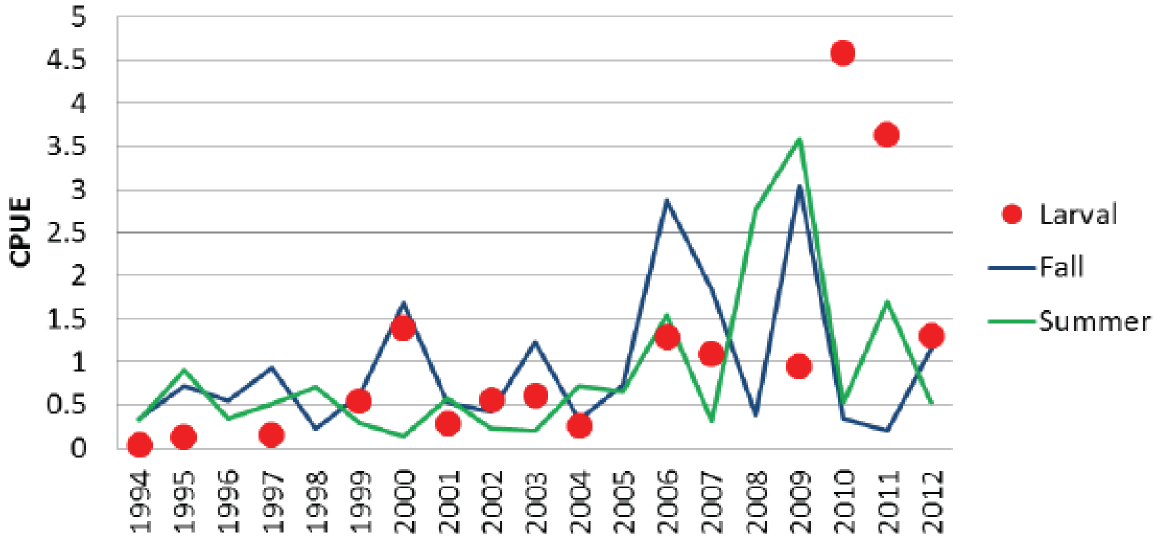
Building off the Gulf of Mexico red snapper stock assessment (A.G Pollack et al. 2012; A.G. Pollack et al. 2012; SEDAR 2013), the Trustees used SEAMAP fisheries-independent data and modeling to calculate abundance indexes of larval, age-0, and age-1 red snapper through 2012, with different surveys targeting specific size classes (Figure 4.4-23). Analyses of the SEAMAP early fall plankton surveys in 2010 and 2011 suggest high red snapper larval abundances in the eastern Gulf of Mexico water column during this time, as shown in Figure 4.4-24 (A.G Pollack et al. 2012; SEDAR 2013; Tetzlaff & Gwinn 2015). However, the SEAMAP fall groundfish surveys provide evidence of low abundances of red snapper in the eastern Gulf of Mexico in 2010 (2010 year class) and 2011 (2011 year class) (A.G. Pollack et al. 2012;

SEDAR 2013; Tetzlaff & Gwinn 2015). These results suggest that although high abundances of red snapper larvae were observed in the water column for the 2010 and 2011 year classes in early fall, these abundant larval populations did not translate into higher recruitment of age-0 fish in late fall for either the 2010 or 2011 year classes (Tetzlaff & Gwinn 2015). However, this trend of low abundance was not observed in the SEAMAP summer ground fish survey in 2011 and 2012 (predominately catching age-1 fish), indicating that the low age-0 abundances did not translate into low age-1 abundances for the same 2010 and 2011 year classes (Figure 4.4-24).



***Source**: From Tetzlaff and Gwinn (2015).*

Figure 4.4-23. Overview of SEAMAP fisheries surveys and red snapper life history. Red snapper typically spawn between May and October, releasing eggs into the water column. The SEAMAP early fall plankton survey samples the newly hatched larvae. The SEAMAP fall groundfish survey captures predominantly age-0 fish. The SEAMAP summer groundfish survey captures predominantly age-1 fish.



Source: *Tetzlaff and Gwinn (2015).*

Figure 4.4-24. Indices of red snapper recruitment in the Eastern Gulf of Mexico. Larval abundance indexes (red dots) are based on SEAMAP early fall plankton surveys. SEAMAP fall groundfish surveys (blue line) predominantly index age-0 fish abundance. SEAMAP summer groundfish surveys (green line) predominantly index age-1 fish abundance. High red snapper larvae abundances were observed in early fall 2010 and 2011 (red dots in grey highlight box); however, these abundant larval populations did not translate into higher recruitment of age-0 fish in late fall for either of these year classes (blue line in grey highlight box).

The observed decline in age-0 red snapper abundances could be explained by the DWH oil spill, impacting the survival of red snapper larvae, age-0 fish, or post-settlement red snapper on the shelf. However, alternative explanations could include poor larval settlement due to oceanographic environmental conditions or increased abundance of predators. Using a before-after-control-impact analysis, the Trustees investigated red snapper abundance before, during, and after the DWH oil spill in the impacted area (i.e., within the area of the Gulf of Mexico that was impacted by the DWH oil spill) and a control area (i.e., outside the area of the Gulf of Mexico that was impacted by DWH oil). A decline in age-0 fish was observed during the spill, relative to the time period before the spill (Tetzlaff & Gwinn 2015). However, this decline was not found to be significant and environmental variables, including water depth, salinity, and dissolved oxygen, explained significant amounts of variance in age-0 red snapper abundance (Tetzlaff & Gwinn 2015).

In addition to using SEAMAP survey data, the Trustees analyzed field data collected from natural and artificial reefs in the northern Gulf of Mexico to investigate whether red snapper recruitment to reefs was affected (Patterson III 2015). Since red snapper typically begin to recruit to reefs as age-1 fish, with the majority recruiting by age-2

Red Snapper Oil Exposure and Effects

Red snapper exposure to DWH oil is supported by carbon isotope ratios and relatively high concentrations of PAH metabolites in bile.

Field-collected data following the DWH oil spill observed changes in the red snapper community, including growth reductions, skin lesions, and shifts in diet.

(Patterson III 2015), a 1- to 2-year delay may occur before measurable oil spill impacts are seen in recruitment to reef populations. Analysis of ROV and vertical long line data from natural and artificial reefs in the northern Gulf of Mexico provided some observations of reduced abundances of the 2010-year class of red snapper in 2011 (age 1) and in 2012 (age 2) (Patterson III 2015). However, these findings were not consistent across age, gear type, and location, and not supported by other studies (Szedlmayer & Mudrak 2014).

In summary, these analyses provide limited evidence of diminished red snapper recruitment in the years following the oil spill (Patterson III 2015; Tetzlaff & Gwinn 2015). Reduced recruitment was observed in age-0 juvenile red snapper from the 2010 and 2011 year classes from the eastern Gulf of Mexico; however, environmental variables explained significant amounts of the variance. Some observations of reduced abundances of the 2010- and 2011-year classes were also detected as the cohort recruited to natural and artificial reefs as older fish; however, these signals are somewhat equivocal in the datasets examined. Although no strong trends were observed, it should be noted that statistical approaches for detecting trends in relative abundance are often limited by statistical power (Peterson et al. 2001) (Section 4.4.5.2). A very large suite of biotic and abiotic factors lead to a high degree of natural variation in estimates of annual abundance, both spatially and temporally, which poses challenges in identifying the effects from an event, such as an oil spill, from other environmental factors (Fodrie et al. 2014). Thus, although there is no strong support for DWH oil impacts on red snapper recruitment, the Trustees cannot rule out the possibility that recruitment was potentially affected by the spill.

Changes in Trophic Structure

Researchers have observed trophic level changes in the red snapper community following the DWH oil spill. Based on gut content and stable isotope analyses of red snapper collected on artificial and natural reef sites in the northern Gulf of Mexico from 2009 to 2011, researchers reported a significant shift in red snapper diet and trophic ecology after the DWH oil spill (Tarnecki 2014; Tarnecki & Patterson 2015). The researchers concluded that their results suggest both an increase in red snapper trophic position and a change from pelagic to benthic prey species (Tarnecki 2014; Tarnecki & Patterson 2015).

Changes in Community Structure

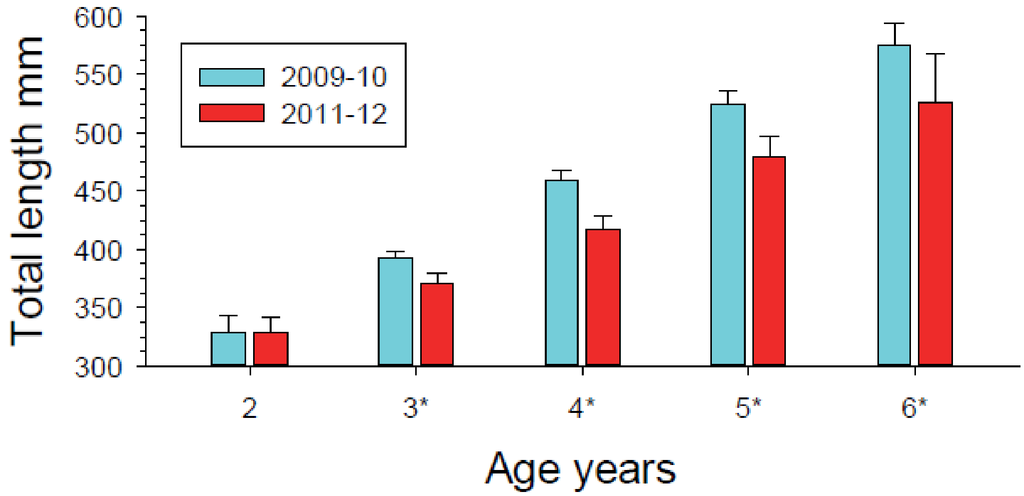
Researchers have reported effects of the DWH oil spill on fish community structure. For example, (Patterson III et al. 2015; Patterson III et al. 2014) observed shifts in reef fish communities beginning in summer 2010. The greatest changes were observed in small demersal fish, such as damselfish, cardinalfish, and wrasses, many of which declined in abundance by 100 percent following the DWH oil spill. Several species of large fishes, including snappers, jacks, and triggerfish, also declined in abundance up to 70 percent the year following the spill (Patterson III et al. 2014). By the fourth year post-spill, fish communities generally showed signs of resiliency, except for small demersal fish, which had persistently lower densities (Patterson III et al. 2015).

As a potentially confounding factor, increases in invasive lionfish have also been observed on reef sites in the northern Gulf of Mexico, with the highest densities of lionfish (*Pterois* spp.) reported on artificial reef sites in 2012 and 2013 (Dahl & Patterson III 2014). Notably, however, lionfish densities were not detected on natural reef sites in 2010, and greater than an order of magnitude lower densities of lionfish were observed on natural reef sites (<0.05 fish per 100 square meters) compared to artificial reef sites (approximately 2 fish per 100 square meters) in 2011 (Dahl & Patterson III 2014). Based on these observations, the Trustees conclude that the lionfish invasion cannot solely explain the observed changes to the reef fish community on natural reef sites in the northern Gulf of Mexico following the DWH oil spill.

Reduced Growth

Researchers observed reduced growth rates and size at age for red snapper collected on reef sites following the DWH oil spill (Herdter 2014; Neese 2014; Patterson III 2015). Through analyzing red snapper otoliths collected in the northern Gulf of Mexico and West Florida Shelf, Herdter (2014) found a significant decline in fish growth corresponding with the timeframe of the DWH oil spill. Similarly, researchers observed significant decreases in size at age of red snapper sampled on reefs off the coast of Alabama and Florida (Neese 2014; Patterson III 2015). As shown in Figure 4.4-25, the length of red snapper sampled after the DWH oil spill (2011–2012) were significantly smaller than those of the same age collected before the spill (2009–2010). As discussed above and below, red snapper diet shifts (Tarnecki 2014; Tarnecki & Patterson 2015) and increased prevalence of lesions (Murawski et al. 2014) were observed following the DWH oil spill. These observations suggest that red snapper were under greater stress that may have negatively affected their growth rates in the years following the spill (Patterson III 2015). Researchers have also reported reduced size at age for tomtate (*Haemulon aurolineatum*), another reef fish, following the DWH oil spill (Norberg & Patterson 2014).

These field results are consistent with observations from laboratory studies and field experiments that have reported decreased growth rates for fish and crustaceans exposed to oil. For example, researchers held brown and white shrimp in field mesocosms in Barataria Bay, Louisiana, along shorelines that were impacted by the DWH oil spill (Rozas et al. 2014). The researchers reported growth rates for brown shrimp to be significantly lower at heavily oiled shorelines compared to those observed at very lightly oiled shorelines and shorelines with no oil (Rozas et al. 2014). Laboratory studies have also documented decreased growth rates of fish and shrimp exposed to oil-contaminated water or sediment (Brewton et al. 2013; Brown-Peterson et al. 2011; Brown-Peterson et al. 2015b; Griffitt et al. 2012; Griffitt et al. 2013; Morris et al. 2015b) (see Section 4.3, Toxicity, for more information).



Source: *Patterson III (2015).*

Figure 4.4-25. Red snapper size at age on reef sites in the north central Gulf of Mexico. Mean (95 percent confidence interval) size at age of red snapper sampled in the north central Gulf of Mexico before (2009–2010) versus after (2010–2012) the DWH oil spill as reported by Neese (2014) and Patterson III (2015). As shown in the figure, red snapper (ages 3–6) sampled after the DWH oil spill (2011–2012) were significantly smaller than those collected before the spill (2009–2010).

Impaired Reproduction

Researchers observed impaired reproductive parameters in spotted (also known as speckled) seatrout collected from Barataria Bay, Louisiana, and the Mississippi Gulf Coast 1 year following the spill, compared to historical data from the same location (Brown-Peterson et al. 2015a). For example, researchers documented both significantly lower gonad weights (relative to body weight) in females and significantly reduced spawning frequency, compared to pre-spill data (Brown-Peterson et al. 2015a).

Adverse Health Effects

Fish health may have also been impacted as a result of the DWH oil spill. For example, researchers and fishermen have reported an increased prevalence of skin lesions in red snapper and other fish species following the DWH oil spill (Burdeau 2012; Murawski et al. 2014; Pittman 2011). Consistent with the decreasing DWH oil exposure and decreasing concentration of PAH metabolites in red snapper bile over time, the overall frequency of lesions declined from 2011 to 2012 (Murawski et al. 2014). These findings are also supported by laboratory experiments exposing fish to a combination of oil and pathogenic bacteria (Ortell et al. 2015). These studies found that oil exposure impaired immune function of fish, increasing their susceptibility to infection, which led to increased death (likely due to pathogenic bacteria) and caused skin lesions in some fish (Ortell et al. 2015).

In addition, researchers found that 19 percent of red drum caught in Barataria Bay and near the Mississippi River Delta suffered from anemia (i.e., low numbers of red blood cells), while none of the fish caught at reference sites in Terrebonne Bay displayed signs of anemia (Harr et al. 2015). Fish collection sites in Terrebonne Bay were selected because the closest shorelines were classified in the no-oil-observed Shoreline Cleanup and Assessment Team (SCAT) category, while Barataria Bay and Mississippi River Delta sites were selected based on their classification in the heavy oiling SCAT category (Harr et al. 2015). In laboratory toxicity tests, oil caused low red blood cell counts in fish and birds (Section 4.3, Toxicity) (Bursian et al. 2015a; Bursian et al. 2015b; Dorr et al. 2015; Fallon et al. 2014; Harr et al. 2015; Ortell et al. 2015). Animals with reduced red blood cell counts are less able to transport oxygen throughout their body and, therefore, have less energy. As a result, organisms with anemia are at a competitive disadvantage in terms of catching prey, escaping predators, and other activities important for their fitness.

Physiological and genomic impacts were also observed in resident marsh fish species, as discussed in more detail in Section 4.3. Gulf killifish, a low trophic level forage fish collected from oiled sites in coastal Mississippi and Alabama, had gill damage and changes in gene expression associated with ion regulation, stress response, immune response, developmental abnormalities, and decreased reproductive success (Dubansky et al. 2013; Whitehead et al. 2012). Fish with gill damage are unable to uptake normal levels of oxygen, while the abnormal gene expression patterns reduce the organism's general fitness, because they spend more energy toward addressing these adverse effects (e.g., fighting off infections) and thus have less energy to put toward catching food and escaping predators.

Effects of Oil on Fish

Laboratory studies using DWH oil documented:

- Mortality to early life stages.
- Reduced growth.
- Immune suppression.
- Reduced swimming performance.
- Impaired cardiovascular development.

Post-DWH oil spill field studies observed:

- Reduced growth.
- Impaired reproduction.
- Skin lesions.
- Anemia.
- Trophic shifts.
- Community structure shifts.

4.4.4.2.2 Phytoplankton, Zooplankton, and Bacteria

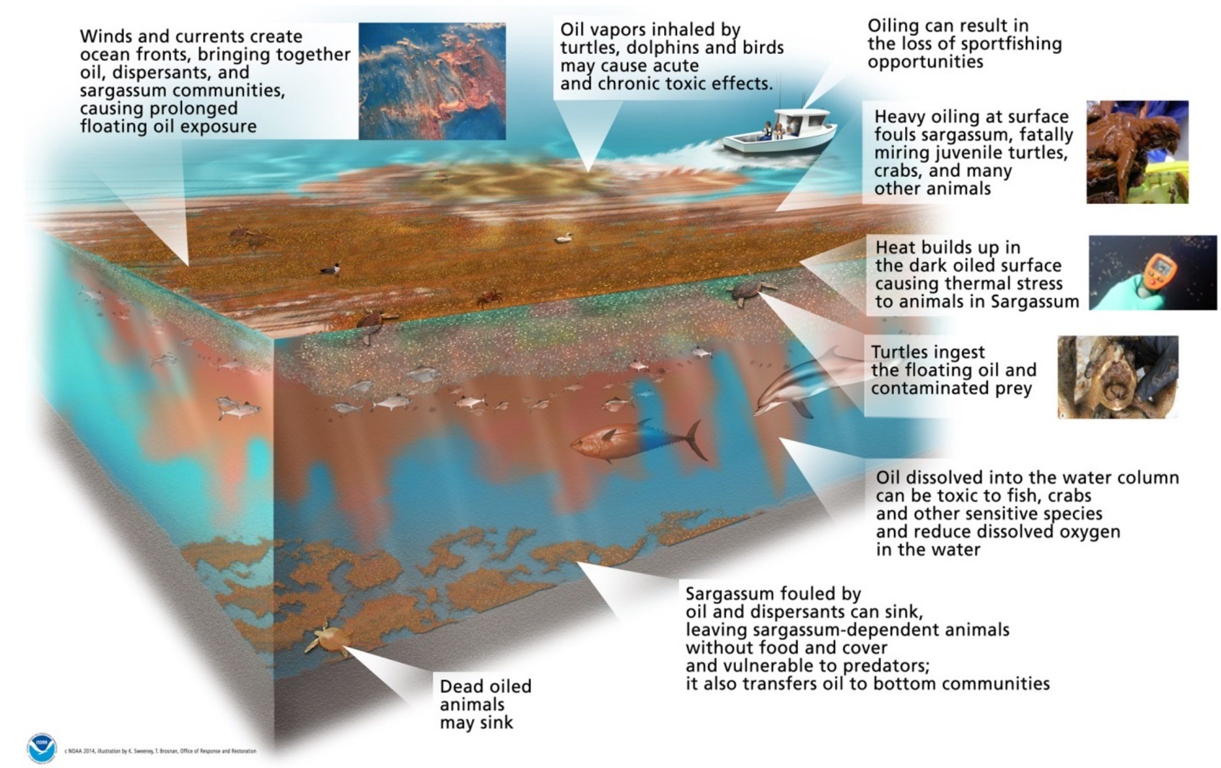
Field studies reported by the scientific community have documented changes in community composition for plankton and bacterial communities following the DWH oil spill. For example, researchers reported significant changes in the mesozooplankton community off the coast of Alabama during May and June 2010, compared to historical data during the same time period (Carassou et al. 2014). In addition, a shift in dominant bacteria was observed in the large subsurface plume (900–1,300 meters beneath the surface) following the DWH oil spill between March and August 2010 (Dubansky et al. 2013). Other studies reported altered community composition of bacteria in the plume compared to sites outside of the plume (Lu et al. 2012; Redmond & Valentine 2012; Yang et al. 2014) and stimulation of oil-degrading bacteria (Hazen et al. 2010).

Laboratory studies have also demonstrated oil directly affects phytoplankton and zooplankton. See Section 4.3 (Toxicity) for additional discussion.

4.4.4.2.3 Effects of Oil on *Sargassum*

Section 4.4.2.3 described the important role of *Sargassum* in the ecosystem and how it provides essential habitat to a wide array of fish, invertebrates, and other animals in the open ocean. When *Sargassum* becomes fouled by oil, it can no longer provide these ecosystem services. Scientific literature supports the conclusion that the physical coating of *Sargassum* with oil causes substantial, acute injury to *Sargassum* and that lower levels of oil likely inhibit or decrease growth (Powers 2012). Furthermore, oiled *Sargassum* harms the invertebrates, fish, and other animals (e.g., sea turtles and birds) found within it and nearby. This harm occurs through physical fouling and direct toxicity from exposure to oil, and potentially by reducing the availability of dissolved oxygen.

In addition, the exposure of *Sargassum* to both oil and dispersant can cause *Sargassum* to sink to the ocean floor, decreasing the area of this important habitat. The sinking of oiled *Sargassum* could serve as an additional pathway of oil exposure to benthic flora and fauna (Powers et al. 2013). Figure 4.4-26 shows the myriad effects of oil exposure on *Sargassum.*



Source: *Kate Sweeney for NOAA.*

Figure 4.4-26. Illustration of the potential impacts of oiled *Sargassum* and associated biota in the water column.

4.4.4.3 Summary

Post-spill water column concentrations measured in the northern Gulf of Mexico exceeded levels that are known to cause lethal and sublethal effects to aquatic organisms, providing evidence of injuries to water column resources. Community and physiological effects were also recorded during field observations following the DWH oil spill. Notably, growth reductions (Herdter 2014; Neese 2014;

Patterson III 2015), shifts in diet (Tarnecki 2014; Tarnecki & Patterson 2015), and increased prevalence of lesions (Murawski et al. 2014) were observed in red snapper collected from the northern Gulf of Mexico; reef fish populations displayed shifts in community structure (Patterson III et al. 2015; Patterson III et al. 2014); and impaired reproduction (Brown-Peterson et al. 2015a) and anemia (Harr et al. 2015) were observed in spotted seatrout and red drum, respectively. Many of these endpoints were also observed in laboratory experiments studying the effects of oil exposure. Additional biological endpoints observed in the laboratory include immune suppression, reduced swimming performance, and impaired cardiovascular development (Section 4.3, Toxicity). Sublethal toxic effects can reduce organisms' health, fitness, and ability to reproduce and survive. For example, oil exposure may interfere with organisms' ability to respond to suboptimal environmental conditions, and the combined effects of naturally encountered stressors (e.g., salinity, hypoxia, pathogens) may contribute to impacts on species fitness as well as to populations and communities (Whitehead 2013).

4.4.5 Injury Quantification

Key Points

- The Trustees used mortality to early life stages of fish and planktonic invertebrates exposed to oil in the surface slick, the subsurface mixed zone, the rising cone, and the deep plume of oil to quantify the number of organisms killed as a direct result of oil exposure.
- The Trustees estimated that the total number of larval fish and invertebrates killed was 2 to 5 trillion and 37 to 68 trillion, respectively. The range of survival for fish larvae to live past 1 year of age ranges from one in a thousand to 1 in 5 million (French McCay et al. 2015d). This translates into a loss of millions to billions of fish that would have reached age 1.
- The estimated total number of fish and planktonic invertebrates killed in estuarine waters was between 0.4 and 1 billion and between 2 and 6 trillion, respectively. The Trustees also estimated the lost growth that some of the killed organisms would have undergone if they had lived their normal lifespan.
- Analyses of long-term fisheries-independent datasets did not detect significant changes to fisheries populations. However, due to the inherent variability in fisheries datasets, the Trustees cannot rule out the possibility for population-level effects.
- Analysis of *Sargassum* found that exposure to oil may have caused the loss of up to 23 percent of this habitat. The total loss of *Sargassum*, including foregone area from lost growth, is 11,100 square kilometers.

As presented in Section 4.4.2, Approach to the Assessment, the Trustees used NRDA empirical data, modeling, and additional studies from the scientific community to document 1) DWH oil contamination throughout the Gulf water column, 2) exposure of DWH oil to water column resources, and 3) lethal and sublethal effects of DWH oil on aquatic organisms. As a next step, the Trustees quantified injuries to water column resources as a result of the DWH oil spill. The integrated water column resource analysis, including the injury quantification of fish and planktonic invertebrate species, is presented in Section

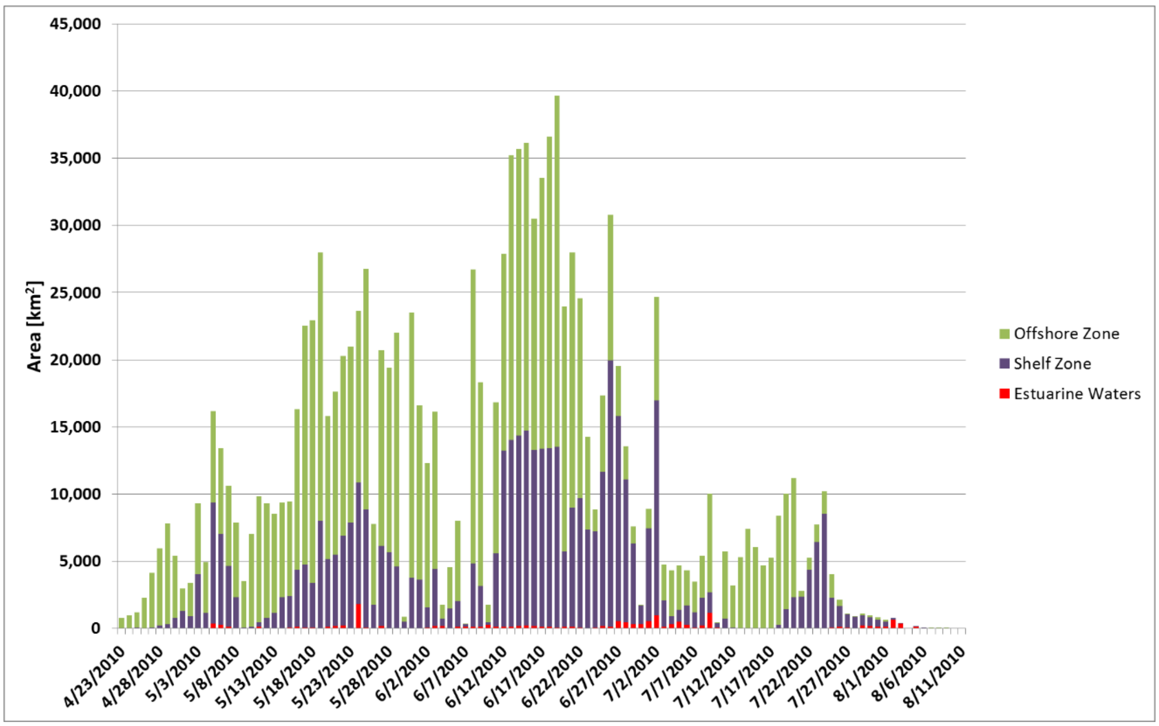
4.4.5.1. Analyses of long-term fisheries-independent datasets are presented in Section 4.4.5.2. The *Sargassum* injury quantification is presented in Section 4.4.5.3.

4.4.5.1 Integrated Water Column Resource Analysis

This section presents the Trustees' integrated water column resource analysis to quantify injuries to resources found within the estuarine, shelf, and offshore water column. The quantification of injuries is based on the mortality of water column resources as a result of the DWH oil spill. Separate analyses were performed for the surface slick and subsurface mixed zone for the offshore, shelf, and estuarine areas (Section 4.4.5.1.1) and for the rising oil and deep plume of oil at depth (Section a).

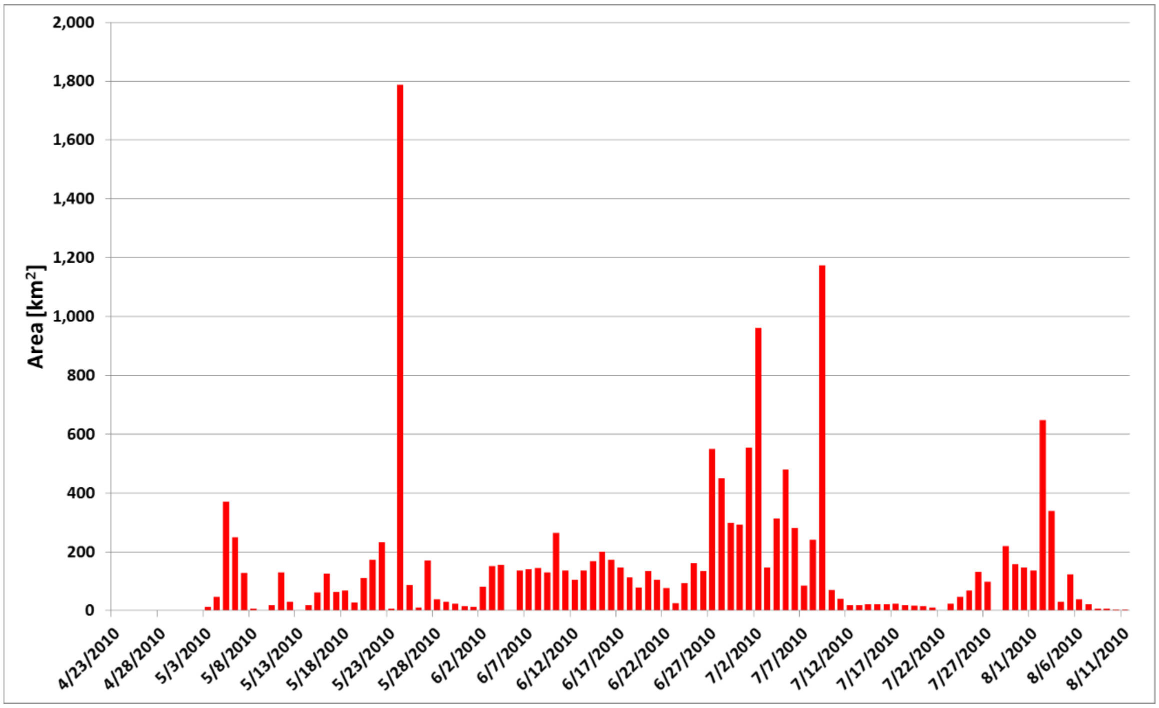
4.4.5.1.1 Surface Slick and Subsurface Mixed Zone

The SAR images demonstrate that oil was present on the water in the Gulf of Mexico from at least April 23, 2010, until August 11, 2010. Figure 4.4-27 plots the areal extent of surface oil detected in the offshore, shelf, and estuarine areas by day; Figure 4.4-28 plots the same information for estuarine waters only.



Source: Travers et al. (2015b).

Figure 4.4-27. Estimated areal extent of surface oil during the DWH spill in offshore, shelf, and estuarine waters.

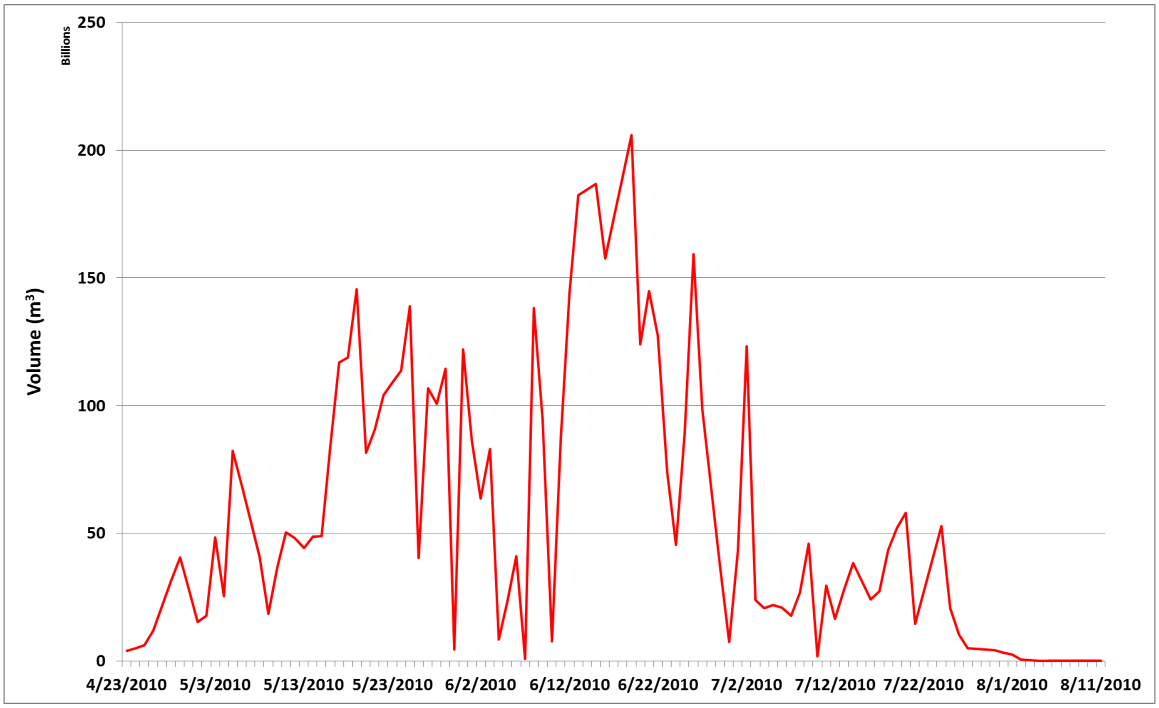


***Source**: Travers et al. (2015b).*

Figure 4.4-28. Estimated areal extent of surface oil during the DWH spill in estuarine waters.

Biota were exposed to oil in the upper water column on a daily basis over the 113 days that oil was present on the water surface. To account for this daily exposure, the daily areal extent of surface oil was summed for the duration of the spill. Across all surface waters, an estimated 1.23 million square kilometer-days (475,000 square mile-days) were exposed to floating oil. The potentially affected volume of water under the floating oil for each day of the spill was calculated assuming a depth of 20 meters (65 feet) for shelf and offshore areas and an estimated 26 percent of that water under floating oil exceeded 0.5 µg/L (Travers et al. 2015a). Figure 4.4-29 shows the total volume of water over each day; it shows the maximum volume of water potentially affected by the spill on 1 day is 210 billion cubic meters on June 19, 2010. The average daily volume of water exceeding a TPAH50 concentration of 0.5 µg/L is 57 billion cubic meters. To provide some context, the average annual discharge from the Mississippi River at New Orleans is 600,000 cubic feet per second (NPS 2015) or 1.5 billion cubic meters per day. Thus, the volume of water exceeding a TPAH50 concentration of 0.5 µg/L in the upper mixed layer was approximately 40 times the average daily discharge (based on average annual discharge) in the Mississippi River.

In addition, because oil was on the surface of the Gulf of Mexico for 113 days, the daily exposure can be added for each day of the spill to describe the cumulative affected volume in gallon-days. Summing the volume of water estimated to exceed 0.5 µg/L every day for the upper water column offshore and shelf areas, 6.3 trillion-cubic-meter days of water were potentially affected by DWH oil.



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

Figure 4.4-29. Estimated cubic meters of water affected by surface oil slicks from the DWH spill. The red line represents the estimated volume of water in billions of cubic meters exceeding a TPAH50 concentration of 0.5 µg/L from April 23 to August 11, 2010.

Based on the empirical chemistry data, the Trustees estimated that 26 percent of the water at depths between 0 and 20 meters in the offshore and shelf zones had TPAH50 concentrations greater than 0.5 µg/L (Travers et al. 2015a), which is sufficiently high to harm sensitive life stages of biota in the presence of UV light (Section 4.3, Toxicity).

Using the methods described, the Trustees estimated 4–6 percent mortality for invertebrates offshore and 21–45 percent mortality for larval fish offshore (Table 4.4-5). The total number of larval fish and invertebrates killed in the upper 20 meters of the offshore surface waters was estimated to be between 2 and 5 trillion and 37 and 68 trillion, respectively. The fish larvae killed included 30 to 400 billion herring (menhaden and relatives), 80 to 500 billion anchovies, 20 to 100 billion snappers, and 70 to 400 billion tunas and mackerels. Detailed calculations, tabulating results by species, are in the RPS Applied Science Associates (ASA) Technical Report (French McCay et al. 2015d).

Note that not all larval fish are expected to survive past a year. Depending on the species, the range in survival, from the larval size captured in survey nets to 1 year of age, is from approximately one in a thousand to one in 5 million. Looking across all species, this gives a vast range, from millions to billions, of fish that would have reached a year old if they had not been killed by the spill. Additionally, the larval fish that were killed but would not have survived to age 1 are a significant loss. Section 4.4.1.2 explains that larval fish are an important component of the plankton community that form the base of the aquatic food web and provide an energy source for other components of the ecosystem.

Table 4.4-5. Estimated percent mortality from oil exposure.

	Less Sensitive Species	More Sensitive Species
Offshore	**Percent of total in upper 0–20 meters of water column**	
Eggs and larval fish	21%	45%
Invertebrates	4%	6%
Estuarine	**Percent of total 2.5 meters of water column**	
Eggs and larval fish	4%	6%
Invertebrates	5%	5%

For estuarine waters, the Trustees estimate 16,000 square kilometer-days (6,200 square mile-days) of the total estuarine surface area were exposed to floating oil for the duration of the spill. The volume of water affected by the floating surface oil in estuaries, assuming UV light penetrates to a depth of 0.2 meters (0.6 feet) in the turbid estuarine waters, was 3 billion cubic-meter days. Based on a range of sensitivities, the Trustees estimate 4 to 6 percent mortality for larval fish in the estuarine waters and the total number of larval fish killed is 0.4 to 1 billion. The estimated larval invertebrate and small zooplankton (French McCay et al. 2015d) mortality of 5 percent in estuarine waters results in 2 to 6 trillion planktonic invertebrates killed. Table 4.4-6 summarizes injury quantification metrics for the surface and subsurface mixed zone in the offshore, shelf, and estuarine areas (French McCay et al. 2015d; Travers et al. 2015b).

Table 4.4-6. Metrics used by Trustees for upper water column injury quantification.

Metric	Quantification (with Range Where Applicable)
Area covered by surface oil slick	
Maximum daily areal extent of oil on surface during the spill (offshore, shelf, and estuarine)	39,700 km^2 (15,300 mi^2) on June 19, 2010
Average daily areal extent of oil on surface during the spill (offshore, shelf, and estuarine)	11,100 km^2 (4,300 mi^2)
Cumulative areal extent of oil on surface during the spill (offshore, shelf, and estuarine)	112,000 km^2 (43,300 mi^2)
Area of oil on surface for duration of the spill, sum of daily footprints (offshore, shelf, and estuarine)	1,229,000 km^2-days (475,000 mi^2-days) [a]
Area of oil on estuarine waters for duration of spill, sum of daily footprints	15,600 km^2-days (6,000 mi^2-days)
Volume of water affected by surface oil slick	
Maximum daily volume of water under the slick exceeding TPAH50 of 0.5 µg/L (offshore/shelf)	2.1×10^{11} (210 billion) m^3 on June 19, 2010
Average daily volume of water under the slick exceeding TPAH50 of 0.5 µg/L (offshore/shelf)	5.7×10^{10} (57 billion) m^3
Volume of water under the slick exceeding TPAH50 of 0.5 µg/L, sum of daily volumes (offshore/shelf)	6.3×10^{12} (6.3 trillion) m^3-days[b]
Average daily volume of affected estuarine waters	3.1×10^{7} (31 million) m^3
Affected volume of estuarine waters, sum of daily volumes	3.1×10^{9} (3 billion) m^3-days

Metric	Quantification (with Range Where Applicable)
Percent mortality	
% invertebrate mortality in 0–20 m (offshore)	4–6%
% egg and larval mortality in 0–20 m (offshore)	21–45%
% invertebrate mortality, average depth 2.5 m (estuaries)	5%
% egg and larval mortality, average depth 2.5 m (estuaries)	4–6%
Biota directly killed	
Direct kill: Total number planktonic invertebrates killed (offshore)	37×10^{12} to 68×10^{12} (37 to 68 trillion)
Direct kill: Total number larval fish killed (offshore)	2×10^{12} to 6×10^{12} (2 to 5 trillion)
Direct kill: Total number planktonic invertebrates killed (estuaries)	2×10^{12} to 6×10^{12} (2 to 6 trillion)
Direct kill: Total number larval fish killed (estuaries)	4×10^{8} to 1×10^{9} (0.4 to 1 billion)

[a] A km^2-day is a compound unit that means 1 square kilometer for one day, in any combination of area and time. For example, 100,000 km^2-days could mean 1,000 km^2 for 100 days, 10,000 km^2 for 10 days, or 100,000 km^2 for 1 day.

[b] An m^3-day is a compound unit that means 1 cubic meter for 1 day, in any combination of area and time. For example, 100,000 m^3-days could mean 1,000 m^3 for 100 days, 10,000 m^3 for 10 days, or 100,000 m^3 for 1 day.

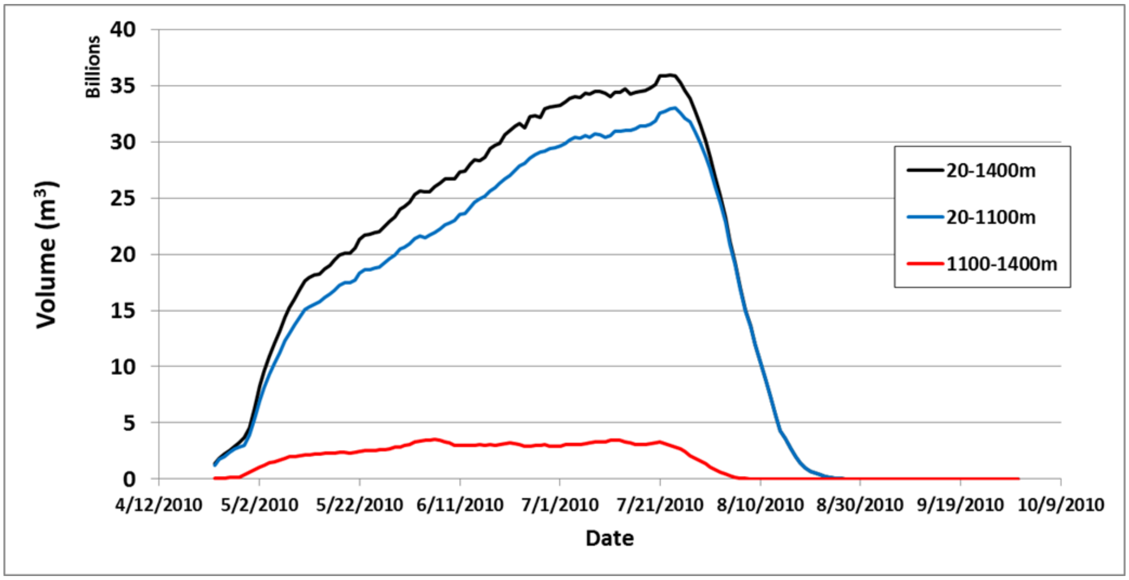
4.4.5.1.2 Rising Oil and Deep Plume of Oil at Depth

The model results provide a daily estimate of TPAH50 concentration distributions at depth. Table 4.4-7 summarizes the maximum daily volume of water with TPAH50 concentrations greater than 0.5 µg/L for the three depth zones.

Table 4.4-7. Model results for volume of deep water exceeding 0.5 µg/L TPAH50 and maximum TPAH50 concentration.

Portion	Depth Range		Maximum over time			Cumulative volume	
	Meters	Feet	Max Conc. (µg/L)	Volume (m^3)	Vol (gal)	m^3-days	gal-days
Cone	20–1,100	66–3,609	217	3.3×10^{10} (33 billion)	8.7×10^{12} (8.7 trillion)	2.6×10^{12} (2.6 trillion)	6.5×10^{14} (650 trillion)
Deep Plume	1,100–1,400	3,609–4,600	872	3.5×10^{9} (3.5 billion)	9.3×10^{11} (930 billion)	2.6×10^{11} (260 billion)	6.7×10^{13} (67 trillion)
Total	20–1,400	66–4,600	872	3.5×10^{10} (35 billion)	9.5×10^{12} (9.5 trillion)	2.7×10^{12} (2.7 trillion)	7.2×10^{14} (720 trillion)

Figure 4.4-30 shows the result of the modeled daily water volumes with TPAH50 concentrations greater than 0.5 µg/L in the depth of the rising cone and the deep plume for the duration of the oil release. The total cumulative volume of water exceeding this concentration was calculated by adding the volume for each day over the spill. The maximum volume was estimated at 35 billion cubic meters and the cumulative volume was 2.7 trillion cubic meter-days.



Source: *French McCay et al. (2015c).*

Figure 4.4-30. Daily volume of deep water with TPAH50 concentrations greater than 0.5 µg/L for oil released between April 22 and July 15, 2010. The lines represent the volume of water, in billions of cubic meters, for different water column depth zones. The rising cone (blue line) represents water depths up to 1,100 meters, and the deep plume (red line) represents depths between 1,100 and 1,400 meters beneath the surface. The black line is the sum of the total volume contributions from the two different subsurface zones.

The Trustees estimate the total number of larval fish killed in the deep water offshore is between 86 million and 26 billion and the total number of invertebrates killed is between 10 million and 7 billion (Table 4.4-8). The invertebrate life stages that were captured by the sampling gears were used to estimate baseline densities. They were also assessed in the injury quantification and include: microzooplankton that spend their entire life in the plankton (e.g., copepods, amphipods, mysids, krill), planktonic larval stages of benthic invertebrates (e.g., worms, barnacles, anemones, mantis shrimp), and planktonic early life history (primarily larval) stages of larger invertebrates (e.g., shrimp, crabs, lobsters, jellyfish, comb jellies, cephalopods, sea slugs, tunicates).

Table 4.4-8. Metrics used by Trustees for deep water injury quantification.

Metric	Quantification
Direct kill: Total number invertebrates killed (offshore)	$1x10^7$ to $7x10^9$ (10 million to 7 billion)
Direct kill: Total number larval fish killed (offshore)	$8.6x10^7$ to $2.6x10^{10}$ (86 million to 26 billion)

Direct Kill Estimate Considerations

The abundance estimates for juvenile fish and invertebrates throughout different water column areas are likely underestimates based on the inefficiency of the nets being used to estimate abundance (Johnson & Morse 1994; Morse 1989; Somerton & Kobayashi 1989) and the small volume of samples over a vast area. The nets used to capture fish larvae and invertebrates are not 100 percent efficient with some larger larvae avoiding the nets and smaller larvae potentially going through the nets' mesh. In addition, net samples were used in areas that do not include *Sargassum*, where the densities of some fish larvae and invertebrates are higher. By applying these lower abundance estimates, the number of fish and invertebrates estimated to be killed would also be an underestimate.

Despite the fact that the number of fish and invertebrates killed may be considered an underestimate, we can evaluate minimal and maximal impacts of the spill on fish larvae by considering the percentage of fish larvae that overlapped with oil compared to the entire U.S. Gulf of Mexico Exclusive Economic Zone (EEZ). By using this approach and considering larval densities as an index instead of an absolute value, for the species investigated the total percent overlap with floating oil ranged from approximately 0.1 percent to 7.5 percent of the larvae spawned across the entire EEZ during the spill (Quinlan et al. 2015). Further considering the toxicity under the footprint yields an impact from 0.05 percent to 3.38 percent of the total spawn during the spill.

Though this discussion presented impact in terms of percentages of the entire EEZ, it must be noted that impacts could easily be much more pronounced and that localized impacts may be particularly important. If, rather than considering the entire EEZ, the analyses considered only high quality habitat or some reduced area, the percentages impacted would be higher. Additionally, production in some areas may be critically important for some species. There is a vast literature on connectance in larval fish ecology and the importance of spawning in the correct time and place (Cowen et al. 2007; Cowen & Sponaugle 2009; Paris & Cowen 2004; Quinlan et al. 1999). Injury to areas that produce fish that settle elsewhere could mean that the net impact was larger and more nuanced than depicted in these analyses.

4.4.5.1.3 Production Foregone

Quantification of production foregone is presented for example species in Table 4.4-9. Production foregone was only calculated for selected species—those for which the Trustees determined that the needed vital rates were reliable (i.e., for species well-studied by fisheries managers). Production foregone totals across all species affected by the spill were not calculated or assessed.

The examples shown in Table 4.4-9 serve to illustrate that the weight of larvae killed is only a small portion of the impact to the species and ecosystem, as the larvae would have grown and been predated over their natural lifespan. The direct kill numbers in the table are the weights of larvae killed, whereas production foregone numbers are the weight gains that they would have undergone if not killed by the spill. For some species of fish that grow very large, such as amberjack, large tunas, and mahi-mahi, growth after the larval stage is very rapid and the production foregone represents the majority of the biomass loss. On the other hand, small fish (e.g., spot, anchovies) do not grow as rapidly and their mortality rates are much higher, so production foregone is of similar magnitude to the weight of the directly killed larvae. Results for these and other species may be found in the RPS ASA Technical Report (French McCay et al. 2015d).

Table 4.4-9. Production foregone calculations for example fish and invertebrate species in the offshore, shelf, and estuarine waters.

Species	Direct Kill #'s	Direct Kill (kg)	Production Foregone (kg)	Total Injury (kg)
Red snapper (*Lutjanus campenchanus*)	2×10^9 –7×10^9 (2 billion to 7 billion)	8×10^2 –3×10^3 (800 to 3 thousand)	5×10^4 –2×10^5 (50 thousand to 200 thousand)	5×10^4 –2×10^5 (50 thousand to 200 thousand)
Seatrout (*Cynosicon* spp.)	2×10^{10} –1×10^{11} (20 billion to 100 billion)	1×10^3 –7×10^3 (1 thousand to 7 thousand)	8×10^3 –5×10^4 (8 thousand to 50 thousand)	9×10^3 –6×10^4 (9 thousand to 60 thousand)
Spot (*Leiostomus xanthurus*)	4×10^8 –1×10^{10} (400 million to 10 billion)	2×10^1 –5×10^2 (20 to 500)	3×10^1 –7×10^2 (30 to 700)	5×10^1 –1×10^3 (50 to 1 thousand)
Atlantic croaker	1×10^5 –6×10^9 (100 thousand to 6 billion)	5×10^{-3} –3×10^2 (5 thousandths to 300)	2×10^{-2} –8×10^2 (2 hundredths to 800)	3×10^{-2} –1×10^3 (3 hundredths to 1 thousand)
Spanish mackerel (*Scomberomorus maculatus*)	3×10^9 –3×10^{10} (3 billion to 30 billion)	3×10^2 –2×10^3 (300 to 2 thousand)	4×10^2 –4×10^3 (400 to 4 thousand)	7×10^2 –6×10^3 (700 to 6 thousand)
Amberjack (*Seriola* spp.)	2×10^8 –3×10^{10} (200 million to 30 billion)	2×10^1 –7×10^2 (20 to 700)	6×10^3 –2×10^5 (6 thousand to 200 thousand)	6×10^3 –2×10^5 (6 thousand to 200 thousand)
Large tunas (*Thunnus* spp.)	2×10^{10} –1×10^{11} (20 billion to 100 billion)	2×10^3 –8×10^3 (2 thousand to 8 thousand)	1×10^6 –4×10^6 (1 million to 4 million)	1×10^6 –4×10^6 (1 million to 4 million)
Mahi-mahi (*Coryphaena* spp.)	2×10^8 –4×10^9 (200 million to 4 billion)	4×10^1 –8×10^2 (40 to 800)	8×10^4 –2×10^6 (80 thousand to 2 million)	8×10^4 –2×10^6 (80 thousand to 2 million)
Anchovies (Engraulidae)	8×10^{10} –7×10^{11} (80 billion to 700 billion)	4×10^4 –2×10^5 (40 thousand to 200 thousand)	6×10^4 –4×10^5 (60 thousand to 400 thousand)	1×10^5 –6×10^5 (100 thousand to 600 thousand)
Gulf shrimp (Penaeids & similar)	2×10^{11} –5×10^{11} (200 billion to 500 billion)	3×10^3 –1×10^4 (3 thousand to 40 thousand)	2×10^5 –9×10^5 (200 thousand to 900 thousand)	2×10^5 –9×10^5 (200 thousand to 900 thousand)
Blue crabs (*Callinectes* spp.)	2×10^9 –1×10^{10} (2 billion to 10 billion)	2×10^2 –1×10^3 (200 to 1 thousand)	1×10^2 –7×10^2 (100 to 700)	3×10^2 –2×10^3 (300 to 2 thousand)

4.4.5.2 Fisheries Analysis

NOAA's NWFSC analyzed fisheries-independent survey data to examine changes in fish and invertebrate populations before and after the DWH oil spill. The analysis of the SEAMAP data series did not detect any major or unusual changes in CPUE in pre- or post-spill periods, although there was evidence of modest post-spill decreases in areas west of the Mississippi River Delta and concurrent modest increases in areas east of the Delta. Similarly, analyses of LDWF data did not detect any widespread or unusual changes in CPUE in pre- or post-spill periods.

Although no substantial or widespread changes in fisheries populations were detected, the Trustees cannot rule out the possibility that fisheries were impacted at a population level. Estimates of population abundance take several years to complete and verify, making these difficult to evaluate for a large-scale event, such as an oil spill. Changes in life history features, such as size at age structure, growth rates, liver function, and condition are immediate and better indicators of a stress event.

In addition, statistical approaches for detecting trends in relative abundance are often limited by the statistical power to detect changes. A very large suite of biotic and abiotic factors lead to a high degree of natural variation in estimates of annual abundance, even in the absence of a major impacting event such as an oil spill. The effects of such an impact on fish abundance, even though it may be substantial, can be statistically undetectable unless the magnitude of the effect is large enough to exceed the degree of natural variation. When statistical power is limited, it can be increased by increasing sample sizes or by reducing the degree of variation in the sampling process. However, for practical reasons (e.g., the historical record of fish abundance is already completed and not specifically designed for injury assessment), neither of these tactics is an available option in this injury assessment.

4.4.5.3 *Sargassum* Injury Quantification

The Trustees quantified injury related to *Sargassum* in two ways: 1) area of *Sargassum* oiled and 2) percent of *Sargassum* area foregone due to lost growth.

4.4.5.3.1 Area of Oiled *Sargassum*

Based on an analysis of satellite and low-altitude aerial imagery (described in Section 4.4.2.3), the Trustees determined that 2.8 percent of the northern Gulf of Mexico was covered by *Sargassum* during the period of the spill (95 percent confidence interval ranging from 1.8 to 3.8 percent). Multiplying these percent cover estimates by the area of surface oiling values in Section 4.4.3.3, we produced the area of *Sargassum* oiled by the DWH spill (Table 4.4-10). The total amount of *Sargassum* oiled ranges from 843 to 1,749 square kilometers within areas where the surface was covered by greater than 5 percent thick oil. This includes 479 to 993 square kilometers within areas where coverage was greater than 10 percent thick oil. Overall, 23 percent of the *Sargassum* in the northern Gulf of Mexico was lost due to co-location with ocean surface areas with greater than 5 percent thick oil, and 13 percent of the *Sargassum* was lost due to co-location with ocean surface areas with greater than 10 percent thick oil.

Table 4.4-10. Area of *Sargassum* within oiling footprint (km^2).

% Thick Oil	Lower Bound	Central Estimate	Upper Bound
> 5%	843	1,296	1,749
> 10%	479	736	993

4.4.5.3.2 *Sargassum* Area Foregone

An additional measure of *Sargassum* injury is the surface area foregone due to lost growth caused by oil exposure. As *Sargassum* moves across the northern Gulf of Mexico, it grows at a rate of 4 percent per day (LaPointe 1986; as cited in Powers 2012). Therefore, the Trustees were able to calculate ranges of surface area foregone of 4,524 to 9,392 square kilometers for the greater than 5 percent thick oil footprint with a subset of 2,569 to 5,334 square kilometers for the greater than 10 percent thick oil footprint (Table 4.4-11).

Table 4.4-11. *Sargassum* surface area foregone (km²).

% Thick Oil	Lower Bound	Central Estimate	Upper Bound
> 5%	4,524	6,958	9,392
> 10%	2,569	3,952	5,334

Sargassum represents a rich environment in the open ocean that supports a high density of fauna, ranging from fish larvae and invertebrates that live on it, to the larger fish, sea turtles, and sea birds that rely on it for foraging and protection from predators. Therefore the oiling of this *Sargassum* and the loss of *Sargassum* area due to oiling and lost growth represents an important aspect of the overall water column injury.

4.4.5.4 Lack of Observed Dead Biota

Although the Trustees have concluded that exposure to DWH oil killed an unprecedented number of water column organisms in the northern Gulf of Mexico, it is important to understand that there is no reason to expect those effects to have manifested as a large, observable number of dead fish and invertebrates. A number of factors contribute to the low likelihood of visually observing the mortality quantified by the Trustees, including the following:

- Much of the injury to fish and invertebrates occurred in small, early-life stage organisms that would not have been seen. For context, these early-life stage organisms are generally smaller than the letters on this page.
- The spatial extent of injury was vastly greater than any practicably observable area and the exposure and resulting injuries were not uniform over the timespan and geographic extent of the spill.
- Dead fish would have been subject to rapid predation and decomposition. The likelihood of observing dead fish prior to predation or decomposition is extremely low given the spatial scales involved in the injury.
- The toxic effects of oil exposure to juvenile and adult fish are more likely to result in chronic injuries that will manifest differently by individual, rather than an acute effect that immediately kills a large school of fish.

This conclusion is also supported in the DWH Phase III expert report, which concludes that “oil pollution does not usually produce large fish kills, but affects populations through adverse effects on survivability, reproduction, prey, and habitats” (Boesch 2014).

4.4.6 Conclusions and Key Aspects of the Injury for Restoration Planning

The DWH incident resulted in a large, continuous release of oil into the northern Gulf of Mexico. The oil was released 1,500 meters deep over 87 days. It reached the surface and was transported hundreds of kilometers by currents, winds, and waves. As a result, a highly diverse group of water column inhabitants were exposed to oil and injured.

As described in further detail below, the injury assessment showed the following:

- The Trustees estimated the spill resulted in a surface slick that covered a cumulative area of at least 112,100 square kilometers (43,300 square miles) for 113 days in 2010. The average daily extent of the oil footprint was 11,100 square kilometers (4,300 square miles).
- The estimated average daily volume of water under surface oil slicks exceeding a TPAH50 concentration of 0.5 µg/L was 57 billion cubic meters. As a comparison, this volume is approximately 40 times the average daily discharge of the Mississippi River at New Orleans.
- Water column resources injured by the spill include species from all levels in the northern Gulf of Mexico food web. Affected organisms include bacteria, estuarine-dependent species (e.g., red drum, shrimp, seatrout), and large predatory fish (e.g., bluefin tuna) that can migrate from the Gulf of Mexico into the Atlantic and as far as the Mediterranean Sea.
- The Trustees estimated that 2 to 5 trillion larval fish and 37 to 68 trillion invertebrates were killed in the surface waters as a result of floating oil and mixing of that oil into the upper water column. In the deep waters, the Trustees' assessment showed that exposure to DWH oil resulted in the death of between 86 million and 26 billion fish larvae and between 10 million and 7 billion planktonic invertebrates. Of these totals, 0.4 to 1 billion larval fish and 2 to 6 trillion invertebrates were killed in estuarine surface waters. The larval loss likely translated into millions to billions of fish that would have reached a year old had they not been killed by the spill. Larval fish that were killed but would not have survived to age 1 are also a significant loss; they are an energy source for other components of the ecosystem. The Trustees determined that additional injuries occurred, but these were not quantified. Examples include adverse effects to fish physiology (e.g., impaired reproduction and reduced growth) and adverse effects to reef fish communities (e.g., reductions in abundance and changes in community composition).
- The Trustees also quantified injury to *Sargassum,* a brown algae that is habitat for many marine animals. Up to 23 percent of the *Sargassum* in the northern Gulf of Mexico was lost due to direct exposure to DWH oil. The total loss of *Sargassum*, including foregone area from lost growth due to exposure to this oil, is 11,100 square kilometers.

The Trustees considered all of these aspects of the injury in restoration planning, and also considered the ecosystem effects and recovery information, described below.

4.4.6.1 Exposure

The Trustees used remote sensing imagery to quantify the area of surface oil observed floating on the ocean surface for the duration of the spill. Based on this imagery, the Trustees estimated the spill resulted in a surface slick that covered a cumulative area of at least 112,100 square kilometers (43,300 square miles) for 113 days in 2010. This surface oil slick occurred in an area of high biological abundance, diversity, and productivity. Furthermore, the event occurred during a time of year (spring and summer) when seasonal productivity peaks in the northern Gulf of Mexico.

To estimate the spill's impacts on water column biota, the Trustees quantified the direct kill and production foregone of fish and invertebrates exposed to DWH oil both in the surface slick and in the subsurface mixed zone. The concentrations of PAHs in water below the surface slick were estimated using empirical chemistry data from water samples collected during the time oil was present on the water. The number of biota exposed to either the surface slick or lethal concentrations of PAHs was estimated from historical biological collections, NRDA field studies, and the literature. The number of biological organisms killed due to direct oil slick exposure or due to exposure to lethal concentrations of PAHs was estimated using data synthesized from NRDA-specific field studies, NRDA toxicity testing studies, and the published literature.

In addition to impacts from exposure to the surface slick and entrained oil from the surface slick, biota occupying the deeper water column (more than 20 meters beneath the surface) were also impacted by the cone of rising large oil droplets, dissolved components, and the deep water plume—a "cloud" of small oil droplets and dissolved contaminants. The NRDA sampling efforts in the deep ocean highlight the diversity and abundance of animals exposed to oil in the deep pelagic waters of the Gulf of Mexico. The Trustees used modeling and empirical data to quantify the direct kill of fish and invertebrates exposed to the rising cone of oil and to the deep water plume.

The Trustees estimated that 2 to 5 trillion larval fish and 37 to 68 trillion invertebrates were killed in the surface waters as a result of floating oil and mixing of that oil into the upper water column. Of these totals, 0.4 to 1 billion larval fish and 2 to 6 trillion invertebrates were killed in estuarine surface waters. In the deep waters, the Trustees' assessment showed that exposure to DWH oil resulted in the death of between 86 million and 26 billion fish larvae and between 10 million and 7 billion planktonic invertebrates, respectively. Depending on the species, survival from the larval size captured in survey nets to 1 year of age ranges from approximately one in a thousand to one in several million (French McCay et al. 2015d). This translates into a loss of millions to billions of fish that would have reached age 1. Additionally, the larval fish that were killed but would not have survived to age 1 are a significant loss; they are an energy source for other components of the ecosystem.

In addition to the lethal injuries quantified by the Trustees, injuries to fish physiology and reef fish communities were observed at a number of locations following the DWH oil spill. PAH concentrations measured in the Gulf of Mexico water column exceeded levels known to cause sublethal toxic effects to water column organisms. Sublethal toxic effects can reduce an organism's health, fitness, and ability to reproduce and survive. Following the DWH oil spill, field-collected data documented effects on fish physiology, including impaired reproduction and reduced growth, which can be associated with reduced survival and fecundity. Tissue lesions were also observed in red snapper and other bottom dwelling fish species on the continental shelf in the northern Gulf of Mexico (Murawski et al. 2014). Furthermore, injuries to shelf-reef communities were observed, including reductions in abundance and changes in community composition (Patterson III et al. 2015; Patterson III et al. 2014). Species-specific data for red snapper, a key recreational and commercial species and focus of intensive fisheries management effort, indicate that other injuries included growth reductions (Herdter 2014; Neese 2014; Patterson III 2015), shifts in diet (Tarnecki 2014; Tarnecki & Patterson 2015), and increased prevalence of tissue lesions (Murawski et al. 2014). Exposure of DWH oil to these species was observed in carbon isotope ratios (Tarnecki & Patterson 2015) and was indicated by relatively high concentrations of PAHs in fish liver and bile (Murawski et al. 2014; Romero et al. 2014). Overall, although explicit quantification of these various injuries is not possible at this time, the Trustees concluded that fish and fish communities suffered physiologically and demographically important injuries in hard-bottom habitats along portions of the continental shelf.

The Trustees also quantified injury to *Sargassum,* a brown algae that creates essential habitat for invertebrates, fish, birds, and sea turtles. Trustees quantified the loss of *Sargassum* resulting from direct oiling and also the area of *Sargassum* foregone due to lost growth. Up to 23 percent of the *Sargassum* (1,749 square kilometers) in the northern Gulf of Mexico was lost due to direct exposure to DWH oil on the ocean surface. In addition, foregone *Sargassum* area from lost growth due to exposure to this oil was estimated to be as large as 9,400 square kilometers.

4.4.6.2 Ecosystem Effects

In addition to the quantification of lost individuals, the Trustees also considered potential ecosystem effects, including foodweb and ecological function impacts.

As discussed above, the Gulf of Mexico is a complex and interconnected ecosystem, composed of diverse habitats and species and important ecological processes. When natural resources are injured, cascading ecological effects can occur, including changes in trophic structure (such as altering predator-prey dynamics), community structure (such as altering the composition of organisms in an area), and ecological functions (such as altering the flow of nutrients).

Numerous studies have modeled the extensive food web of the Gulf of Mexico (e.g., Althauser 2003; Clough et al. 2015; de Mutsert et al. 2012; Masi et al. 2014; Tarnecki et al. 2015; Walters et al. 2008) with energy flowing from primary producers to large predators and trophic relationships connecting the nearshore and offshore as well as the surface and deep. Impacts to a specific resource could cause direct and indirect effects cascading throughout the food web (Fleeger et al. 2003; Peterson et al. 2003) (Fodrie et al. 2014; Tarnecki et al. 2015). For example, bottom-up trophic impacts could occur if an important food base, such as plankton or a forage fish, were impacted. Alternatively, impacts to a species higher on the food chain could reduce predation pressure, resulting in potential changes to the community structure and interspecific (between species) and intraspecific (within species) dynamics. Resources discussed in other sections, such as sea turtles, birds, and marine mammals, are also connected to the water column ecosystem through foodweb dynamics. Injuries to these resources could also cause potential trophic cascades to water column species.

Another potential concern includes impacts to ecological functions and processes. As discussed in Section 4.4.1, water column resources are important vectors of energy, both vertically and horizontally. Thus, impacts to a particular resource could alter the flow of organic carbon or nutrients through the water column, resulting in indirect effects to additional species and habitats.

As discussed in other sections of this document, nearshore and benthic environments are important habitats for many species found within the water column, serving as nursery grounds, foraging habitat, and refuge from predators. During some part of their life cycles, many estuarine-dependent fish and crustaceans rely on nearshore habitats, such as marshes, submerged aquatic vegetation, and oyster reefs. As such, habitat losses, as described in Section 4.6 (Nearshore Marine Ecosystem), could cascade to impacts to water column resources.

4.4.6.3 Uncertainties

Given the magnitude and ongoing nature of the DWH oil spill, it was not possible to measure the locations of all species and water column oil concentrations over all affected areas. Thus, the Trustees inferred this information from available data and models. The approach to estimating mortality in the upper water column requires several assumptions that introduce uncertainty regarding the precision of the inferences and estimates that cannot be readily quantified. Among these are:

- The actual vertical distribution of the eggs and invertebrates in the upper water column is well-represented by the assumed distribution.
- The observed TPAH50 distribution is representative of the distribution of concentrations that were present in the areas of floating oil during the spill.
- The duration of egg or invertebrate exposure to TPAH50 is representative of exposure durations in the toxicity tests.
- The laboratory toxicity tests are applicable to the field conditions during the DWH spill.

Because this uncertainty cannot be known and quantified, the resulting effect on the upper and lower range of estimated mortalities cannot be determined, and the range may be greater than that expressed. However, the Trustees have relied on the best available information when making the above assumptions and believe the range of injury presented is reasonable and provides sufficient certainty to aid in restoration planning.

4.4.6.4 Recovery

The water column contains a diverse array of species, occupying different niches and interacting in a complex web of production, consumption, and decomposition. Water column resources injured by the spill include species from all levels in the food chain. Affected organisms include bacteria, estuarine-dependent species (e.g., red drum, shrimp, seatrout), and large predatory fish (e.g., bluefin tuna). With so many species, zones, and interconnections (see Section 4.4.1 for more detail), predicting natural recovery of the Gulf of Mexico water column ecosystem is necessarily also complex. While some organisms are expected to recover quickly, others may take many years to decades to fully recover. Small forage fish typically have high rates of turnover and thus might be expected to recover more quickly than longer lived fish like large tunas and some reef fish that can live for decades. However, the interactions among species and the feedbacks from one organism to another may alter these perceived recovery trajectories. Restoring key parts of the system that were injured will increase recovery rates for components of the ecosystem that were impacted and help to compensate for the losses that occurred over the recovery period.

4.4.6.5 Restoration Considerations

As described in Chapter 5, to restore for injuries to water column resources, the Trustees identified restoration that could benefit water column resources directly by reducing excess sources of mortality. The Trustees also identified restoration that can benefit water column resources indirectly, by restoring the habitats and biological relationships that these resources depend on, including restoring coastal and benthic habitats.

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4.5 Benthic Resources

What Is in This Section?

- **Executive Summary**
- **Introduction and Importance of the Resource (Section 4.5.1):** What are benthic resources and why do we care about them?
- **Approach to the Assessment (Section 4.5.2):** How did the Trustees assess injury to benthic resources?
- **Exposure (Section 4.5.3):** How and to what extent were benthic resources exposed to *Deepwater Horizon* (DWH) oil and response activities?
- **Injury Determination (Section 4.5.4):** How did exposure to DWH oil affect benthic resources?
- **Injury Quantification (Section 4.5.5):** What was the magnitude of injury to benthic resources?
- **Conclusions (Section 4.5.6):** What are the Trustees' conclusions about injury to benthic resources, ecosystem effects, and restoration considerations?
- **References (Section 4.5.7)**

Executive Summary

Diverse and abundant natural resources are typically plentiful on the ocean floor across the northern Gulf of Mexico. Corals, fish, crabs, and a myriad of small animals and microbes live in a variety of habitats on the sea bottom and are part of the foundation of life and food webs in the northern Gulf of Mexico. The effects of the DWH oil spill were documented across a wide variety of these benthic and shoreline habitats and communities. The Trustees designed and implemented an assessment of injuries to representative benthic resources generally grouped by depth for purposes of the Natural Resource Damage Assessment (NRDA). These include benthic resources in the deep sea, on the continental slope, and on the continental shelf.

Study designs and assessment priorities were based on a conceptual model developed by the Trustees to assess contaminant pathways and exposures of benthic resources. The study designs incorporated results from research and NRDA activities. Study priorities reflected information available from spill response activities and from efforts incorporated into investigative cruises planned prior to the spill. The benthic assessments focused on a variety of resources, including animals that live on and in the prevalent soft-bottom sediments, on isolated and rare hardground coral habitats, and on mesophotic reefs along the continental shelf edge. Despite constraints, including the challenges of working in the deep ocean, the vastness of the spill itself, and limitations on our understanding of deep-sea ecosystems, the Trustees documented a footprint of over 770 square miles (2,000 square kilometers) of injury to benthic habitat surrounding the wellhead. That footprint is described in this section as four separate zones with varying types of injuries documented in each of the zones.

The zones appear as a bull's-eye pattern around the wellhead, with the area closest to the wellhead documented with multiple and the most severe spill-related losses to the benthos. Moving away from the wellhead, the zones increase in total area, the numbers and types of injuries documented are fewer, and uncertainty increases. The innermost zone, representing an area of 11 square miles (28 square kilometers) exhibited injuries ranging from smothering with drilling muds, toxic concentrations of oil, and a reduction by half in the diversity of sediment-dwelling animals that did survive. The second and third concentric zones (covering areas of 75 and 306 square miles [195 and 793 square kilometers], respectively), exhibited a different suite of ecological impacts, ranging from the mortality of corals at hardground sites to less dramatic reductions in the diversity of animals living in the sediment. Ultimately, within the outermost zone spanning 492 square miles (1,275 square kilometers), the chemical quality of the seafloor habitat was adversely affected by contamination, the food web was fouled, polycyclic aromatic hydrocarbon (PAH) concentrations in sediments from some locations in the zone exceeded toxicity values (LC20 and LC50), and PAH concentrations exceeded values reported by Schwing et al. (2015) as correlating with substantial declines in abundances of benthic foraminifera.

Outside of the zones noted above, an additional approximately 4 square miles (10 square kilometers) of mesophotic reef habitat on the continental shelf edge was also determined to have experienced significant losses to resident corals and fish. These losses likely contributed to a decline in ecological functions provided by this biologically rich and important location on the shelf edge.

The overall magnitude of ecological impacts from the resource losses that were quantified is not fully understood. The Trustees expect, though, that some impacts extend beyond these quantified areas, based on the dynamics of the Gulf, movements of animals, marine processes such as carbon recycling, and the overall interconnectedness of marine ecological functions. A larger area, approximately 3,600 square miles (9,200 square kilometers), of potential exposure and uncertain impacts from the spill extends beyond and between the areas where the Trustees quantified injury. The time needed for these habitats to naturally recover from effects of the spill without restoration will vary based on the sensitivities, growth rates, reproduction, and recruitment of individual component resources. In general, resource recovery is expected to be on the order of decades to hundreds of years, based on the uniformity of environmental conditions and slow progression of change in deep-sea environments, and the fact that some organisms killed by the spill were hundreds of years old (e.g., deep-sea coral).

4.5.1 Introduction and Importance of the Resource

Key Points

- Corals, fish, crabs, and a myriad of small animals and microbes live in a variety of habitats on the sea bottom and are part of the foundation of life and food webs in the northern Gulf of Mexico.
- Soft-bottom sediment is by far the dominant substrate type in the northern Gulf of Mexico. Hard substrate (including artificial reefs, oil and gas platforms, and natural reef or rock substrates) accounts for the remaining 4 percent.

- Both hard and soft substrate types support a wide variety of marine life, and many mobile animals move back and forth between the soft- and hard-bottom habitats.
- For purposes of the injury assessment, the Trustees grouped benthic resources based on the general depths at which they occur and evaluated resource injuries in the deep benthos, along the continental slope, and along the continental shelf.

Shortly after the well blowout and the explosion occurred on the DWH platform, oil spread across the sea surface. It was not immediately clear what was happening below the surface, and whether or not oil would spread underwater and affect natural resources in the water column, or settle onto benthic habitats, persist, and affect seafloor life—especially at such great depth. However, it was not long before the uncontrolled flow of oil at depth was well documented through live camera feeds, and the public learned about rising oil, subsurface plumes, and a variety of unsuccessful response activities employed to stem the flow of oil over a 3-month period until the flow was finally stopped. Given the release of oil at depth for months, the Trustees undertook an assessment of natural resources along the sea floor.

Diverse and abundant natural resources are typically plentiful on the ocean floor across the northern Gulf of Mexico (Gage 1996; Gjerde 2006; Grassle & Maciolek 1992; Llodra & Billet 2006; Rex & Etter 2010; Ruppert & Barnes 1993). Rare corals, fish, crabs, and a myriad of small animals and microbes live in a variety of habitats on the sea bottom and are part of the foundation of life and food webs in the northern Gulf of Mexico. The seafloor habitats and resident communities in the northern Gulf of Mexico are collectively referred to as benthic marine resources.

The Gulf of Mexico sea floor is a complex, heterogeneous environment. Sediment transported by the Mississippi River dominates the continental shelf and the deep sea (Balsam & Beeson 2003). Soft-bottom sediment is by far the dominant substrate type in the northern Gulf of Mexico (Love et al. 2013; Rezak et al. 1985). Froeschke and Dale (2012) attribute 96 percent of the Gulf floor to soft-bottom, and the total hard substrate (including artificial reefs, oil and gas platforms, and natural reef or rock substrates) accounts for the remaining 4 percent of the total area of the bottom. This hard substrate provides Essential Fish Habitat in the U.S. Exclusive Economic Zone of the Gulf of Mexico. Both hard and soft substrate types support a wide variety of marine life, with some species differences that tend to change with depth, among other environmental factors (Etnoyer 2009; Gallaway et al. 2001).

The deep-sea floor covers over half of the earth's surface and is dark and seemingly inhospitable—yet it has extensive and unique marine life adapted to the darkness, cold, and extreme pressures.

The deep-sea environment is arguably the least explored frontier on earth. We know less about the sea floor than the surface of the moon, and it is home to many rare and yet-to-be described species.

The deep-sea food web relies upon detritus, biological matter and debris, falling from above. The animals and microbes break down organic matter and recycle the carbon through the ocean system, which is vital to continued life on earth.

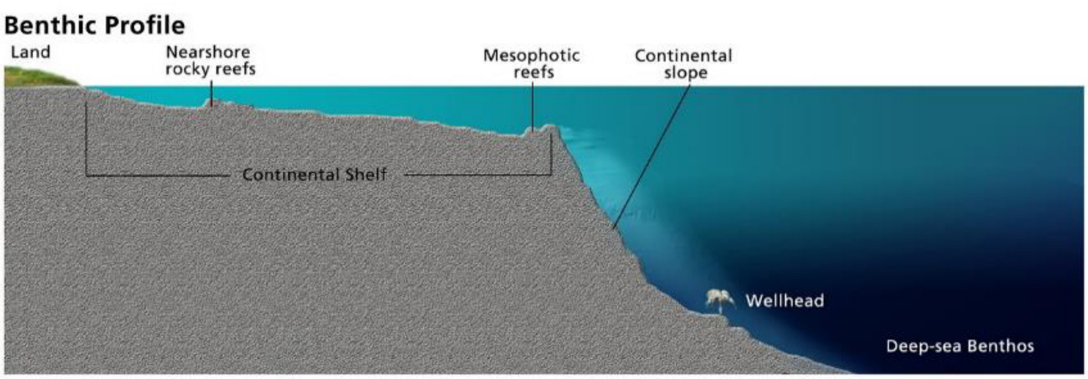
For purposes of the NRDA, the Trustees grouped benthic marine resources based on depths where they occur and by various prominent physical and biological features. There are no absolute biological or physical lines separating individual benthic habitats and communities that extend from the depths up across the continental shelf to the shoreline. Rather, as with all ecosystems, what appear to be distinct habitats in fact have transition zones, and many biota move between habitats and/or may thrive at the edges of habitat types.

The general regions of benthic habitat described in this section include the following (moving from the blown-out well toward shore):

1. **Deep benthos** (>800 meter depth), where life is adapted to the cold, dark, and relatively stable deep ocean and typically thrives on whatever food sources settle from the shallower depths of the ocean. Sediment is typically silt-clay, and hardground is dominated either by particle-scavenging corals or biological communities that are localized around and derive energy from hydrocarbon seeps (e.g., tubeworms) (Gallaway et al. 2001).
2. **Continental slope** (>200–800 meter depth), characterized by relatively rapid changes in depth over shorter horizontal distances. It is occasionally incised by canyons, and hardground is dominated by seeps or corals (Gallaway et al. 2001).
3. **Continental shelf** (10–200 meter depth), where life is dominated by the influence of light, the shoreline, and surface currents. Sediment is typically sand (Cooksey et al. 2014) and hardground habitats can be variable, with some supporting communities of reef forming corals and others supporting non-reef forming corals (e.g., mesophotic reefs in 50–150 meter depths along the edge of the continental shelf) (Sulak & Dixon 2015).

What Are Mesophotic Ecosystems?

Mesophotic coral ecosystems are characterized by the presence of light-dependent corals and associated communities found at water depths where light penetration is low. ("Meso" means "middle" and "photic" means "light.") The dominant communities providing structural habitat in the mesophotic depth zone can be made up of coral, sponge, and algal species. The fact that they contain zooxanthellae (algae that live in the cells of the coral) and require light distinguishes these corals from true deep-sea corals, though their depth ranges may overlap (NOAA 2011).



__Source:__ Kate Sweeney for NOAA.

Figure 4.5-1. Profile of regions of benthic habitats from shore to depth around wellhead.

Nearshore Benthic Resources

The Trustees included some nearshore benthic resources (e.g., oysters, shrimp, killifish, flounder, amphipods, submerged aquatic vegetation) within shoreline assessments, because of the role of these resources in shore edge communities or their usefulness for assessing impacts to shoreline habitat. Those nearshore species are not included in this discussion of impacts to benthic resources. Section 4.6, Nearshore Marine Ecosystem, discusses benthic resources occurring in approximately 10 meters (about 30 feet) or less depth. Many of these nearshore benthic resources are also located behind barrier islands, a useful geographic feature used to characterize nearshore benthic resources in Section 4.6.

4.5.2 Approach to the Assessment

Key Points

- The Trustees developed a conceptual model for evaluating contaminant exposures and conducting and prioritizing benthic assessment activities. The assessment focused on benthic areas in the vicinity of and extending away from the wellhead and where surface oil may have been entrained and sunk to the sea floor through a combination of physical and chemical factors.
- The Trustees considered and accounted for oil contributions from naturally occurring hydrocarbon seeps, though seeps did not play a role in causing resource injury relative to spill-related materials released during the DWH incident.
- The Trustees focused assessment activities on both the predominant soft sediment environment of the benthos, as well as the rarer hardground habitats located throughout the northern Gulf of Mexico. This includes soft sediment benthic biota, deep-sea hardground coral habitats, and mesophotic reef habitats, each of which is described in greater detail in this section.
- The Trustees used a variety of sampling techniques (including photography, videography, and collection of environmental media such as sediment and biological tissues) and statistical techniques (including before-after control-impact comparisons, principal components analyses, and spatial analyses) as part of the benthic assessment.

Scientists who work in deep-sea environments face many logistical challenges, including the difficulty of accessing offshore and deep ocean sites, restricted visibility because of darkness at depth, extreme cold, limited time available for making observations, and other constraints. The Trustees used specialized tools and techniques to overcome many of the logistical challenges. They prioritized the damage assessment work based on a conceptual model of where the oil likely traveled and which benthic resources might be at greatest risk of exposure. This approach led to more intensive sampling closer to the wellhead than farther away. As a consequence of the extremely large area potentially affected by the DWH spill and reduced benthic sampling density with distance from the wellhead, there was less accuracy in defining injuries farther afield. For many areas in the northern Gulf of Mexico, there was also limited pre-spill information for making pre- and post-spill comparisons.

4.5.2.1 Conceptual Model for the Approach

Documenting the multiple potential exposure pathways, and obtaining and analyzing data to confirm a pathway, was a multifaceted process. Much of the initial effort relied on evaluating information collected during response studies focused on areas near the blown-out well. Then the effort moved out in various directions based on what was known about currents and anticipated movement of the oil at the time of the spill. The Trustees also took advantage of partnering with any summer 2010 environmental sampling efforts that had already been planned prior to the oil spill.

Subsequent NRDA work was based on a conceptual model and data collected during and shortly after the active spill. The Trustees posed three possible explanations (hypotheses) related to where and how spilled materials would move and the anticipated fate of DWH oil in the northern Gulf of Mexico:

1. Exposures and impacts were likely to be greater at sites closer to the blown-out well and where subsurface plumes could directly contact habitats.

2. Exposures and impacts were likely to be greater at sites beneath persistent slicks, where contaminated materials could sink and rain down on underlying benthic habitats and biota.

3. Exposures and impacts were likely to be greater at sites where oil and other contaminants would become entrained in the water and potentially move downward to the sea floor, or where physical factors such as currents and bathymetry might work in combination to limit dispersion or even concentrate deposition of spill-related materials.

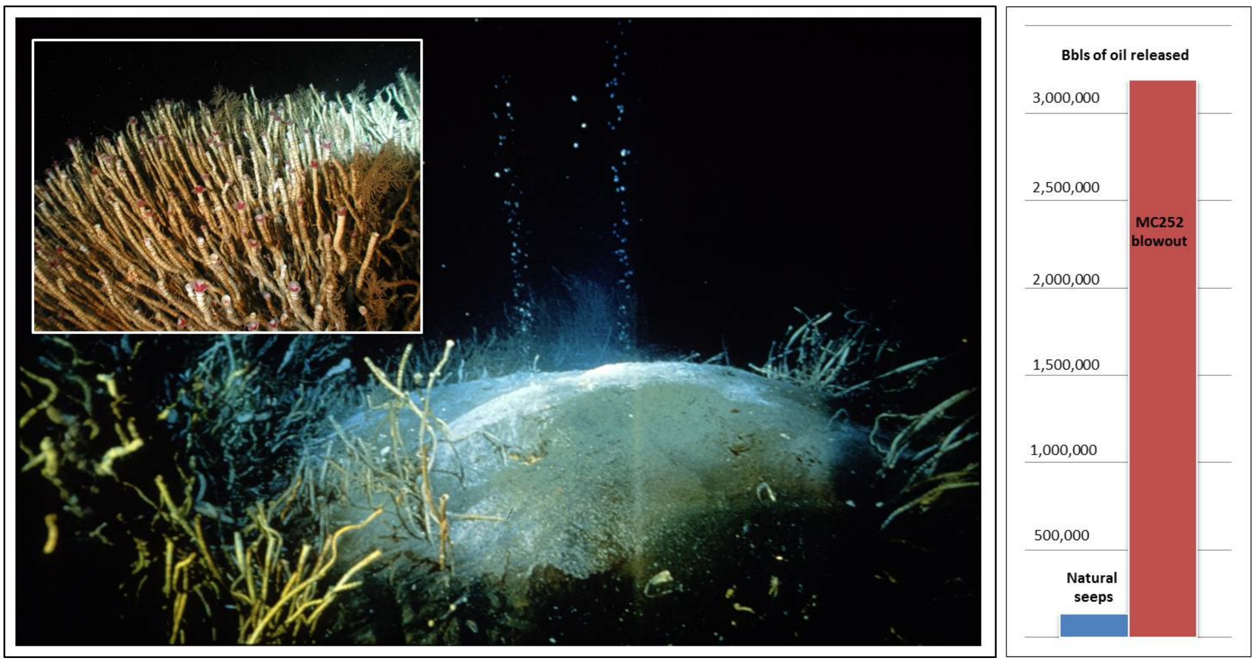
The third category of sites included deep-sea channels or unique bathymetry that, combined with currents, might accumulate oil and other contaminants in seafloor depressions. Another example from the nearshore is the surf zone, which is exposed to winds and crashing waves that could drive the oil and other contaminants into the shallow benthic sediments.

As part of this conceptual model, the Trustees assumed that less oil would reach the benthos of the continental shelf over which the floating oil slick was spreading. This assumption of lower likelihood of spill-related exposures and injuries to shelf habitats and benthic communities was based on two expectations: 1) travel across long distances of sea surface would potentially dilute and reduce surface oil concentrations over a broad shelf area, and 2) offshore waves would have less vertical force driving oil to the deep benthos than exists nearshore in turbulent surf zones. As discussed below in Section 4.5.4, Injury Determination, this assumption was generally shown to be valid, though the Trustees did identify injury to mesophotic reefs along the continental shelf edge and researchers working independently of the NRDA identified additional adverse effects on some shelf resources.

4.5.2.2 Potential Contribution of Natural Seeps

The northern Gulf of Mexico has natural seeps scattered across the sea floor, which contribute hydrocarbons into northern Gulf of Mexico waters and specifically to benthic marine habitats. Seeps are most abundant and most prolific in the central and western regions of the northern Gulf of Mexico, generally to the west of the location of the DWH oil spill (Garcia-Pineda et al. 2014). Nevertheless, as part of developing the approach for benthic assessment, the Trustees took special steps, including the use of forensic chemical techniques, to account for potential baseline contributions of oil from seeps.

Published information related to seeps in the northern Gulf of Mexico (Garcia-Pineda et al. 2014; MacDonald 2011; MacDonald et al. 2002) clearly shows that the total volume of oil released from all known natural seeps in the northern Gulf of Mexico is only a small fraction of the total DWH oil released during a comparable period of time (Figure 4.5-2). Further, the benthic footprint of impact from any natural seep is very limited, because most of the seep oil is weathered and rises to the ocean surface in droplets when it releases from the sea floor (MacDonald et al. 2002; Sassen et al. 1994). In contrast, the depth and physics of the DWH spill combined with the use of dispersants resulted in the distribution of spilled oil throughout the water column in some locations. Consequently, the Trustees determined that natural seeps were not a significant factor in the fouling and degradation of benthic habitats that were documented from the spill.



Source: *Ian MacDonald.*

Figure 4.5-2. Natural hydrocarbon seep and associated community: (a) tube worms; (b) hydrocarbon bubbles (which can include liquid and/or gas) rising from a hydrate mound; (c) oil volume released by all seeps in the northern Gulf of Mexico was approximately 138,000 barrels, compared with 3.19 million barrels over the 3-month period of the DWH spill.

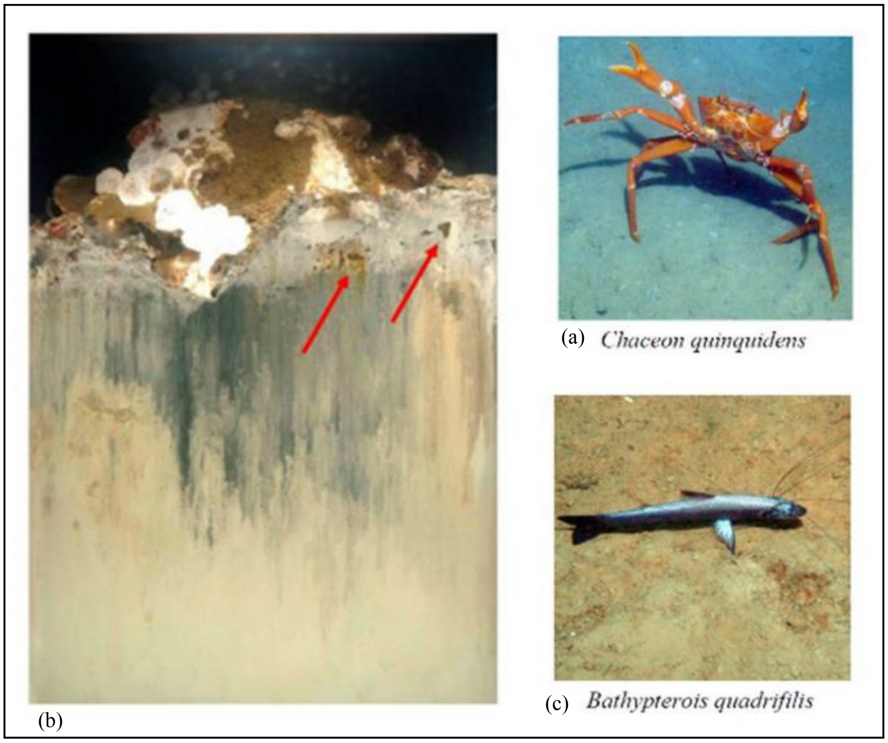
4.5.2.3 Studies to Support the Assessment

The Trustees relied on many sources of information to confirm exposure pathways and investigate potential injuries to seafloor habitats and resident animals and microbes. This included results from spill response activities; numerous targeted NRDA studies; research by academics, nongovernmental organizations, and industry; and studies directed independently by BP.

The intent of the DWH NRDA for benthic resources was to assess injuries to these resources and/or loss of ecological services provided by these communities caused by any aspect of the spill and response to the spill. The Trustees therefore considered the direct adverse effects of the spilled oil as well as any indirect injuries resulting from the response to the oil spill (see Chapter 2, Incident Description, for a more detailed description of the incident and response actions).

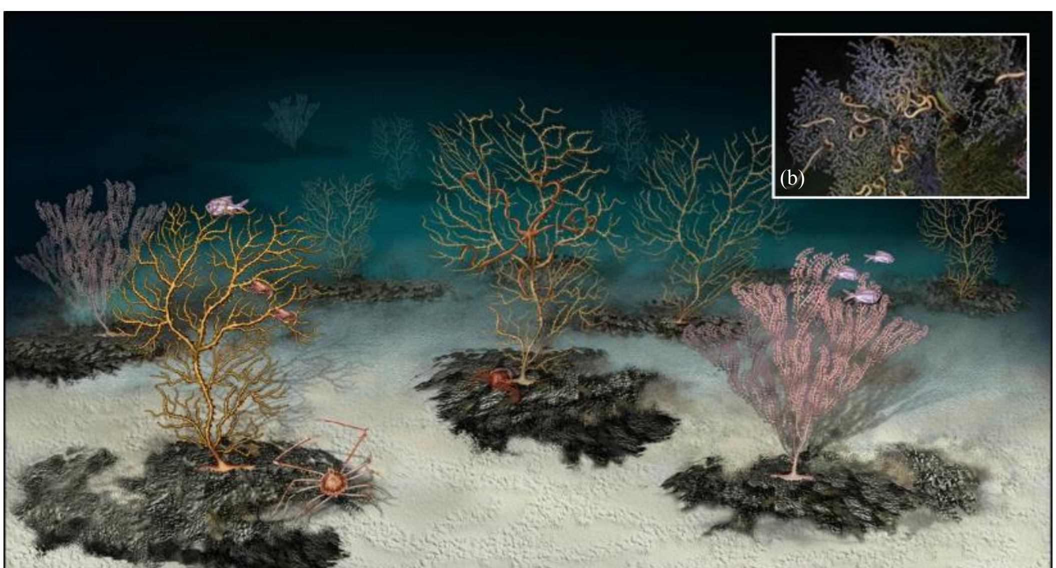
Assessment work specifically focused on investigations of soft sediment (Figure 4.5-3) and hard-bottom communities (Figure 4.5-4), including areas of known biodiversity—particularly the mesophotic reefs in the Pinnacles region along the continental shelf edge (Figure 4.5-5) in the northern Gulf of Mexico. Studies in the soft-bottom habitat targeted communities of animals and microbes living in and on the sediments. Studies in the hard-bottom communities focused on soft corals, and to a limited extent also evaluated potential impacts to other animals such as crabs, brittle stars, urchins, and sea cucumbers. As noted above, many mobile animals such as fish, crabs, and sea cucumbers move back and forth between the soft- and hard-bottom habitats.

The Trustees also evaluated possible impacts to shallower habitats and communities moving up the continental slope, onto the shelf, and into the shallow nearshore benthos seaward of the barrier islands (see Section 4.6 for the assessment of nearshore resources, including benthic resources landward of the barrier islands). Some of these shallower communities, in particular coral reefs in the mesophotic zone along the continental slope, were beneath documented surface slicks for months or underneath areas of dispersant spraying or burning of slicks. In deeper waters, benthic habitat was known to be beneath or in the direct path of subsurface plumes of dispersed oil or exposed to anoxic water, drilling mud, or other debris related to the DWH spill, and/or beneath documented surface slicks and areas of dispersant spraying and burning of slicks.



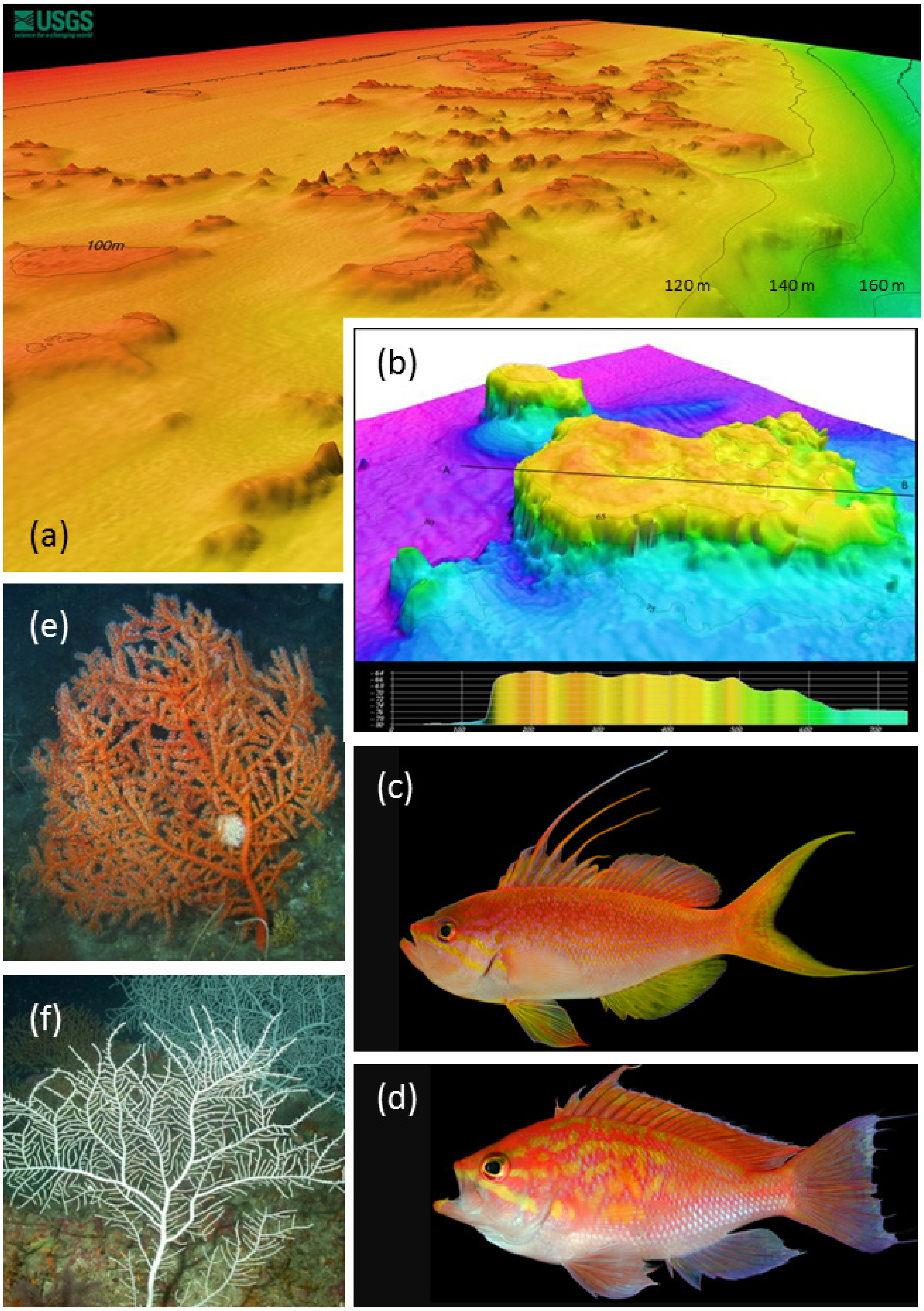
Source: *(a) Benfield (2014); (b) Germano and Associates Inc. et al. (2012); (c) Benfield (2014).*

Figure 4.5-3. (a) Soft-bottom sediment and a red crab (*Chaceon quinqudens*); (b) Sediment Profile Image (SPI) of sediment in the close vicinity of the Macondo wellhead (Station 000-200) showing various deposition layers, including "non-soluble liquid inclusions trapped within organically enriched surface depositional layer (red arrows)"; (c) soft-bottom sediment and the deep-sea tripod fish *Bathypterois quadrifilis*.



Source: *(a) Kate Sweeney for NOAA; (b) Charles Fisher.*

Figure 4.5-4. (a) An artist's depiction of deep-sea hardground coral habitat and community with (b) photo of healthy *Paramuricea sp*. corals and associated biological organisms, including brittle stars.



***Source:** USGS.*

Figure 4.5-5. Pinnacles, mesophotic reef community characteristics, clockwise from top. (a) Side-scan image of Pinnacles with (b) inset side-scan image of Roughtongue Reef. Dominant resident planktivorous fish on the Pinnacles reefs: (c) Roughtongue Bass (*Pronotogrammus martinicensis*) and (d) Red Barbier (*Hemanthias vivanus*). (e) *Swiftia sp*. mesophotic reef coral. (f) *Hypnogorgia sp*. mesophotic reef coral.

4.5.2.4 Tools Available and Technical Considerations

Knowing that DWH oil was released at the sea floor, transported throughout the northern Gulf of Mexico, and had a multitude of possible pathways to reach benthic resources, the Trustees assessed exposure and injury to benthic resources using a variety of field data, chemical analyses, laboratory toxicity evaluations, video analyses, biological analyses, statistical analytical techniques, and comparisons to published results from scientific literature.

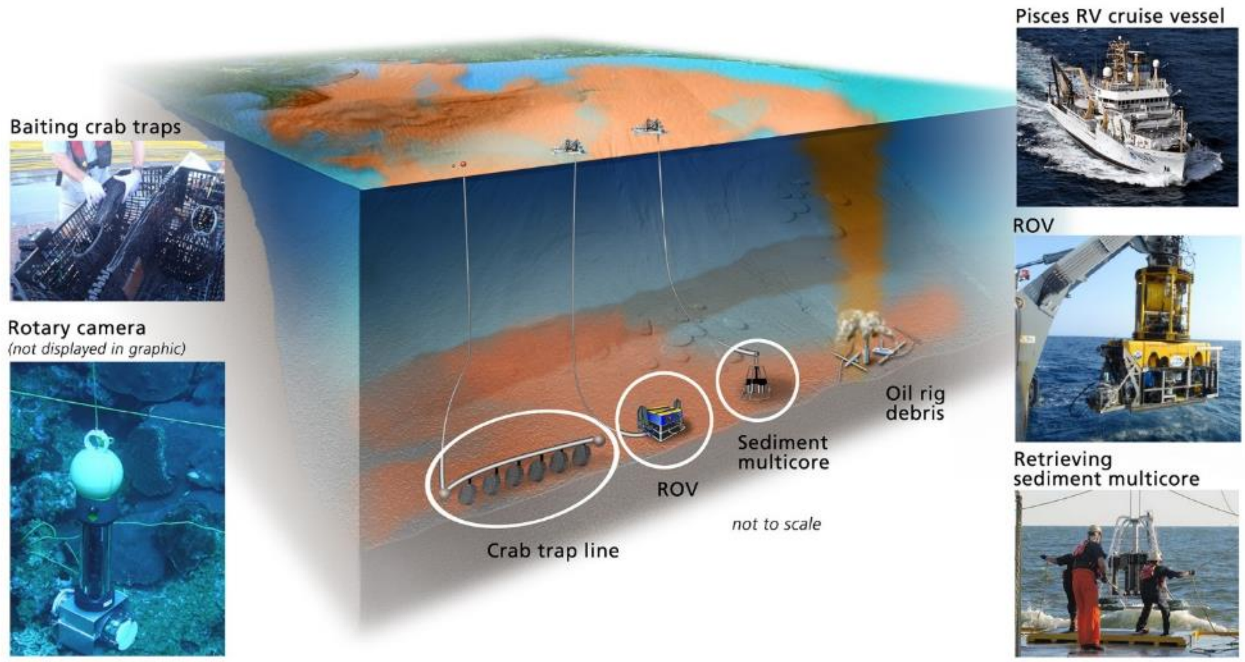
Challenging Conditions Demand Patience, Planning, Persistence, and Resources

"I imagined trying to understand cloud-shrouded New York City from an aircraft flying high above, towing a net through the streets, snaring a taxi, a few pedestrians, some bushes, a piece of building, shards of glass, a dog or pigeon or two. From high in the sky, what could I discover about human society, our music, art, sense of humor or connections that make our civilization function?"

Sylvia Earle (2014)—on the challenges scientists face when working from the deck of a rolling ship and trying to learn about life in the deep sea below.

The depths at which oil was released and dispersed from the DWH spill meant that natural resources located in the deep ocean had to be assessed during a series of offshore cruises using specialized equipment, such as autonomous underwater vehicles (AUVs), remotely operated vehicles (ROVs), towed cameras, and remote coring devices. The equipment was often deployed to extreme depths (Figure 4.5-6). The extreme depths necessarily limited the amount of time and effort that could be expended to investigate potential spill-related impacts. For example, deployment and retrieval of a sediment corer to sample sediment from a mile below the sea surface takes approximately 1 hour. In some cases, this action needed to be repeated multiple times during field sampling cruises, either because replicate samples were needed or an incomplete sample was collected (Montagna & Cooksey 2011). The Trustees overcame a variety of logistical challenges to collect an unprecedented amount of information related to environmental impacts stemming from the oil spill. Nevertheless, as is detailed in the Injury Determination and Injury Quantification sections, areas of uncertain impacts remain.

At the outset of the spill, debris, including wreckage of the DWH drilling rig and portions of riser pipe, all came to rest on the sea floor in and around the wellhead. This debris fell within an exclusion zone where cruise efforts were not allowed to target sampling. However, information gathered from cruises conducted as part of the response and NRDA and other independent investigations have allowed the Trustees to assess the adverse impacts of the spill on the deep benthos over the past several years.



Source: Kate Sweeney for NOAA (illustration); NOAA (cruise vessel); Jim Payne (ROV); Ian Hartwell (sediment multicore); Ian MacDonald (rotary camera); Harriet Perry (crab traps).

Figure 4.5-6. Types of sampling tools used as part of the deep benthic injury assessment, including crab trap line, ROV, sediment multicore, and rotary camera.

4.5.2.5 Cruises

Beginning in 2010, immediately after the blowout of the Macondo well, offshore cruises were conducted as part of the response to assess oil exposure and injury to natural resources located at depths in and around the wellhead and throughout the northern Gulf of Mexico. During the rest of 2010 and during the 2011 and 2014 field seasons, the Trustees conducted additional offshore cruises as part of the NRDA; academic researchers conducted numerous independent investigative cruises (Table 4.5-1).

Soon after the spill started, the Trustees leveraged data collection efforts on several cruises planned prior to the oil spill, on which participants agreed to cooperate with the Trustees and collect samples and data for use in the NRDA. Given the time and effort involved in planning and implementing an offshore research cruise, and because many vessels located in the Gulf were already being used in response operations, these cooperative cruises allowed for the rapid collection of ephemeral data for the NRDA. Early findings and impressions from these cruises subsequently allowed the Trustees to narrow their focus on certain resources and habitats.

Table 4.5-1. Cruises instrumental in providing data to support the benthic assessments. In addition to the cruises listed below, numerous academic cruises were conducted outside of the purview of the Response and NRDA. The Trustees, in some cases, also incorporated information from independent academic cruises in the NRDA.

Sampling Effort	NRDA or Non-NRDA	Dates	Primary Activities/Objectives
NRDA Tier 1 SPMD Detection of Hydrocarbons in Water Column Immediately over NEGOM Shelf-Edge Pinnacle Reefs- Small vessel	NRDA	June 2010	One at-sea day to deploy hydrocarbon sampling equipment for subsequent retrieval during July Tier-1 deep coral impacts cruise.
NRDA Tier 1 for Deepwater Communities—RV *Nancy Foster*	NRDA	July 13–August 8, 2010	Assessment of mesophotic reef and deep-sea coral habitat over two cruise legs. Data collection included photography and videography and limited environmental sample collection.
NOAA Continental Shelf Benthic Study—RV *Nancy Foster*	Non-NRDA	August 12–August 22, 2010	Cruise to assess benthic health, planned prior to the spill. Included collection of sediment samples for various contaminant analyses, including hydrocarbons.
Sediment Sampling Response Cruise—RV *Gyre*	Non-NRDA	September 16–October 27, 2010	Assessed the magnitude and extent of oil residues in sediment, and possible biological impacts in the Gulf of Mexico following the DWH spill.
Sediment Sampling Response Cruise—RV *Ocean Veritas*	Non-NRDA	September 19–October 9, 2010	Assessed the magnitude and extent of oil residues in sediment, and possible biological impacts in the Gulf of Mexico following the DWH spill.
Lophelia II Project to research deep water coral communities—RV *Ron Brown*	NRDA/Non-NRDA	October 14–November 4, 2010	Independent cruise planned prior to the spill, during which researchers agreed to collect environmental samples for the NRDA. Sampling and photography also targeted deep-sea hardground communities around the wellhead.
Reconnaissance Survey of Hard-Ground Megafauna Communities in the Vicinity of the *Deepwater Horizon* Spill Site—RV *Gyre*	NRDA	October 25–November 5, 2010	Reconnaissance cruise to identify potential hardground communities using a drift camera.
NSF Rapid Project—RV *Atlantis*	NRDA/Non-NRDA	December 6–December 12, 2010	Independent cruise funded by NSF in response to the spill, during which researchers agreed to collect environmental samples for the NRDA. Photography and videography of deep-sea hardground habitats.

Sampling Effort	NRDA or Non-NRDA	Dates	Primary Activities/Objectives
Time Lapse Camera and Sediment Trap Retrieval and Redeployment Plan—MV *HOS Sweetwater*	NRDA	March 9–March 13, 2011	Short, dedicated cruise to retrieve and redeploy a time lapse camera and sediment trap located at a deep-sea hardground habitat.
Offshore and Deepwater Soft Bottom Sediment and Benthic Community Structure Survey, Sediment Profile Imaging—MV *Sarah Bordelon*	NRDA	April 7–April 23, 2011	Cruise dedicated to the collection of a series of sediment profile images extending in all directions at various distances from the wellhead.
AUV Reconnaissance Survey II of Hard-Ground Megafaunal Communities in the Vicinity of the *Deepwater Horizon* Spill Site—RV *MacArthur*	NRDA	April 20–May 22, 2011	Cruise to deploy an AUV to identify potential hardground communities in the vicinity of the wellhead.
Deepwater Sediment Sampling to Assess Potential Post-Spill Benthic Impacts from the *Deepwater Horizon* Oil Spill—MV *Sarah Bordelon*	NRDA	May 23–June 16 2011	Sediment sampling cruise to assess potential spill-related impacts on deep water sediments and benthic infauna.
Deepwater Megafauna Cruise 1—MV *HOS Sweetwater*	NRDA	June 8–June 22, 2011	Cruise to collect data to quantify biodiversity, distribution, and abundance of benthic and demersal megafauna at selected locations around wellhead.
ROV Sediment and Bottom-Water Sampling Cruise—MV *HOS Sweetwater*	NRDA	July 14–August 7, 2011; August 22–September 1, 2011; September 10–25, 2011	Collected a variety of environmental samples at stations in the vicinity of the Macondo well site and areas to the southwest along transects of potential exposure.
Assessment of Impacts from the *Deepwater Horizon* Oil Spill on Red Crabs—RV *Pisces*	NRDA	July 27–Aug 7, 2011; Aug 8–Aug 17, 2011	Collected and documented potential exposure of red crabs to spill-related contaminants, collected tissue samples to document potential reproductive and histological effects of exposure to spill-related contaminants, and collected information on catch per unit effort at selected study locations.
Deepwater Megafauna Leg 2—MV *HOS Sweetwater*	NRDA	August 10–August 22, 2011	Follow-up cruise to collect data to quantify biodiversity, distribution, and abundance of benthic and demersal megafauna at selected locations around wellhead.

Sampling Effort	NRDA or Non-NRDA	Dates	Primary Activities/Objectives
Mesophotic Reef Follow-Up Cruise—MV *Holiday Chouest*	NRDA	September 15–30, 2011	Cruise to collect photography, videography, and some limited environmental samples, and to deploy permanent markers for re-survey of mesophotic reefs along the continental shelf edge. This was a follow-up cruise to the portion of the NRDA Tier 1 cruise targeting mesophotic reefs.
Offshore and Deepwater Soft Bottom Sediment and Benthic Community Structure Survey—Follow-up Cruise—Sediment Profile Imaging—MV *Sarah Bordelon*	NRDA	September 22–October 27, 2011	Follow-up cruise dedicated to the collection of a series of sediment profile images extending in all directions at various distances from the wellhead, targeting sampling locations left un-photographed from the first cruise.
Deepwater ROV Sampling to Assess Potential Impacts to Hardbottom Coral Communities and Associates from the *Deepwater Horizon* Oil Spill—MV *Holiday Chouest*	NRDA	October 2011	Cruise to collect photography, videography, and some limited environmental samples from deep-sea hardground communities. This was a follow-up cruise to the portion of the NRDA Tier 1 AUV cruise targeting deep-sea hardground communities.
Assessment of Impacts from the *Deepwater Horizon* Oil Spill on Red Crabs—RV *Pisces*	NRDA	August 22–Sep 12, 2014	Follow-up cruise to the 2011 cruise studying red crabs.
Deepwater Sediment Sampling to Assess Potential Post-Spill Benthic Impacts from the *Deepwater Horizon* Oil Spill—MV *Irish*	NRDA	May 28–June 11, 2014; June 14–June 26, 2014	Follow-up cruise to the 2011 cruise collecting soft-bottom sediment samples.
Mesophotic Reef Follow-Up Cruise—RV *Walton Smith*	NRDA	June 22–July 13, 2014	Follow-up cruise to the 2011 cruise studying mesophotic reefs, including re-survey of marked corals in 2011.

In many instances, the Trustees relied on photographic and video information obtained using the tools and techniques described above, along with standardized sampling along transects or repeat sampling at specific locations. This provided a successful way to deal with many of the logistical challenges of working at great depths and allowed for detailed scrutiny of images after completing work in the field. These approaches, when used for repetitive sampling over several years, allowed the Trustee scientists to compare photographic and video images and assess obvious and overt signs of resource injury such as mortality, absence of biota, and shifts in biological communities over time.

When possible, the Trustees used statistical approaches designed to identify changes in the condition of resources understood to be affected by the spill (e.g., the before-after control-impact comparisons design for sampling and subsequent data analysis, as detailed in Section 4.1, Approach to the Injury Assessment). In some other instances, the Trustees used sampling designs that followed spatial gradients away from the release at the blown-out well to look for spatial and temporal trends correlating with the presence of spill-related contaminants. The Trustees also made use of spatially explicit statistical techniques, including but not limited to spatial interpolation combined with principal components analyses to identify impacts from the spill and tie them to specific geographical locations.

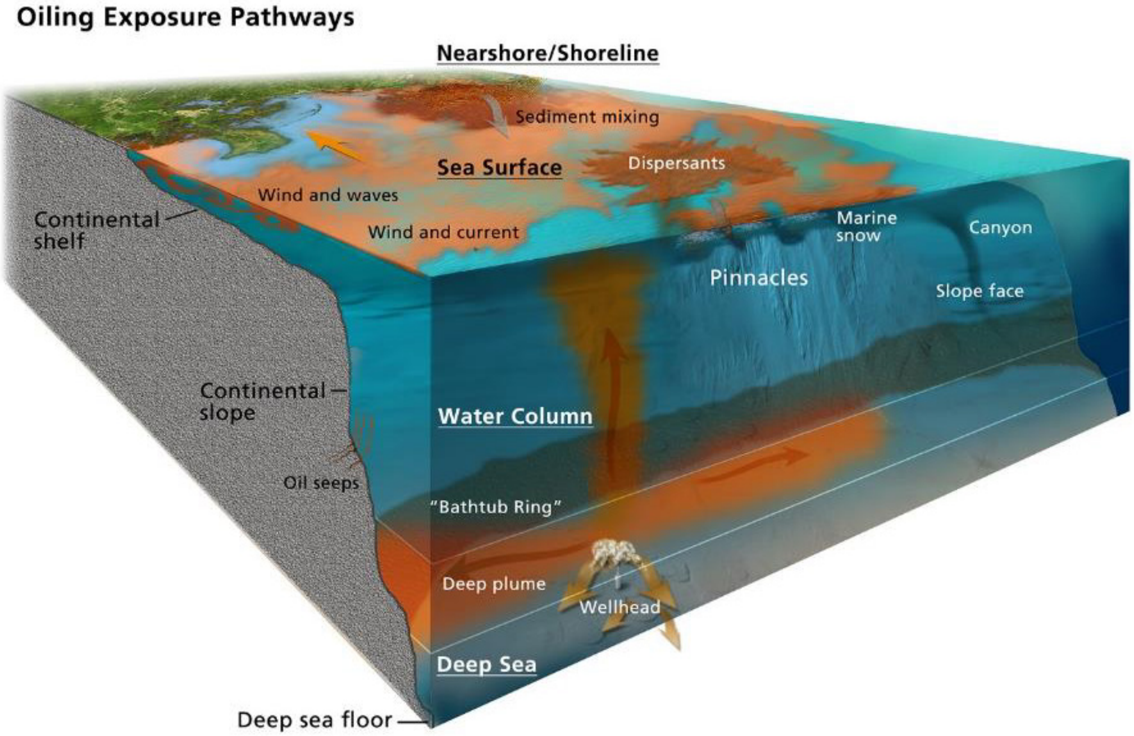
4.5.3 Exposure

Key Points

- Exposure of benthic resources to oil and other spill-related constituents occurred via four primary pathways: underwater plumes, contaminated marine snow, direct contact with contaminated sediments, and uptake of contaminated food.
- Benthic resources were exposed across a large swath of the northern Gulf of Mexico, though exposure decreased and became patchy with increasing distance from the wellhead.
  - Benthic resources were confirmed to have been contaminated with DWH oil at distances of more than 35 miles (57 kilometers) from the wellhead.
  - Patchy exposure likely occurred below where DWH oil spread across the sea surface or in the deep plume.

As discussed in Section 4.2, oil was released at a depth of approximately 1,500 meters, which resulted in the dispersion of oil directly into the water column. Further, at various times throughout the 87 days that oil was actively being released, dispersants were added to the oil streaming from the riser pipe or directly to floating oil on the sea surface. This effectively distributed the oil to a greater degree into the water column (see Section 4.2, Natural Resource Exposure, and Section 4.4, Water Column). Subsequent exposure of benthic resources to spilled contaminants occurred through one or more of four primary pathways (Figure 4.5-7):

1. Direct contact with underwater plumes of DWH oil, dissolved hydrocarbons, and dispersant that persisted for months at various depths in the water column and near the deep-sea floor.
2. Contact with marine snow—a naturally occurring mix of organic and inorganic detritus—that was contaminated with DWH oil and dispersants before being deposited on the sea floor.
3. Contact with contaminated sediments (contaminated either directly through contact with oil and dispersant droplets or contaminated marine snow).
4. Consumption of contaminated prey/food.



Source: *Kate Sweeney for NOAA.*

Figure 4.5-7. Exposure pathways to benthos.

4.5.3.1 Underwater Plumes

Application of dispersants, at depth and at the sea surface, was intended to, and did, disperse oil into the water column. Dispersed DWH oil was documented in deep subsurface plumes and tracked to a distance greater than 249 miles (400 kilometers) in the water column along a pathway extending southwest of the release point. Hydrocarbons were also detected at shallower depths near the sea surface (Stout & Litman 2015). In some cases, because of topography, plumes came near the benthos and deposited oil and dispersants to the sea floor. This has been referred to in the peer-reviewed literature as a "bathtub ring" of oil left behind on the sediment in areas where the plume moved (Figure 4.5-7). In this manner, sediment and biological organisms living on or near the sea floor in these areas were exposed to the contaminated water column (Valentine et al. 2014). Readers are also referred to Section 4.4, which discusses injuries to resources in the water column.

4.5.3.2 Contaminated Marine Snow

Marine snow is a natural phenomenon that is ubiquitous in all oceans. It consists of aggregations of marine particles (including bacteria, the bodies of small plants and animals, fecal pellets, clay minerals, and other natural materials) that sink to the sea floor (Silver & Allredge 1981).

Large amounts of marine snow were observed following the DWH incident. Specifically, stringy "floc" associated with surface slicks were reported at the sea surface (Passow et al. 2012). "Floc" covering a vast area of the sea floor was also reported, particularly in areas where dispersants were applied

(Passow 2014) in areas of known heavy oiling, and where sediment from the Mississippi River may have been distributed along with oil from the spill (Brooks et al. 2015; Fu et al. 2014; Hartwell 2015) (Figure 4.5-8). The large aggregations and character of marine snow observed following the spill, and the increased depositional pulse to benthic sediments, was unlike anything previously observed in many parts of the northern Gulf of Mexico. Brooks et al. (2015) reported that sediments below the layer of excess spill-related floc are generally homogeneous and contain no evidence of previous similar depositional events. This suggests that either what occurred with respect to marine snow following the spill was unique, or that deposits resulting from such events have not been preserved in the sedimentary record.

Source: *Jeff Baguley.*

Figure 4.5-8. Photos of sediment cores taken aboard the R/V *Ocean Veritas* response cruise. (a) A representative pre-spill sediment core with compacted sediments and lacking floc. (b) A sediment core showing the presence of an overlaying, loosely aggregated light-brown flocculent layer. Although sedimentation of marine snow is understood to be a natural phenomenon, data suggest that a large sedimentary event was associated with the oil spill, and, furthermore, this mass transport of floc resulted in transport of oil to the benthos.

Increased amounts of marine snow and rapid sinking also led to entrainment of oil by the marine snow and subsequent deposition of oil to the sea floor (Passow et al. 2012; Stout & German 2015; Stout & Passow 2015). The oil and dispersants contaminated, and thereby adversely affected, the vital pathway through which food, sediments, and other organic debris are transported downward to support benthic marine life. Chemical analysis of marine snow collected in settling traps from locations southwest and northeast of the blown-out well confirmed contamination with DWH oil (Stout & German 2015; Stout & Passow 2015) (Section 4.2, Natural Resource Exposure). Such results show that benthic resources were exposed to contaminated marine snow at least up to, and likely exceeding, 35 miles (57 kilometers) away from the wellhead.

The presence of floc on the sea floor corresponds well to areas beneath surface slicks and where dispersants were applied at the water surface (Figure 4.5-9). Chemical analysis also confirmed the presence of DWH oil in the floc on soft corals approximately 13 kilometers from the wellhead in the westerly direction and dispersant residues approximately 23 kilometers from the wellhead in the southeasterly direction (H.K. White et al. 2012; White et al. 2014). However, many floc samples from northeast of the wellhead and through Desoto Canyon did not contain significant quantities of petroleum hydrocarbons, nor did the Trustees confirm DWH oil fingerprints in many of these floc samples. The Trustees therefore documented contaminated marine snow at distances up to 35 miles (57 kilometers) of the wellhead, but such contamination was understood to be patchy in nature (Stout & German 2015).

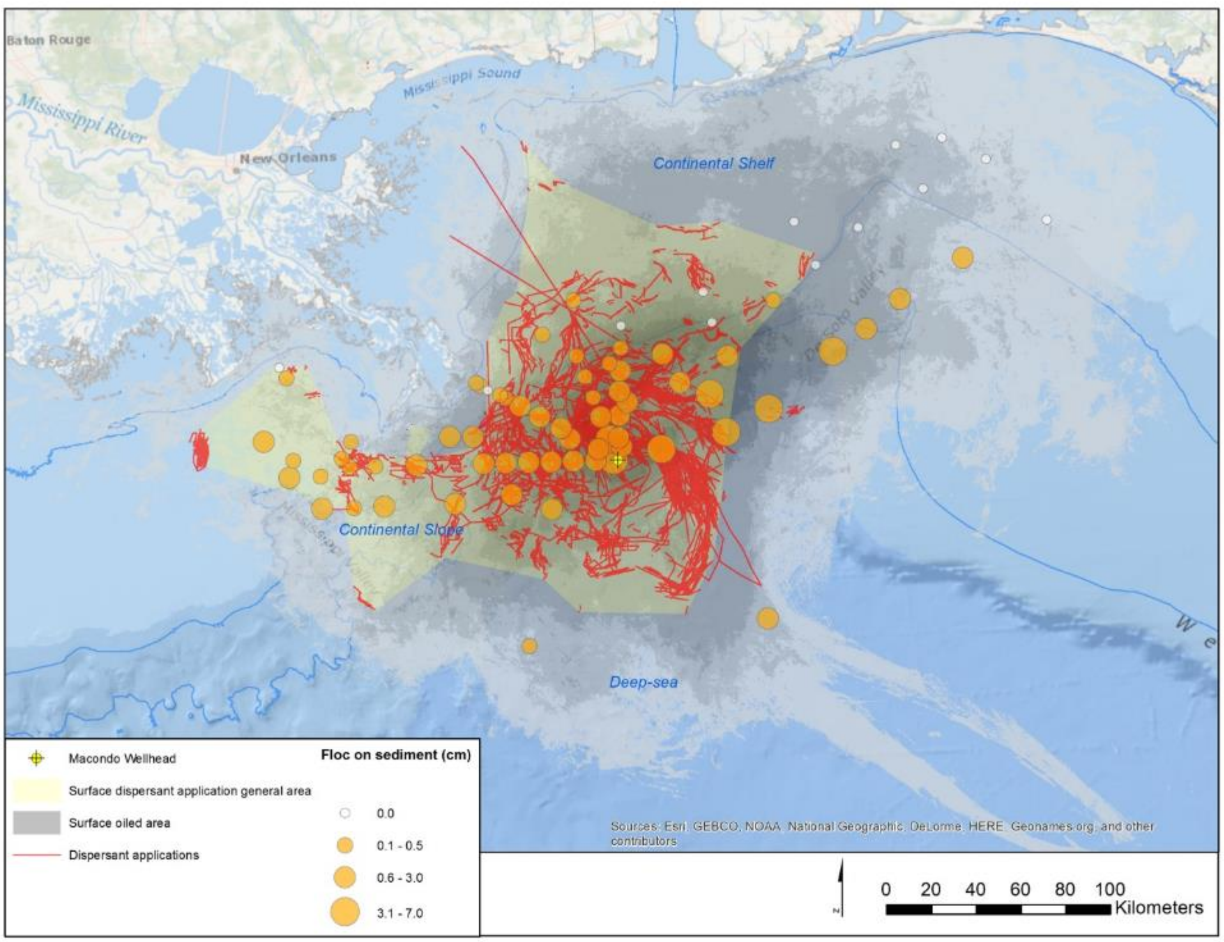


Figure 4.5-9. Map overlaying surface dispersant application area, surface oiled area, and floc thickness (cm) found on the deep-sea sediments. Larger quantities of floc were generally observed on the sea floor beneath areas experiencing persistent surface oil and the application of dispersants (which were applied in areas of heavy surface oiling). Depth of floc also generally decreased with increasing distance from the wellhead.

Additionally, marine snow interacted with the subsurface plume, which extended over 400 kilometers to the southwest of the wellhead, and it likely increased the oily floc footprint in the deep-sea benthos. This is described in Section 4.2, Natural Resource Exposure.

Although the Trustees documented the settling of contaminated marine snow and increased flocculation layers extending up the continental slope and onto the shelf, the Trustees did not confirm extensive oil contamination of continental shelf sediments. Specifically, NOAA conducted a sediment sampling cruise that was planned prior to the spill, but was implemented after the spill in the fall of 2010. NOAA sampled multiple locations across the continental shelf, and results from this effort are published in a NOAA technical memorandum (Cooksey et al. 2014). The scientists did not observe toxic concentrations of PAHs in locations where they sampled on the continental shelf in 2010, roughly 3 months after the spill began.

Mesophotic reefs, however, were exposed to oil and likely dispersants. For 30 or more days during the DWH oil spill, petroleum slicks were documented directly above Roughtongue and Alabama Alps Reefs, and aerial dispersants were used nearby. Waterborne dissolved hydrocarbon sampling devices (semi-permeable membrane devices, or SPMDs) revealed elevated PAHs and “fingerprints” consistent with exposures from a broad-boiling petroleum, such as crude oil (Stout & Litman 2015). Summer 2010 deployments consisted of four SPMD devices, and each had comparable fingerprints to one another. Furthermore, “fingerprints” from the summer deployments appeared slightly “fresher” (less weathered) than the four SPMD “fingerprints” obtained from the second SPMD deployments in the fall of 2010 (Stout & Litman 2015). These findings contrast with a relative lack of petroleum hydrocarbons sampled by SPMDs deployed outside of the influence of surface oil off Cedar Key, Dry Tortugas, Florida Bay, and Biscayne Bay from May through August 2010. Furthermore, Roughtongue Reef lies just upslope of the continental slope location where the Trustees documented deposition of DWH oil-contaminated marine snow, indicating that the reefs may have been exposed to both dissolved and particulate oil.

The northern and eastern portions of the Gulf of Mexico also have shallow-water reefs scattered across the continental shelf from approximately 15 kilometers offshore to the shelf edge. In the north-central portion of the Gulf of Mexico, south of Alabama and Florida, the reefs are primarily composed of sandstone and limestone with extensive covering by sponges and supporting rich communities of fishes and other animals. Farther to the south, near the southern tip of Florida, reef-forming (hermatypic) corals grow and dominate many nearshore shallow reef habitats. The Trustees searched for, but did not confirm, a pathway of oil and dispersants leading to shallow-water coral-reef habitats, and exposure to spill-related contaminants was not demonstrated (Goodwin 2015). Consequently, the Trustees did not pursue assessment activities to characterize exposure or document injuries to shallow-water coral reef communities as a result of the spill.

4.5.3.3 Contaminated Sediment

Benthic infauna and epifauna (animals living in and on top of the sea floor, respectively) exposures to contaminants resulted from these animals’ and microbes’ close contact and interaction with spill-affected bottom sediments. Oil and dispersant came to be located in marine sediments either through direct contact of oil droplets, dispersed oil, or dispersant alone with the sediment as the chemical constituents settled out of the underwater plumes, or through the deposition of contaminated marine snow or floc. Many benthic animals ingest sediment routinely as part of their normal feeding behavior and also as part of burrowing into and reworking sediments, a process known as bioturbation. Bioturbation resulted in some mixing of the contaminated surface materials and floc with underlying sediments, thereby spreading contaminants in the soft-bottom habitat. Contaminated sediment, especially when loosely aggregated, also has the propensity to be remobilized and dispersed through the influence of erosion by bottom currents (Germano and Associates Inc. et al. 2012).

The area of the benthos within which DWH oil was affirmatively fingerprinted and quantified in sediment samples extends beyond 2,000 square kilometers. Benthic resources were certainly exposed over a significantly larger area. However, any exposure in this larger area would likely be patchy (Valentine et al. 2014). For example, as noted above, the NOAA sediment sampling program across the continental shelf did not reveal widespread PAH contamination in the sediment (Cooksey et al. 2014).

4.5.3.4 Contaminated Prey in Marine Food Webs

Benthic marine animals living on and in sediments, as well as animals such as corals living on rocky hard-bottom habitat, were collected and their tissues analyzed for DWH oil and petroleum hydrocarbons in general. A broad variety of animals, including deep-sea red crabs, sea urchins, corals, and sea cucumbers, were confirmed to be contaminated with DWH oil (Douglas & Liu 2015), and many of these animals are known to be consumed as food by other animals. DWH oil was measured in the tissues of marine animals at up to 57 kilometers of the wellhead (Douglas & Liu 2015). These animals themselves could have been exposed via consumption of contaminated food or sediments, or directly exposed to DWH oil.

4.5.4 Injury Determination

Key Points

- Assessing resources across the three depth regions of the assessment, the Trustees documented a variety of injuries to benthic resources primarily in two areas in the northern Gulf of Mexico: within a large area of deep-sea benthic habitat surrounding the wellhead, and along the edge of the continental shelf at the mesophotic Pinnacles reefs.
- The types of natural resource injuries documented in the deep-sea benthos included degradation of the physical and chemical quality of the sediment, smothering by debris and drilling mud, toxicity of sediment as measured using standard laboratory toxicity tests, adverse effects to the structure of infaunal and epifaunal communities, injuries to red crabs and deep-sea hardground coral colonies, and adverse shifts in microbial communities.
- Some reports of injuries to natural resources along the continental slope and shelf were identified in the peer-reviewed literature, but these injuries were not reported to be widespread. The exception was degradation of mesophotic reef habitat, as documented through observations of widespread injury to corals and a severe reduction in the abundance of site-attached planktivorous fish.

The Trustees identified three primary types of spill-related adverse effects, or types of injuries, to benthic resources stemming from the DWH oil spill: 1) contamination resulting in a chemical change and degradation of habitat quality and structure, 2) changes in resource and ecosystem health or function, and (3) mortality. These types of injuries were either:

- The direct result of exposure to spilled oil or other spill-related constituents such as dispersants or burn residues.
- Impacts from the wreckage itself.
- Impacts from drilling muds or other response-related activities.
- Effects related to burial and smothering.

Contamination and degradation of habitat and ecosystem quality occurred both physically and biologically. For example, sediments were contaminated with oil, dispersants, drilling muds, and other debris—all of which degraded the physical properties and quality of the habitat. Similarly, some of these contaminants, such as toxic PAHs, were taken up in tissues of animals exposed to the spill, so that the quality of food provided by these animals to higher trophic level organisms was degraded. Changes in resource or ecosystem health or functionality occurred to individuals, to colonies, and to communities. Examples include degradation of coral colonies by smothering from a coating of contaminated flocculent material, mortality through direct contact with oil compounds, and shifts in species dominance and diversity of benthic infauna and epifauna that affect overall functionality of the community. Finally, mortality was documented not only at the individual level, but also to groups of individuals, such as colonies of corals (Etnoyer et al. 2015; Fisher et al. 2014a; Fisher et al. 2015; Hsing et al. 2013; Silva et al. 2015; H.K. White et al. 2012), or populations of certain species of fish (Sulak & Dixon 2015) and invertebrates (Baguley et al. 2015; Montagna et al. 2013). In some instances, mortality was documented through shifts in abundances of animals that led to changes in community composition, which in turn affected the functionality of the community. Therefore, one type of loss at the level of the individual, if occurring to a significant degree and affecting many individuals, resulted in another loss at a higher level of biological functioning of the deep-sea communities.

4.5.4.1 Deep Benthos Injuries

The specific injuries documented by the Trustees in the deep benthos are described below.

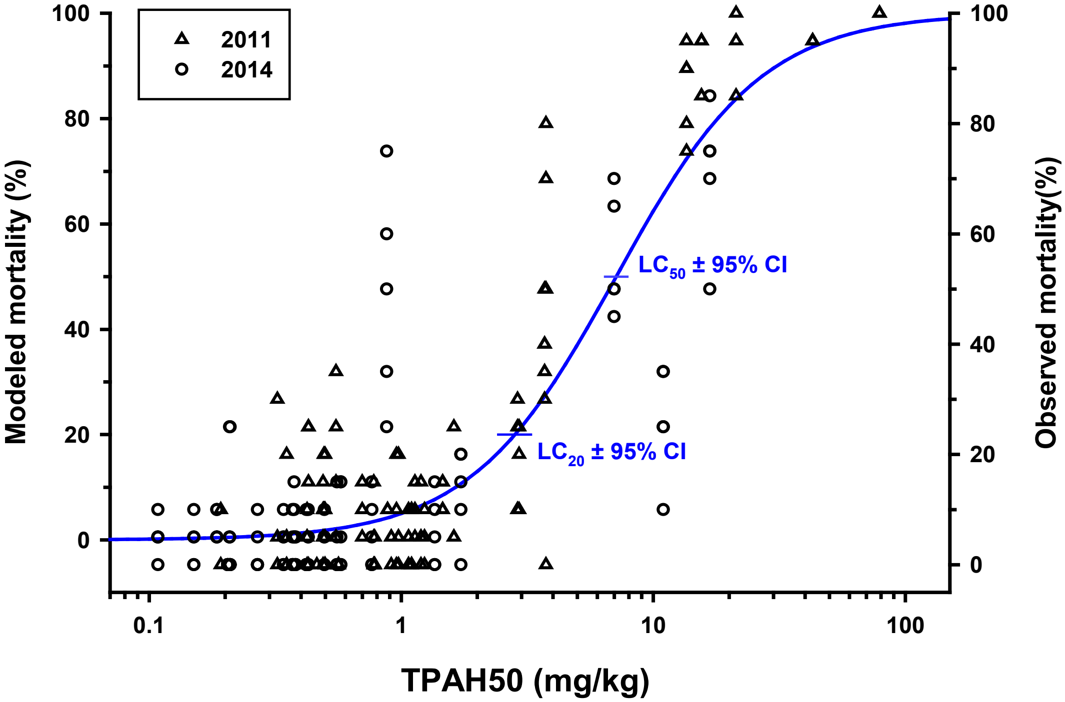
4.5.4.1.1 Smothering by Debris and Drilling Mud

Within the immediate vicinity of the ruptured wellhead, a variety of debris from the destroyed DWH drilling rig wreckage came to be located on the sea floor. This debris, along with layers of drilling muds used in the "top kill" effort, smothered any organisms living on or within the sediment that were unable to escape prior to the spill. Further, this material represents a potential continuing source of contamination to the sediment environment—both from apparent drops of persistent oil and from other contaminants, such as metals, that are present in drilling muds (Germano and Associates Inc. et al. 2012). While the structures themselves may provide some shelter to marine life, the contamination will continue to adversely affect the quality of the sediment environment and its ability to support a healthy and diverse sediment community.

4.5.4.1.2 Sediment Toxicity

Surface sediments from benthic core samples were analyzed for toxicity in standardized tests using the amphipod *Leptocheirus*. Sediments collected within approximately 2 kilometers of the wellhead were measured for TPAH50, and sediments exhibited toxicity to the amphipod *Leptocheirus* (Krasnec et al. 2015). The Trustees fit dose-response curves to the results to estimate LC20 (i.e., modeled concentrations of TPAH50 in sediment that are lethal to least 20 percent of the animals) and LC50 (i.e., modeled concentrations of TPAH50 that are lethal to least 50 percent of the animals) (Figure 4.5-10a) (Krasnec et al. 2015). The Trustees then identified locations from which deep-sea sediment samples were taken (and TPAH50 values measured) that had TPAH50 concentrations in excess of the LC20 and LC50 values (LC20=2.82 mg/kg and LC50=7.12 mg/kg TPAH50). An exceedance of these values indicates that if toxicity tests were run on such sediments, it is likely that they would be toxic. Benthic TPAH50 concentrations exceeded LC20 and LC50 values at locations more than 25 kilometers southwest of the wellhead and to lesser distances in other compass directions (Figure 4.5-10b). Although there was less toxicity observed generally in 2014 relative to 2011, toxicity persisted in 2014 at several comparably located 2011 locations, indicating persistence of toxicity at least 4 years after the spill (Krasnec et al. 2015).

a



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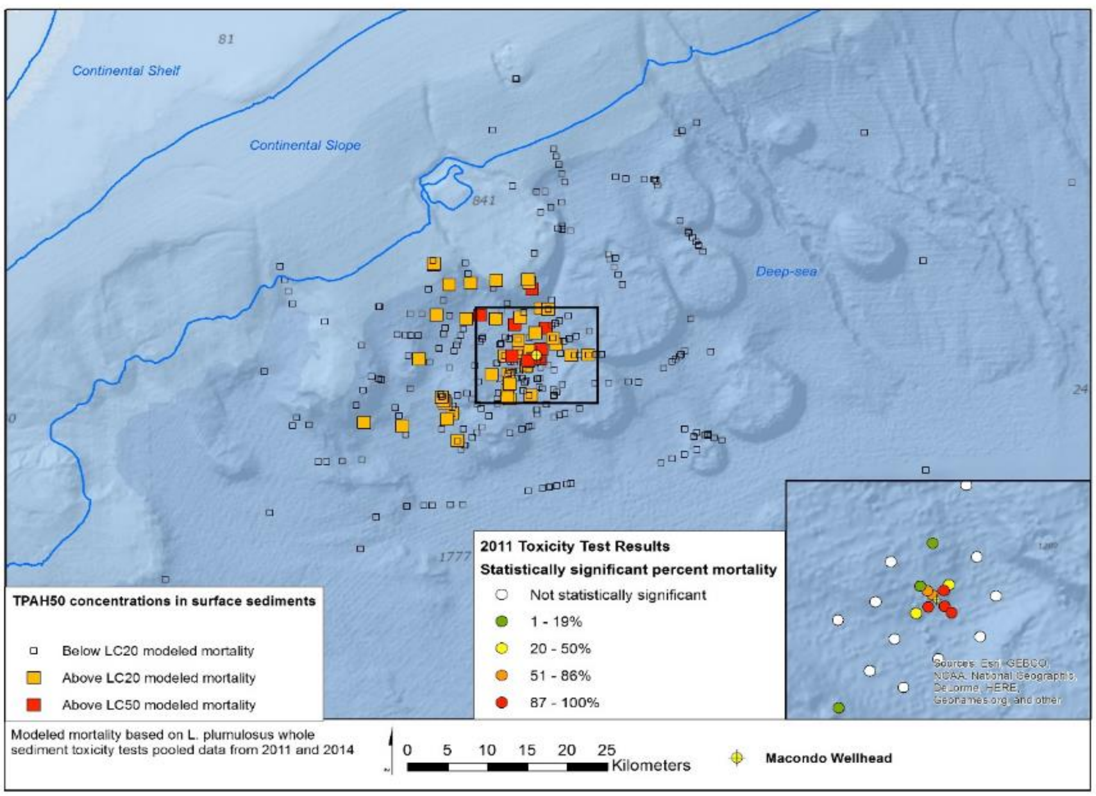


Figure 4.5-10. (a) Sediment toxicity results for deep-sea sediment samples taken in 2011 and 2014 with modeled sediment toxicity indicating LC20 (2.82 mg/kg TPAH50) and LC50 (7.12 mg/kg TPAH50) mortality based on TPAH50 values (Krasnec et al. 2015). (b) Map indicating surface TPAH50 concentrations that exceed LC20 and LC50 values for modeled mortality.

4.5.4.1.3 Adverse Effects to Deep-Sea Biological Community Structure

Injuries were documented to numerous small invertebrates such as worms, crustaceans, and mollusks that dwell in or on the bottom sediments (referred to generally as infauna or epifauna depending on their location either in or on the sediment) and play an important role in the deep-sea food web (Montagna et al. 2013). Changes in the abundances of individual species associated with spill-contaminated sediment were documented, and this shift in species composition resulted in a loss of species diversity (Montagna et al. 2013). Sediments within approximately 3 kilometers of the wellhead experienced a roughly 54 percent reduction in diversity of macrofauna (larger animals living in the sediments) and a 38 percent reduction to meiofauna (very small animals living in the sediments). Between 3 and 15 kilometers of the wellhead, the Trustees documented roughly a 5 percent reduction in diversity of macrofauna and a 19 percent reduction to meiofauna diversity. Beyond 15 kilometers from the wellhead, the diversity of benthic faunal resources was unable to be discerned as being different from background populations across the wider northern Gulf of Mexico (Figure 4.5-11, Table 4.5-2; Montagna et al. (2013). These areas of diversity reductions and related alterations were generally supported by a more recent, closer evaluation of meiofauna data reported by Baguley et al. (2015). These authors reported a significant increase in the nematode to copepod ratio (N:C), indicative of injury to meiofauna.

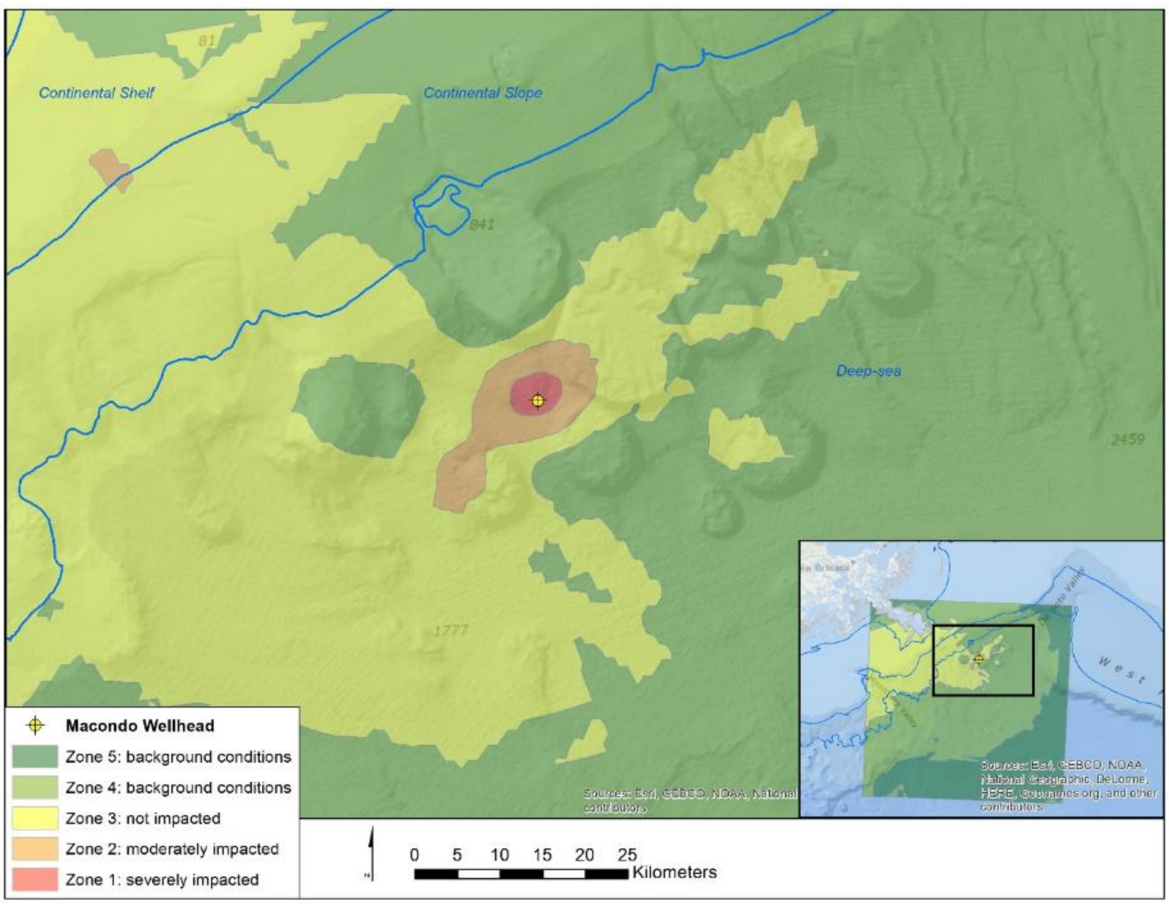


Figure 4.5-11. Footprint of benthic injury to sediment-dwelling infauna and epifauna identified by Montagna et al. (2013) using principle components analysis and spatial interpolation.

Table 4.5-2. Estimates of changes in sediment faunal abundance and diversity within the respective zones identified by Montagna et al. (2013).

Color	Zone	Macrofauna Abundance	Meiofauna Abundance	Macrofauna Diversity	Meiofauna Diversity	Nematode: Copepod Ratio
Red	1	−30.2%	43.2%	−53.7%	−38.3%	240.1%
Orange	2	17.6%	50.9%	−4.5%	−19.0%	20.0%
Yellow	3	25.4%	3.9%	14.5%	−2.4%	−31.3%
Lt Green	4	−13.3%	−43.7%	6.3%	16.4%	−57.5%
Green	5	−7.1%	−27.3%	11.9%	22.8%	−58.4%

doi:10.1371/journal.pone.0070540.t002

In addition, macrofaunal invertebrates prey upon benthic foraminifera (Lipps & Valentine 1970). In the deep sea, benthic foraminifera and other protozoans make up a significant proportion of the biomass, and serve as prey items for numerous macrofaunal organisms. Schwing et al. (2015), working independently from the NRDA, analyzed sediment cores and associated communities of benthic foraminifera. The authors reported changes in benthic foraminiferal densities related to the DWH incident, with declines in density of 80 to 93 percent occurring simultaneously with abrupt increases in sedimentary accumulation rates, PAH concentrations, and changes in redox conditions. They concluded that the decline in foraminiferal density in the surface sediments of the cores was likely caused by the synchronous, significant increase in concentration of low molecular weight (LMW) (2–3 ring) PAHs attributed to the sudden and widespread release of oil during the DWH incident.

4.5.4.1.4 Injuries to Red Crabs

Total numbers of deep-sea red crabs, a top predator that lives on and feeds along the sea floor, were reduced near the wellhead in the year following the spill based on pre- and post-spill data on catch per unit effort (CPUE) (Dixon 2015). In these crab trapping studies, sampling sites tended to either result in no catch (defined as one or no crabs) or resulted in a catch (>11 crabs). This was the case throughout the northern Gulf of Mexico. However, at locations where a catch was reported, the CPUE in 2011 was 40 percent lower than the average catch in all other years where data were available (1987–1989, 2010, and 2014). Further, in 2011, within 50 kilometers of the wellhead, the CPUE decreased to fewer than one crab per trap near the wellhead, but increased steadily moving away from the wellhead (Figure 4.5-12a). For every 12 kilometers in additional distance away from the wellhead, the CPUE doubled. This relationship between distance and the CPUE was no longer evident as of 2014 (Figure 4.5-12a) (Dixon 2015).

Red crabs that survived or did not move out of the area were exposed to and accumulated oil into their tissues. DWH oil was confirmed in red crab hepatopancreas tissues beyond 15 kilometers from the wellhead, and in some locations DWH oil compounds were still present in crab hepatopancreas tissues collected in 2014, more than 4 years after the spill (Douglas & Liu 2015). As of 2011, the presence of that oil signature was also strongly related to the observed decrease in the CPUE. Specifically, a statistical analysis showed that an increase in the exposure of red crabs of 1,240 ppb of TPAH50 in their hepatopancreas was associated with a 50 percent reduction in red crab CPUE. This relationship was no longer present as of 2014, further emphasizing that the cause of the observed decline was the oil spill (Figure 4.5-12b) (Dixon 2015).

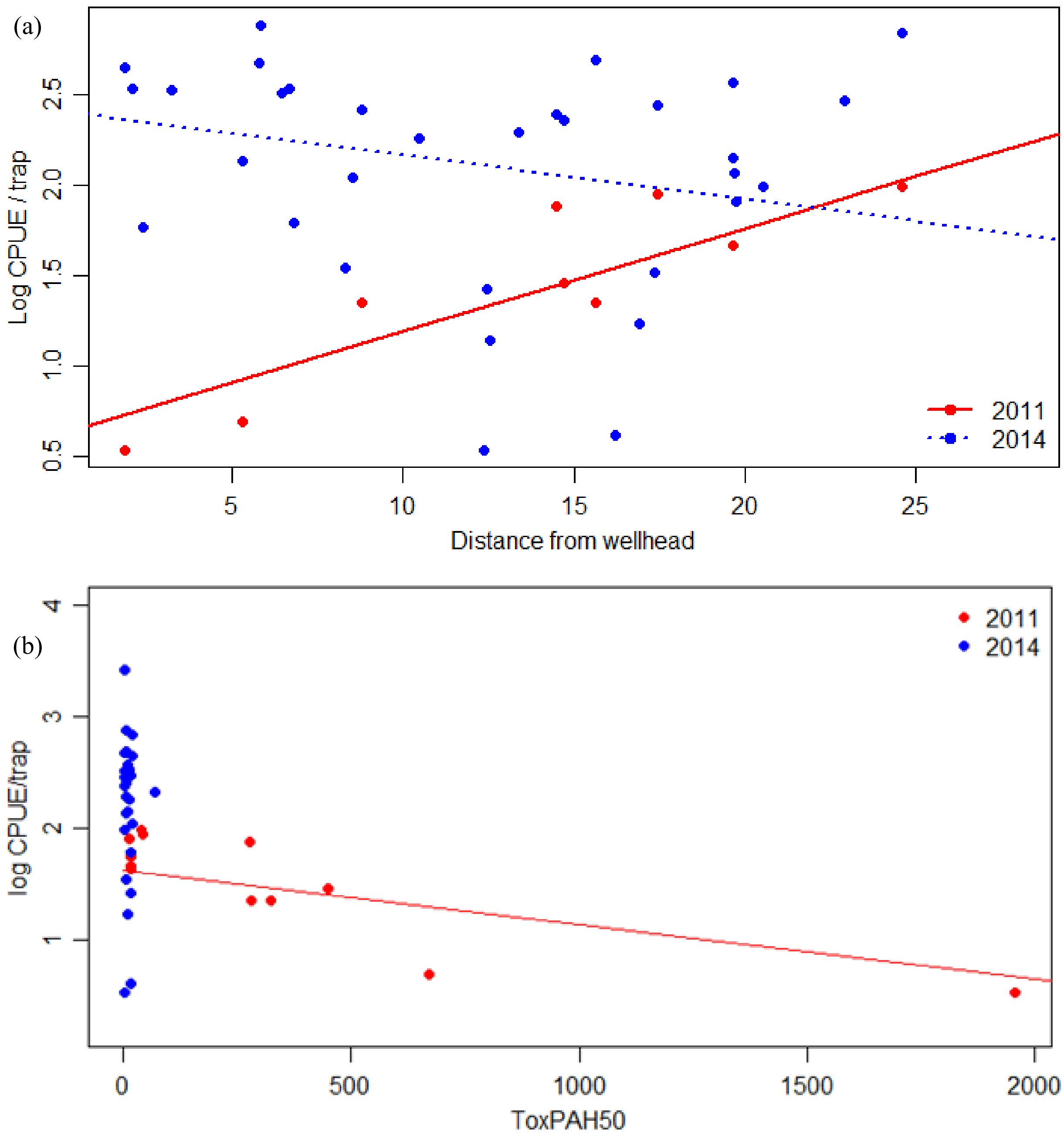


Figure 4.5-12. (a) Plot of log CPUE against distance from wellhead (in kilometers) for sites sampled in 2011 and in 2014. The lines are fitted regression lines for each year. The red line shows that in 2011, CPUE doubled with every additional 12 kilometers of distance from the wellhead, whereas no such relationship was observed in 2014. (b) Plot of log CPUE against average hepatopancreas ToxPAH50, for 2011 and 2014. The line is the fitted regression line for 2011. A 2014 regression line is not fitted because of the small spread in site-average ToxPAH50 values. Data indicate that in 2011, the CPUE decreased by half for each additional 1,240 ppb of PAH exposure, as measured in the hepatopancreas of surviving red crabs.

4.5.4.1.5 Degradation of Hard-Bottom Habitat and Injuries to Corals

Of the seven known hard-bottom or “hardground” coral sites within approximately 25 kilometers of the wellhead, four experienced some degree of injury attributed to the spill. The injury was shown to have occurred coincident in time with the DWH oil spill through a tracking of the progression of observed injury (Figure 4.5-13). This progression of injury showed that the corals initially found covered by floc containing DWH oil and dispersant subsequently experienced mortality and sloughing off of coral tissue. Colonization of injured coral branches by opportunistic hydroid overgrowth followed by tissue death and branch loss is still occurring (Fisher et al. 2015; Hsing et al. 2013; H.K. White et al. 2012; Helen K. White et al. 2012).

The four injured sites lie to the south, southwest, southeast, and east of the wellhead. Both sites within 15 kilometers of the wellhead were injured, with the site closest to the wellhead exhibiting injury to approximately three quarters of coral colonies, and the site slightly farther away exhibiting injury to approximately half of the colonies (Fisher et al. 2014a; 2015; 2014b). The other two injured coral sites lie between 15 and 25 kilometers from the wellhead (Figure 4.5-14). Sampling clearly shows that dispersant and PAHs, toxic constituents of oil, moved to areas at least this far from the wellhead (White et al. 2014). The uninjured coral sites lie upslope to the northwest and northeast of the wellhead. Two are shallower than 1,000 meters of depth, likely outside of the depth zone of the deep-sea plume, and the other is approximately 24 kilometers to the northeast. One of the shallower sites did show injury to two of the 10 corals surveyed, but the presence of fishing line on corals at this site confounded any determination of injury attributable to the spill (Fisher et al. 2014a; 2014b).

Source: *Hsing et al. (2013).*

Figure 4.5-13. Progression of injury to a coral colony at MC 294 from coverage by flocculent material in 2010, through hydroid colonization in 2011 and the onset of terminal branch loss in 2012.

4.5.4 Injury Determination

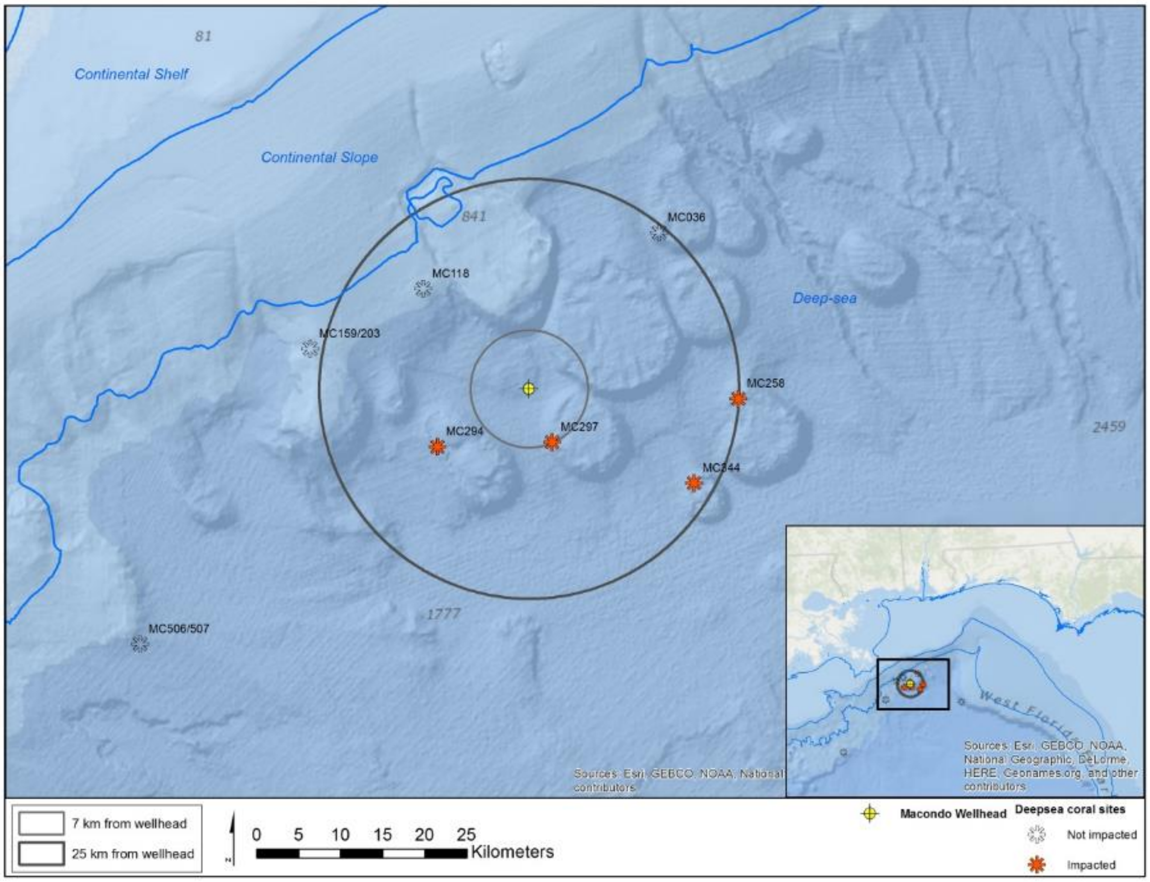


Figure 4.5-14. Map of locations of injured coral sites in relation to the DWH wellhead.

4.5.4.1.6 Adverse Changes to Microbial Community and Sediment Anoxia

A variety of academic studies published independently of the NRDA indicate that microbial communities within 0.5 to 6 kilometers of the wellhead were significantly altered as a result of the oil spill. Alterations in the microbial community are associated with the induction of anoxia and increase in denitrification potential, resulting from degradation of hydrocarbons. These changes represent an alteration of the ability of sediments to recycle carbon and nutrients. Kimes et al. (2013) identified increased proportions of Deltaproteobacteria in proximity to the Macondo well, compared with a distant station. Mason et al. (2014) sampled 64 sites and found that the most contaminated among them were enriched with specific bacterial groups (i.e., Gammaproteobacteria and *Colwellia*). Liu and Liu (2013) also identified unique communities in contaminated sediments near the Macondo well, with a composition similar to natural seep locations. In some cases, the hydrocarbon contamination also appeared to cause microbially induced anoxia within the sediment environment. Kimes et al. (2013) identified benzylsuccinates, metabolic compounds produced during anaerobic biodegradation of hydrocarbons, in sediment cores located in close proximity to the Macondo well. Similarly, Mason et al. (2014) and Scott et al. (2014) used genetic techniques to show an increase in denitrification potential in sediments contaminated with oil from the Macondo well. This indicates a shift away from aerobic metabolism toward more reduced forms of metabolism in the sediments.

4.5.4.1.7 Adverse Changes in the Physical and Chemical Quality of the Deep-Sea Sediment Habitat

PAH concentrations were elevated in sediments contaminated with DWH oil around the wellhead, and in some cases, other locations at significant distances from the wellhead (Stout 2015; Stout & German 2015). The contamination of benthic sediments with chemical constituents generally understood to be toxic represents a measurable adverse change in the physical and chemical quality of the sediment habitat. In some instances, notably closer to the wellhead, PAH concentrations in sediments exceeded toxicity values (LC20 and LC50) determined for deep-sea sediments tested in amphipod toxicity tests (Krasnec et al. 2015). Additionally, PAH concentrations in many benthic sediments collected near the broken well and extending along a northeast-southwest trajectory exceeded concentrations reported as injurious to benthic foraminifera (Romero et al. 2015; Schwing et al. 2015).

4.5.4.2 Continental Slope Injuries

The continental slope is operationally defined by the Trustees as the band of sea floor that is a transition zone from 200 to 800 meters depth between the continental shelf and the deep sea. Similar to other areas in the northern Gulf of Mexico benthos, it is defined by expanses of soft sediment and less prevalent hardground areas, many characterized by populations of corals. Low concentrations of DWH oil were documented in various areas of the continental slope. In particular, the Trustees documented the settling of DWH oil entrained in marine snow (Stout & German 2015), and increased sedimentation of floc extends up the slope (see Figure 4.5-9, above). However, increased sedimentation and oiling of benthic marine resources, notably where sampled in Desoto Canyon, appeared to be diffuse, patchy, and spread across a broad expanse. The Trustees observed concentrations of petroleum in sediment samples to be low. However, Schwing et al. (2015) reported elevated PAH concentrations associated with freshly deposited flocculent material and a large die-off of benthic foraminifera at a location northeast of the wellhead on the continental slope. Multiple visitations by the Trustees to coral locations along the continental slope did not indicate that these habitats were overtly adversely affected by the spill (Fisher et al. 2014a; 2014b). However, oil exposure to deep-sea fish and an associated increase in lesions were reported for some species that feed in the benthos (Murawski et al. 2014). Fish injuries are addressed separately as part of Section 4.4, Water Column, discussing injury to water column resources.

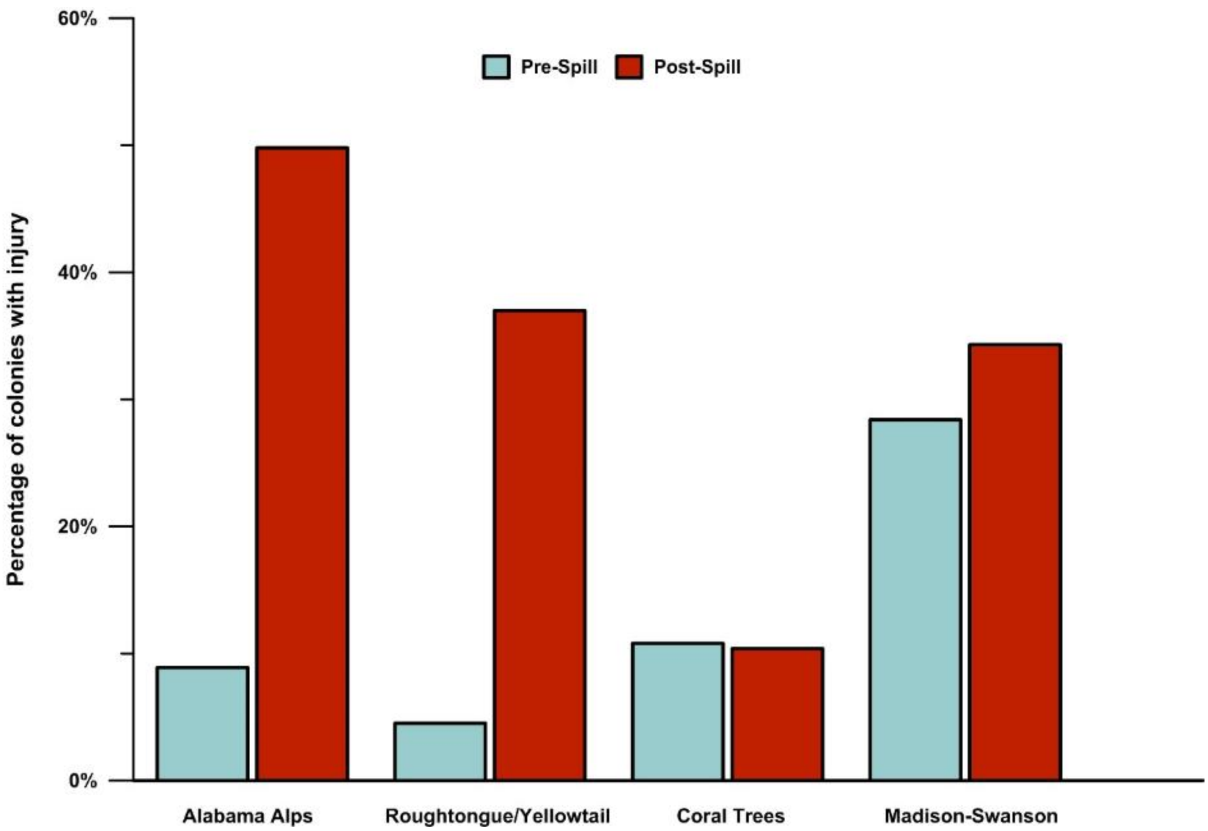
4.5.4.3 Continental Shelf Injuries

The benthos of the continental shelf extends from the nearshore environment, operationally defined by the Trustees as beginning at a depth of approximately 10 meters, out to 200 meters of depth. As with the continental slope and deep-sea regions, the continental shelf sea floor is dominated by soft sediment, with occasional hardground habitats.

As noted in Section 4.5.3.2, the Trustees documented that sediment exposures across the majority of the continental shelf were likely relatively low, but some uncertain higher exposures may have been possible. However, the Trustees identified substantial injury to resources along the edge of the continental shelf and are aware of some accounts of injuries within the larger area of uncertain exposure on the continental shelf published by researchers working independently of the NRDA. For example, Fredericq et al. (2014) observed dramatically reduced amounts of seaweeds and fleshy algae post-spill at rhodolith sites approximately 115 kilometers and 270 kilometers west/southwest of the wellhead. (Rhodoliths are unattached calcium carbonate nodules covered by encrusting algae) (Foster 2001). Researchers also observed declines of decapods associated with these sites (Felder et al. 2014).

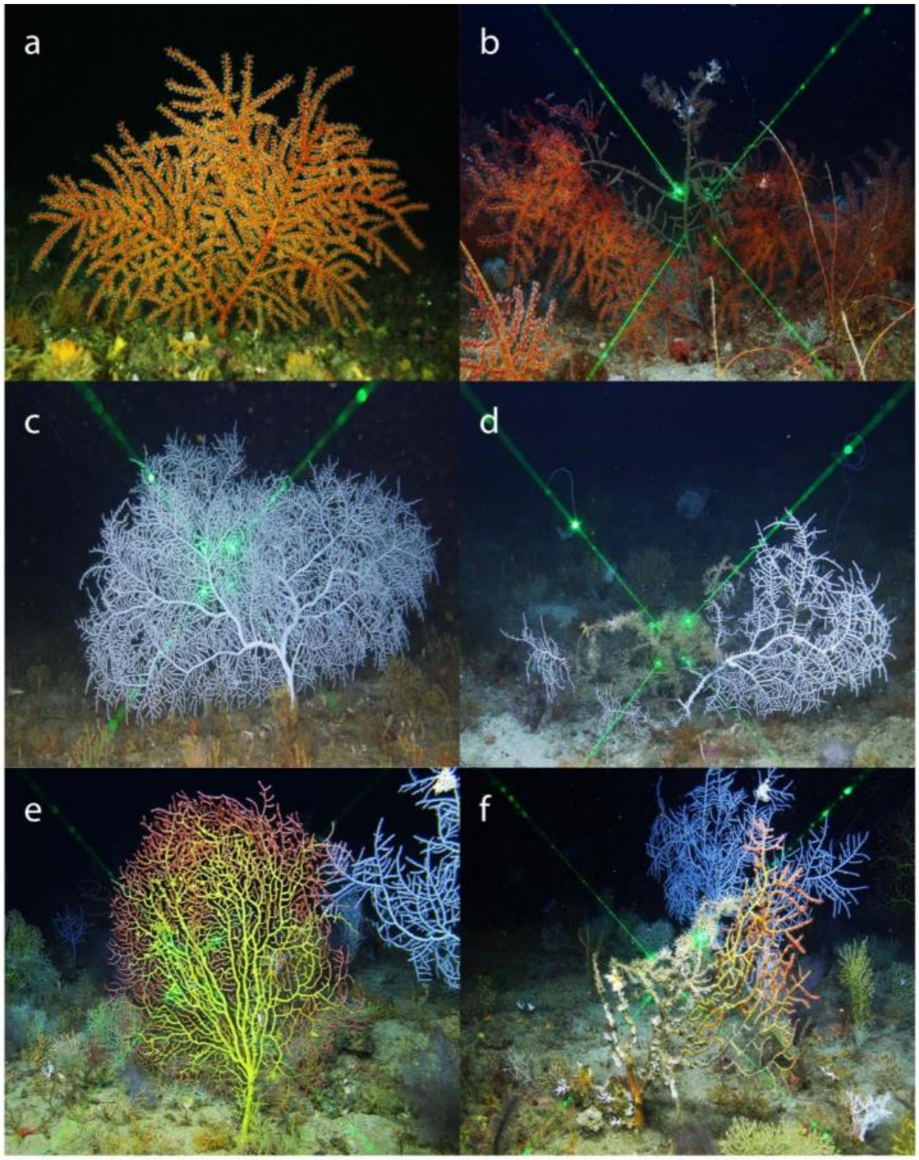
Additional injuries to nearshore rocky reefs have also been suggested. Studies conducted at rocky reefs in the north-central portion of the Gulf of Mexico revealed adverse impacts to fish communities following the spill (Tarnecki & Patterson 2015). Patterson also studied changes in fish community structure at some of these rocky reefs in shallow (15 meters) to mesophotic (90 meters) depths extending south from Alabama to near the continental shelf edge. His findings of reduced numbers of planktivorous fish on the reefs are comparable to findings at the mesophotic reefs (Patterson 2015) where the Trustees documented extensive injury along the continental shelf edge.

Injured mesophotic reefs and their inhabitants on the shelf edge were located underneath the extensive surface slicks as far away as 110 kilometers to the north and northeast of the wellhead. The injured reefs, known as the Pinnacle Reefs, comprise approximately 16 square kilometers of reef-top habitat (Nash & Randall 2015; Nash & Sulak 2015). Based on comparisons to video collected during long-term monitoring projects pre-dating the spill, these diverse biological communities experienced acute mortality of corals—particularly large sea fans and black corals at two reefs studied as part of the NRDA: Roughtongue Reef and Alabama Alps, located approximately 110 kilometers northeast and 60 kilometers due north of the wellhead, respectively (Figure 4.5-15 and Figure 4.5-16) (Etnoyer et al. 2015; Silva et al. 2015)). Depending on the location in areas assessed by the Trustees, approximately one-third to one-half of large sea-fan colonies experienced injury to some degree (Figure 4.5-15). Additionally, order of magnitude decreases in planktivorous fish abundances were observed across the northern Gulf of Mexico (Figure 4.5-17) (Sulak & Dixon 2015). The degradation of mesophotic reef habitat resulting from injuries to sea fans, along with significant reductions of reef-associated fish (relative to pre-spill numbers), was the basis for the Trustees' characterization of severe spill-related effects at affected reefs.



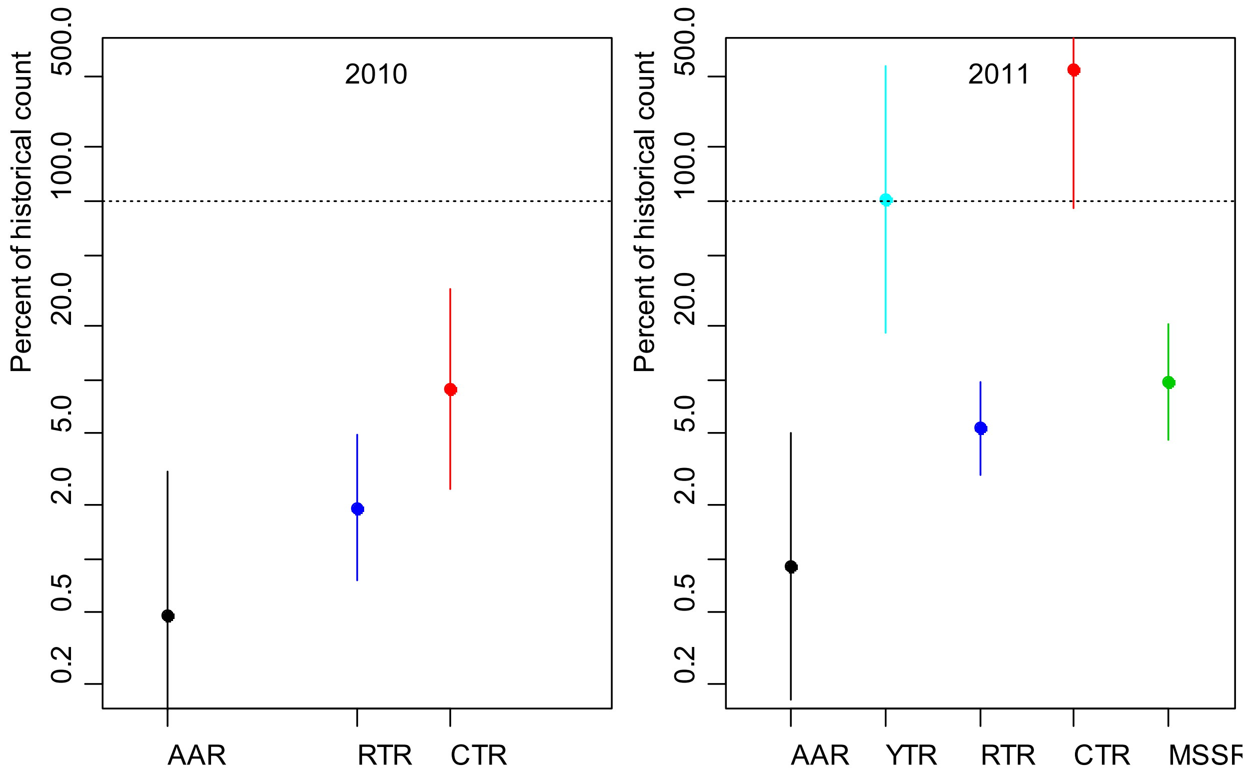
Source: *Etnoyer et al. (2015).*

Figure 4.5-15. Prevalence of injured corals (large sea fans) at mesophotic reef sites in the northern Gulf of Mexico. Bars show the percentage of coral colonies observed in video transects with obvious injuries including bare, denuded, or broken branches; overgrowth; abnormal polyps; or severe discoloration. Pre-spill estimates were derived from video taken in 1989 and 1997 through 2003. Post-spill estimates were derived from video taken in 2010, 2011, and 2014.



Source: *Taken from Etnoyer et al. (2015); photos by Ian MacDonald (a) and Peter Etnoyer (b–f).*

Figure 4.5-16. Examples of healthy (left) and injured (right) colonies of the sea fans (gorgonian octocorals) observed at mesophotic reefs: *Swiftia exserta* (a, b), *Hypnogorgia pendula* (c, d), and *Placogorgia sp*. (e, f).



Source: *Sulak and Dixon (2015).*

Figure 4.5-17. Total count of Anthiinae (planktivorous fish): Estimated percent of historical count, for each sampled reef in 2010 and 2011, with 95 percent confidence intervals. These data show that all three sites sampled in 2010 had a total count that was significantly less than their historical counts in the time period 1997–2003, but the decrease on the two impacted sites (Alabama Alps Reef, AAR; Roughtongue Reef, RTR) was larger than that at the reference site (Coral Trees Reef, CTR). Two of the impacted sites (AAR and RTR) also had decreases from historical counts in 2011 that were significantly larger than the average of the two reference sites (CTR; Madison Swanson Reef, MSSR).

4.5.5 Injury Quantification

Key Points

- The Trustees quantified injuries to resources in two general areas: in the deep-sea benthos around the wellhead and along the continental shelf edge at the mesophotic Pinnacles reefs.
- The footprint of injury to deep-sea benthic habitat around the wellhead was confirmed to encompass over 770 square miles (2,000 square kilometers). The Trustees documented numerous lines of evidence indicating resource injury.
- The magnitude, severity, and frequency of resource injury decreased with increasing distance from the wellhead. The Trustees identified four zones of benthic habitat injury severity, each extending farther from the adjacent inner zone that is closer to the wellhead.
- Although DWH oil was confirmed to have reached areas of the continental slope and shelf (see details in Section 4.2, Natural Resource Exposure), and some evidence of adverse impacts to natural resources was reported in the peer-reviewed literature, concentrations of spill-related contaminants in this area were generally lower than in the deeper benthos. The Trustees did not quantify injuries to natural resources along the continental slope.
- Although exposure of the vast majority of the soft-bottom benthos along the continental shelf to spill-related constituents appears to have been relatively low, the Pinnacles mesophotic reefs on the continental shelf edge were injured. The footprint of injury to mesophotic reefs was identified as encompassing just over 4 square miles (10 square kilometers). An additional approximately 97 square miles (250 square kilometers) of reef hash surrounding the reef-top habitat was identified as encompassing an area of additional potential exposure and injury to mesophotic reef resources.
- Recovery times of resources will be variable. Recovery of soft-bottom sediment and mesophotic reef fish may take years to decades, but recovery of longer lived hardground corals is estimated to be on the order of hundreds of years.
- Benthic resources provide ecological functions such as carbon recycling and production of food, and in some areas provide three-dimensional structure that supports a wide variety of other mobile organisms. While injuries to these resources have the potential to cause more widespread injury to the marine ecosystem, the full set of potential consequences of quantified benthic injuries to the deep-sea ecosystem are not fully understood.

The Trustees quantified injuries to benthic resources by evaluating multiple lines of evidence showing injury. Using geographic information system software to overlay layers of information about benthic areas, the Trustees characterized specific footprints within which resource injuries occurred in the deep benthos around the wellhead and at the Pinnacles mesophotic reefs on the continental shelf. The sub-sections that follow detail the magnitude of habitat injuries by quantifying areas and types of injuries that the Trustees documented in the deep benthos and on the continental shelf.

4.5.5.1 Deep Benthos, Including Soft- and Hard-Bottom Habitat, and Resident Biota

As noted above, the Trustees operationally define the deep benthic zone around the wellhead as the sea floor at depths greater than 800 meters. The benthos in this area was directly affected via all four of the pathways discussed above (underwater plumes, contaminated marine snow, direct contact with contaminated sediments, and uptake of contaminated food):

- The direct fallout of debris and materials associated with the destroyed drilling rig, drilling muds from the failed "top kill" effort, and oil and dispersant droplets entrained in the drilling muds caused smothering of the benthos within 1 to 2 kilometers of the wellhead.
- Additional droplets of oil and dispersants either interacted directly with the sea floor topography or settled to the sea floor as they interacted with marine snow in the water column.
- Oil-associated marine snow settled to the sea floor from the wider area of extensive surface slicks that originated directly above the wellhead.
- PAHs were shown to have been taken up by a variety of organisms, including red crabs (Douglas & Liu 2015).

In total, approximately 2,000 square kilometers of deep-sea benthic habitat immediately around the wellhead was degraded and injured from oil; various spill-related constituents, including drilling muds and dispersants; and debris. NRDA efforts, as well as independent academic studies, showed through forensics and other chemical techniques the presence of DWH oil within this area (Chanton et al. 2015; Stout et al. 2015; Valentine et al. 2014). However, because of the patchiness and unevenness of impacts, injuries to natural resources appear to decrease in severity with increasing distance from the wellhead.

For purposes of injury quantification, the Trustees categorized spill-related injuries to deep-sea resources into the following four zones (see Table 4.5-3 and Figure 4.5-18):

- **Zone 1**, which encompasses an area within approximately 3 kilometers surrounding the wellhead, totals approximately 28 square kilometers. Zone 1 experienced the greatest degree of adverse impacts from the spill as evidenced by the deposition of liquid oil, physical fouling of habitats, presence of smothering debris and toxic sediment, and degradation of habitat sufficient to cause major shifts in both diversity and abundance of animals living on and in the sediment.
- **Zone 2**, located from 3 to 7 kilometers in all directions and extending farther (15 kilometers) to the southwest of the wellhead, totals approximately 195 square kilometers. Zone 2 experienced an adverse shift in sediment faunal diversity, degradation of habitat quality through oil and dispersant contamination of sediment and corals, and mortality of corals, where such contamination was present.
- **Zone 3**, located between 7 and 25 kilometers of the wellhead, totals approximately 793 square kilometers. Zone 3 has lesser amounts of coral mortality (relative to coral sites in Zone 2), less

widespread and/or patchy impacts to sediment-dwelling biota, and persistence of measurable concentrations of the toxic constituents of DWH oil.

- **Zone 4**, located roughly between 25 and 45 kilometers southwest from the wellhead, totals approximately 1,275 square kilometers. Zone 4 represents an area where there was an adverse change in the chemical quality of the habitat by the deposition of DWH oil. Sediments at some locations in this zone had TPAH50 concentrations that exceeded toxicity values determined in laboratory tests with amphipods exposed to deep-sea sediments collected in the vicinity of the wellhead. Mortality to test animals of *Leptocheirus*, an amphipod genus that occurs in the deep-sea benthos (Figure 4.5-19), suggests mortality would occur to comparable organisms exposed to similarly or more contaminated sediments in this zone. An additional account by Schwing et al. (2015) reported declines in foraminiferal density correlated with elevated concentrations of LMW PAHs. Sediments in scattered areas of Zone 4 exceeded these LMW PAH concentrations, further supporting an assertion of adverse impacts because of oil contamination of sediments (Figure 4.5-20). The magnitude of injury to the biota from the degradation of habitat quality is considered patchy based on uneven deposition of oil and floc throughout this zone. Some resident species such as red crabs were documented to have tissues contaminated with DWH hydrocarbons, and this contamination represents a degradation of food quality for organisms that prey on red crabs.

An additional zone of uncertain exposure and injury extends approximately 400 kilometers to the southwest of the wellhead. This area represents benthic habitat that likely was exposed to some degree by the subsurface oil/dispersant plume that migrated with ocean currents to the southwest and followed the bottom topography between 800 and 1,600 meters depth (Section 4.2, Natural Resource Exposure). This area was not sampled extensively for biological impacts because of its broad footprint, extreme depth, and the Trustees' focus for assessment activities on areas closer to the wellhead where injuries were anticipated to be greatest.

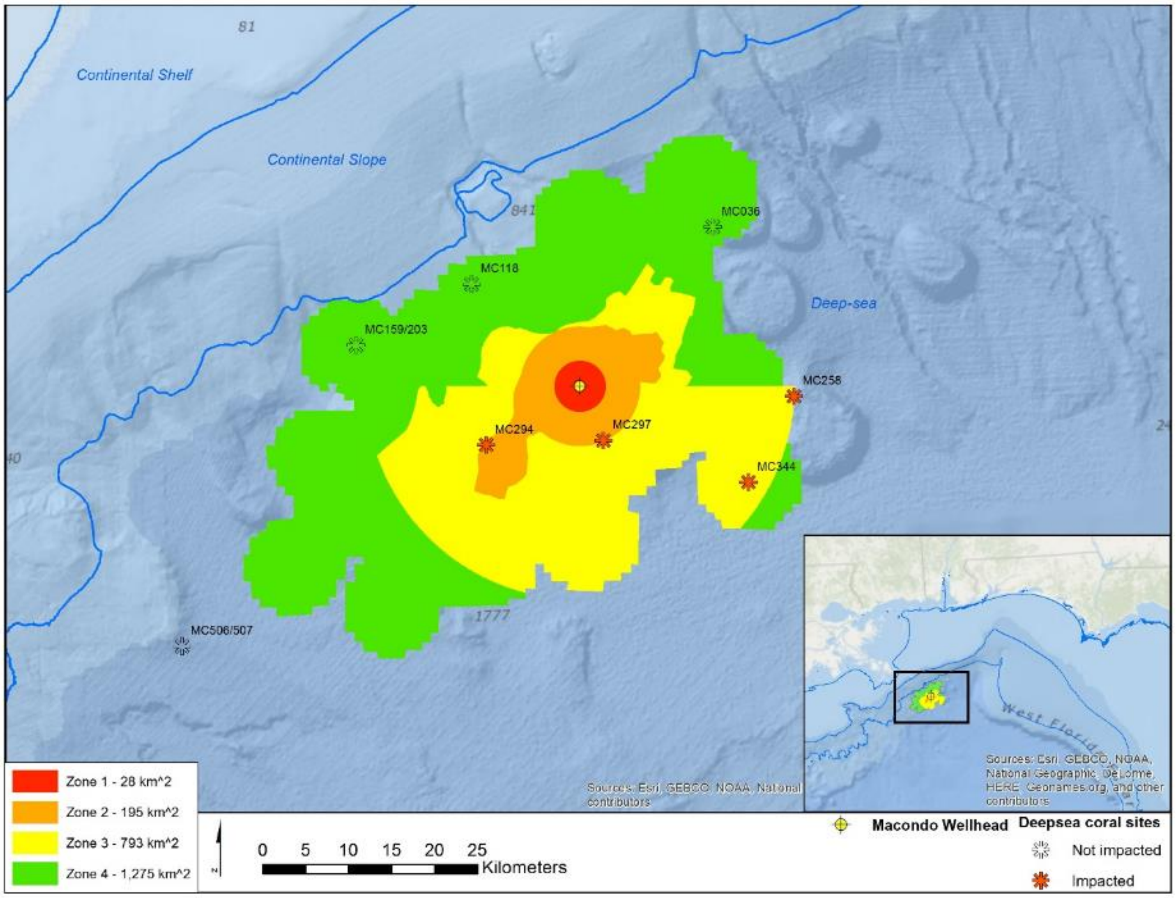


Figure 4.5-18. Quantified footprint of the DWH oil spill within which injuries to deep-sea resources are identified. Multiple lines of evidence suggest that habitat degradation and adverse impacts on marine residents in the vicinity of the wellhead appear to decrease with increasing distance from the wellhead. Table 4.5-3 provides a description of the exposure and injuries documented within each quantified zone.

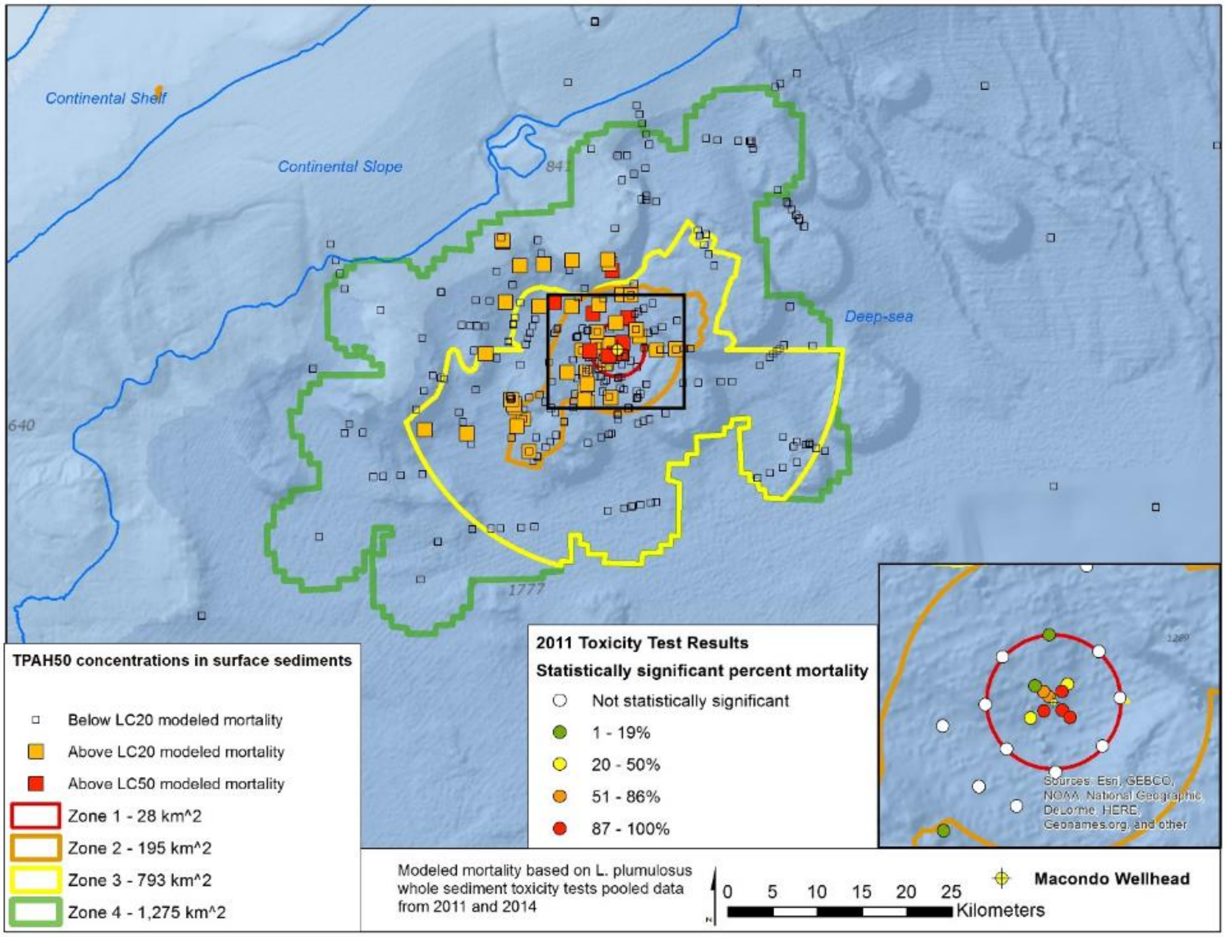


Figure 4.5-19. Map indicating surface TPAH50 concentrations that exceed LC20 and LC50 values for modeled mortality compared with quantified zones of injury.

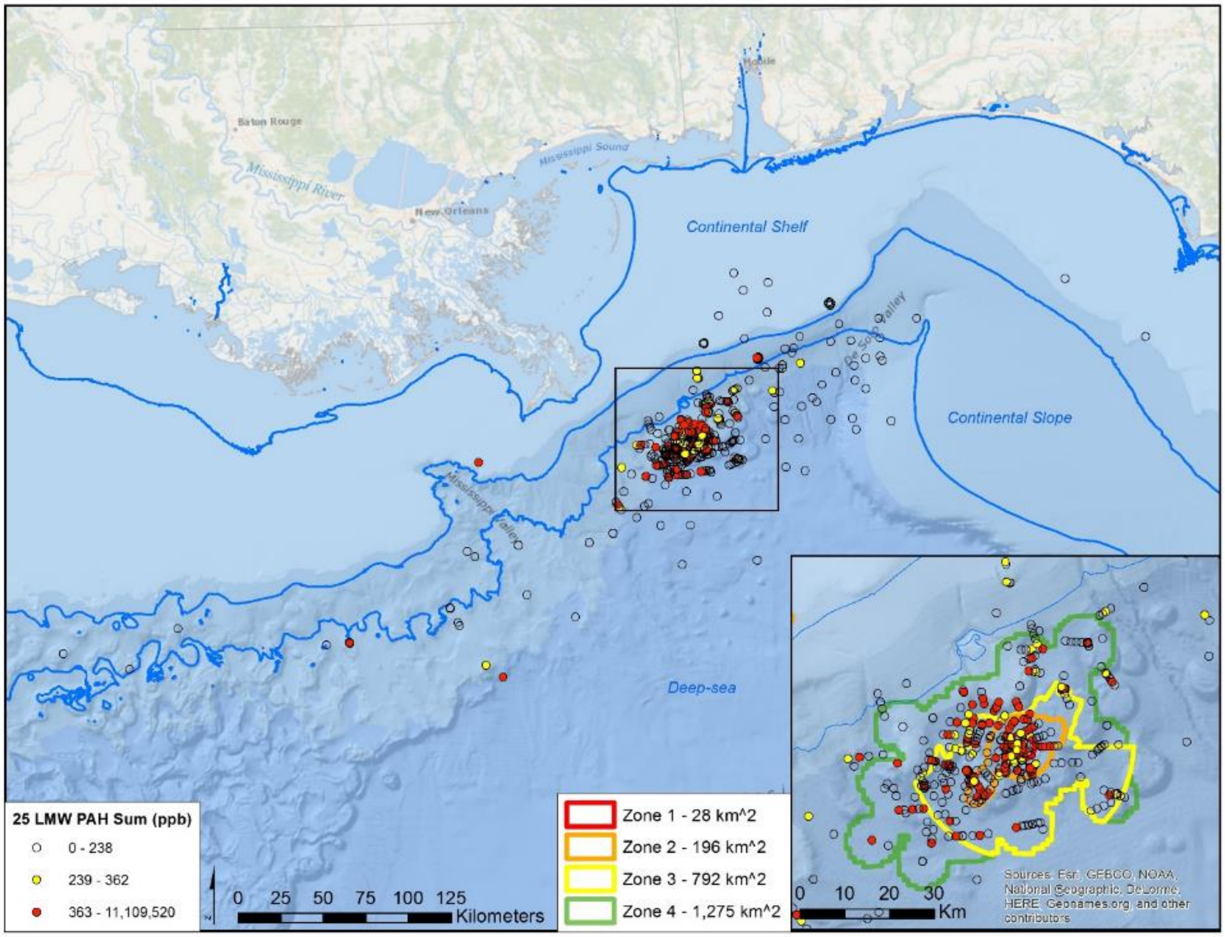


Figure 4.5-20. Map showing exceedance of LMW PAH concentrations (238 ppb for one site [PCB06] and 362 ppb for another site [DSH08]) reported by Schwing et al. (2015) as correlating strongly with 80 to 93 percent declines in foraminiferal densities. *Inset:* Close-up detail of zones 1–4 and distribution of sediments that exceed the sum of LMW (2–3 ring) PAHs. LMW PAHs include 1-methylnaphthalene; 1-methylphenanthrene; 2,6-dimethylnaphthalene; 2-methylanthracene; 2-methylnaphthalene; 2-methylphenanthrene; 3-methylphenanthrene; 4/9-methylphenanthrene; acenaphthene; acenaphthylene; anthracene; C1-dibenzothiophenes; C1-naphthalenes; C1-phenanthrenes/anthracenes; C2-dibenzothiophenes; C2-naphthalenes; C2-phenanthrenes/anthracenes; C3-naphthalenes; C3-phenanthrenes/anthracenes; C4-naphthalenes; C4-phenanthrenes/anthracenes; dibenzothiophene; fluorene; naphthalene; phenanthrene.

Table 4.5-3. Zones of deep-sea benthic injuries.

Injury Zone	Adverse Effects			Injury/Exposure Data	Supporting Documentation
	Contamination: Degradation of Habitat and Ecosystem Quality	Changes in Habitat and Ecosystem Health or Functionality	Mortality		
Deep-Sea					
Zone 1 **28 km^2**	X			Presence of nonsoluble liquid inclusions (interpreted as liquid oil).	Germano and Associates Inc. et al. (2012)
	X			Presence of drilling mud.	Germano and Associates Inc. et al. (2012)
	X			Presence of in situ burn residues on sea floor.	Stout and Payne (2015)
	X			DWH oil fingerprinted PAHs in sediments.	Stout et al. (2015)
	X			Presence of dioctyl sodium sulfosuccinate (DOSS), a dispersant constituent, in surface sediments.	NRDA 2011 Hardbottom Plan and Soft-bottom Sediment Sampling Plan results
		X	X	Benthic community response in this zone: 54% decrease in macrofauna diversity. 38% decrease in meiofauna diversity. 30% decrease in macrofauna abundance.	Montagna et al. (2013)
		X	X	Statistically significant mortality between 19–100% of *L. plumulosus* for 10 out of 17 samples in this zone.	(EEUSA & Cardno ENTRIX 2011a, 2011b)
		X		809% change in N:C ratio for 0.35 km^2 in this zone. 220% change in N:C ratio for the majority of the remaining area in this zone.	Baguley et al. (2015)
Zone 2 **195 km^2**	X			DWH oil fingerprinted PAHs in sediments.	Stout et al. (2015)
	X			DWH oiled sediments in sediment traps	Stout and Passow (2015)
	X			Presence of DOSS in surface sediments.	NRDA 2011 Hardbottom Plan and Soft-bottom Sediment Sampling Plan results
			X	Total coral colony losses at MC297.	Fisher et al. (2014a); (2014b)
		X	X	Benthic community response in this zone: 5% decrease in macrofauna diversity. 19% decrease in meiofauna diversity.	Montagna et al. (2013)

Injury Zone	Adverse Effects			Injury/Exposure Data	Supporting Documentation
	Contamination: Degradation of Habitat and Ecosystem Quality	Changes in Habitat and Ecosystem Health or Functionality	Mortality		
	X		X	TPAH50 concentrations exceed LC20 and LC50 modeled mortality.	Krasnec et al. (2015)
		X		220% change in N:C ratio for approximately 43 km^2 of this zone. 83% change in N:C ratio for the majority of the remaining area in this zone.	Baguley et al. (2015)
Zone 3 **793 km^2**	X			DWH fingerprinted PAHs in sediments.	Stout et al. (2015)
	X			DOSS present in floc sampled from corals and/or in surface sediments at MC294 and MC344 (MC388).	H.K. White et al. (2012) White et al. (2014)
	X			DWH fingerprinted PAHs in red crab tissues.	Stout et al. (2015)
	X			Presence of DOSS in surface sediments.	NRDA 2011 Hardbottom Plan and Soft-bottom Sediment Sampling Plan results
		X	X	Coral colony injuries at MC294, MC297, MC344 (MC388), and MC258.	Hsing et al. (2013) Fisher et al. (2014a); (2014b) Fisher et al. (2015)
			X	TPAH50 concentrations exceed LC20 and LC50 modeled mortality.	Krasnec et al. (2015)
		X		83% change in N:C ratio for approximately 150 km^2 in this zone.	Baguley et al. (2015)
Zone 4 **1,275 km^2**	X			DWH fingerprinted PAHs in sediments.	Stout et al. (2015)
	X		X	TPAH50 concentrations exceed LC20 and LC50 modeled mortality. LMW PAH values exceed concentrations reported to cause declines in densities of benthic foraminifera.	Krasnec et al. (2015) Schwing et al. (2015)
	X			DWH fingerprinted PAHs in red crab tissues.	Douglas and Liu (2015)
	X			Presence of DOSS in surface sediments.	NRDA 2011 Hardbottom Plan and Soft-bottom Sediment Sampling Plan results

4.5.5.1.1 Ecological Implications of Deep Benthic Injuries

In the case of the large footprint of deep benthic resource injury around the wellhead, the Trustees evaluated multiple lines of evidence, which, taken together, indicate both significant mortality and the degradation of habitat available to support life. The lines of evidence used to support the Trustees' conclusions related to resource injury suggest that ecosystem level impacts were experienced, with injuries to biological resources at multiple levels of the food web, as well as habitat contamination that severely degraded those areas of the sea floor closest to the wellhead.

The ecological significance of these injures varies with their severity. For example, within Zone 1, toxic sediment, inclusions of liquid oil, and the presence of debris have made this area of the benthos closest to the wellhead unable to support the kinds of animals that lived in and on the sediment prior to the spill. Within Zone 2, the Trustees documented major shifts in the numbers and types of animals that live in and on the sediment. These shifts could have been the result of changes in the food web beginning with the microbes that inhabit the sediment and reverberating upward, or the direct result of exposure to oil or other spill-related constituents. It is reasonable to assume that such shifts continue to reverberate upward through the food web, affecting larger, more motile predators and thus extending the geographic influence of such change. Different prey sources have different energetic benefits, and even subtle shifts in the dominant prey available for higher trophic-level organisms can lead to shifts in the associated predator populations (NRC 2006). Further, such changes in the food web have the ability to change some of the most important ecological services that the deep benthos provides, principally the recycling of energy and nutrients from detritus falling to the sea floor back up into the water column (Kristensen et al. 2014).

Within Zone 3, injury was patchy, but the injuries have ecological significance nevertheless. For example, injuries to hardground corals manifested over time in the breakage of branches and overall reductions in the sizes and health of coral colonies. While the full suite of ecosystem functions of these unique hardground corals are still only sparingly understood in the deep ocean, ecological functions from other fan-like coral species growing in shallower habitats include increased vertical structure yielding cover and protection to mobile biota seeking refuge from predation and places to live and breed. It is reasonable to believe that similar services would be provided by the deep-sea fan-like corals. In fact, three-dimensional structure provided by deep-sea coral habitats is associated with increased biodiversity (Buhl-Mortensen et al. 2010). Deep water corals can therefore be considered to be sentinel species, providing a lasting visible record of deleterious impact that cannot be detected for most of the deep living mobile species (Fisher et al. 2014a; 2014b).

Within Zone 4, degradation of the chemical quality of the benthic habitat by contamination with DWH oil was confirmed throughout this area (Stout et al. 2015). DWH oil also was confirmed in the tissues of red crabs collected in this area (Douglas & Liu 2015). The documentation of biological uptake and contamination of prey indicates some degree of fouling of the food web within this zone. The concentrations of TPAH50 in some sediments from this zone exceeded LC20 and LC50 values for benthic amphipods, which suggests injury to sediment-dwelling fauna. Additionally, sediments collected from areas widespread throughout Zone 4 exceed total concentrations of LMW PAH that were previously reported by Schwing et al. (2015) as detrimental to foraminifera. Potential impacts to this community of protozoans, which is an essential prey component for benthic macroinvertebrates, further supports an assertion of adverse impacts because of oil contamination of sediments. The ecological implications of spill-related losses within Zone 4 are not fully known.

Injuries documented by the Trustees may also have other unknown ecosystem impacts. Emerging information suggests that hardground coral habitats provide valuable and perhaps even unique ecological services that may have been reduced as a result of the observed injuries. For example, fishes, including some commercially significant species, have been shown to have elevated abundance near *L. pertusa* mounds in the South Atlantic Bight (Ross & Quattrini 2007). In the northern Gulf of Mexico, the goosefish (*Sladenia shaefersi*) has only been observed associated with deep-sea hardground habitats (Figure 4.5-21: photos of *Sladenia* fish, only observed in deep-sea hardground habitat) (Pietsch et al. 2013). Baillon et al. (2012) provide evidence that redfish larvae use cold water corals as nursery habitat. Finally, Etnoyer and Warrenchuk (2007) and Fisher et al. (2014a) and (2014b) have reported that live deep water octocorals, including *Paramuricea*, can host egg cases of a chain cat shark in the northern Gulf of Mexico. These are but several examples of potentially unique but important roles that deep-sea habitats can play in supporting the larger marine ecosystem. The extent to which quantified injuries result in additional adverse effects to these associated organisms is unknown.

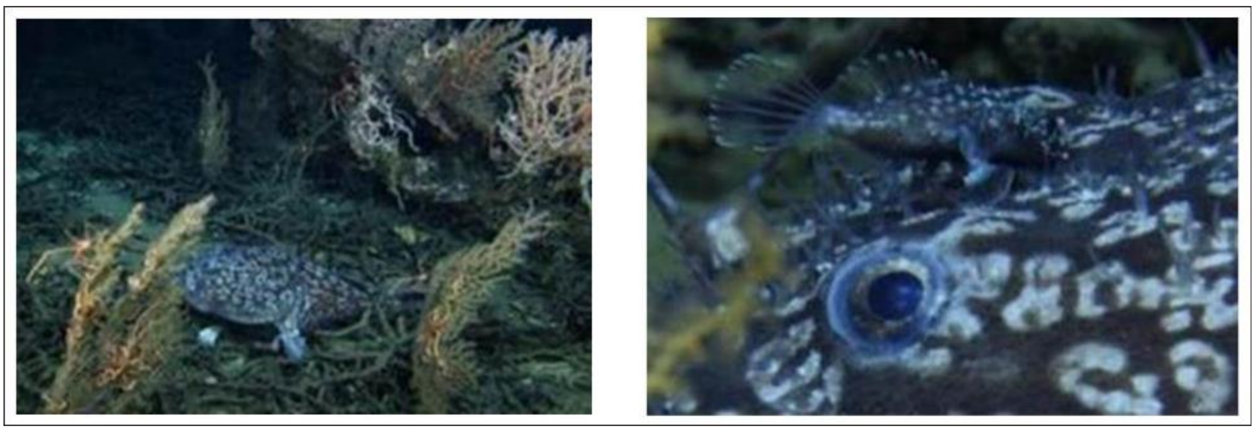


Figure 4.5-21. Photos of goosefish (*Sladenia haefersi*) at a deep hardground coral habitat. This fish species has only been observed in deep-sea hardground coral habitats. The precise relationships between this rare fish and the habitat in which it lives are not fully understood; therefore, injuries to coral habitats may have unknown consequences for other organisms, such as this goosefish.

4.5.5.2 Continental Slope

The Trustees did not quantify specific injuries to natural resources within this area because of uncertainty associated with the extent of resource exposure and injury along the continental slope. However, patchy spill-related impacts over at least 3,300 square kilometers of benthos are likely where oil persisted on the sea surface, dispersants were frequently applied in significant quantities, and increased amounts of flocculent material settled on benthic sediments. Figure 4.5-22 shows the footprint where all three phenomena (surface oil, surface dispersant spraying, and sediment floc) overlap.

The likelihood of patchy injury is supported by data from Schwing et al. (2015) documenting smothering of resident foraminifera in a floc-impacted location on the continental slope. Some degree of oil and dispersant exposure to benthic organisms is expected along the slope from the surface oil and dispersant sinking to the benthos as part of contaminated marine snow and the extreme sedimentation event, known as the "dirty blizzard" (Brooks et al. 2015). This area was not sampled extensively for biological impacts as part of the NRDA, because of the Trustees' focus closer to the wellhead where injuries were anticipated to be greatest. Where samples were taken, exposures were generally low (Stout & German 2015).

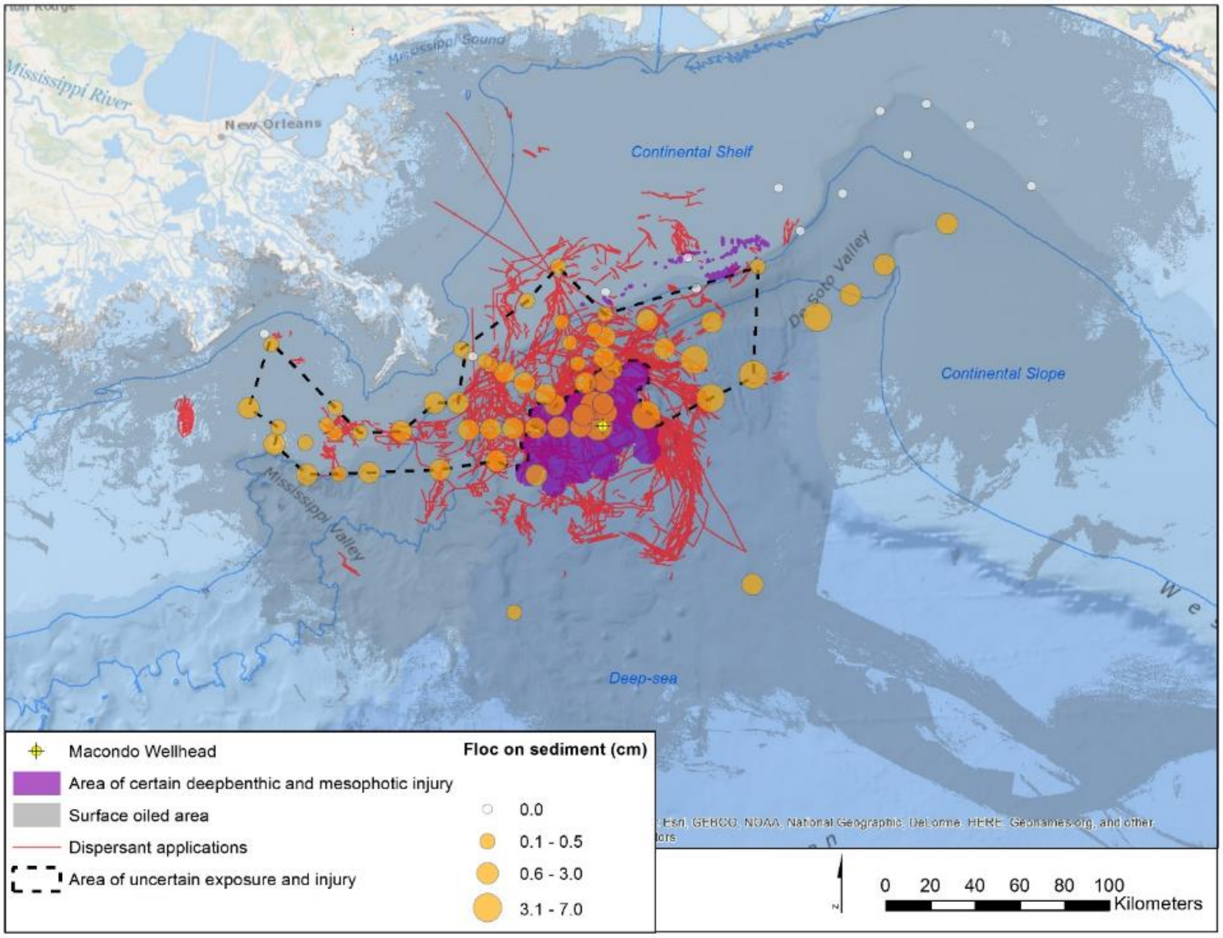


Figure 4.5-22. Potentially exposed and affected areas of benthos (indicated as the area contained within the dashed line) extend beyond the area of certain deep-sea and mesophotic reef affected areas based on multiple lines of evidence, including the surface oil area, surface dispersant applications, flocculent layer, and discrete areas of fingerprinted DWH oil in sediment and benthic fauna tissue.

4.5.5.3 Continental Shelf

The information on mesophotic reef fish and coral injury presented above suggests that habitat degradation and adverse impacts on marine residents within an area spanning at least the distance between Alabama Alps and Roughtongue Reef has occurred. The Trustees therefore quantified injury to the area of the Pinnacles reefs to the west of Roughtongue Reef. The area of reef-top habitat within this portion of the Pinnacles reef tract is estimated at approximately 10.4 square kilometers (Figure 4.5-23 and Table 4.5-4). Furthermore, a larger area of reef hash substrate (scattered rocks and rubble) that surrounds the elevated reefs themselves, totaling approximately 250 square kilometers, represents an area of unknown impacts (Figure 4.5-23) (Nash & Randall 2015; Nash & Sulak 2015). Although the larger reef hash area does not support the density of corals that the reef tops do, this larger reef hash apron is a destination for a variety of mobile species that feed along the Pinnacles tract (Sulak & Dixon 2015).

An additional footprint of approximately 3,300 square kilometers on the continental shelf north of the wellhead (see Figure 4.5-22 above) represents an area where oil persisted on the sea surface, dispersants were frequently applied in significant quantities, and increased amounts of flocculent material were observed atop benthic sediments. As noted above, the Trustees identified several published accounts of injuries within this larger area of uncertain exposure and injury on the continental shelf, particularly to the far west/southwest of the wellhead and to the far east of the wellhead (Fredericq et al. 2014; Schwing et al. 2015). However, the Trustees did not quantify losses across this larger area of presumed patchy and uncertain exposure and injury.

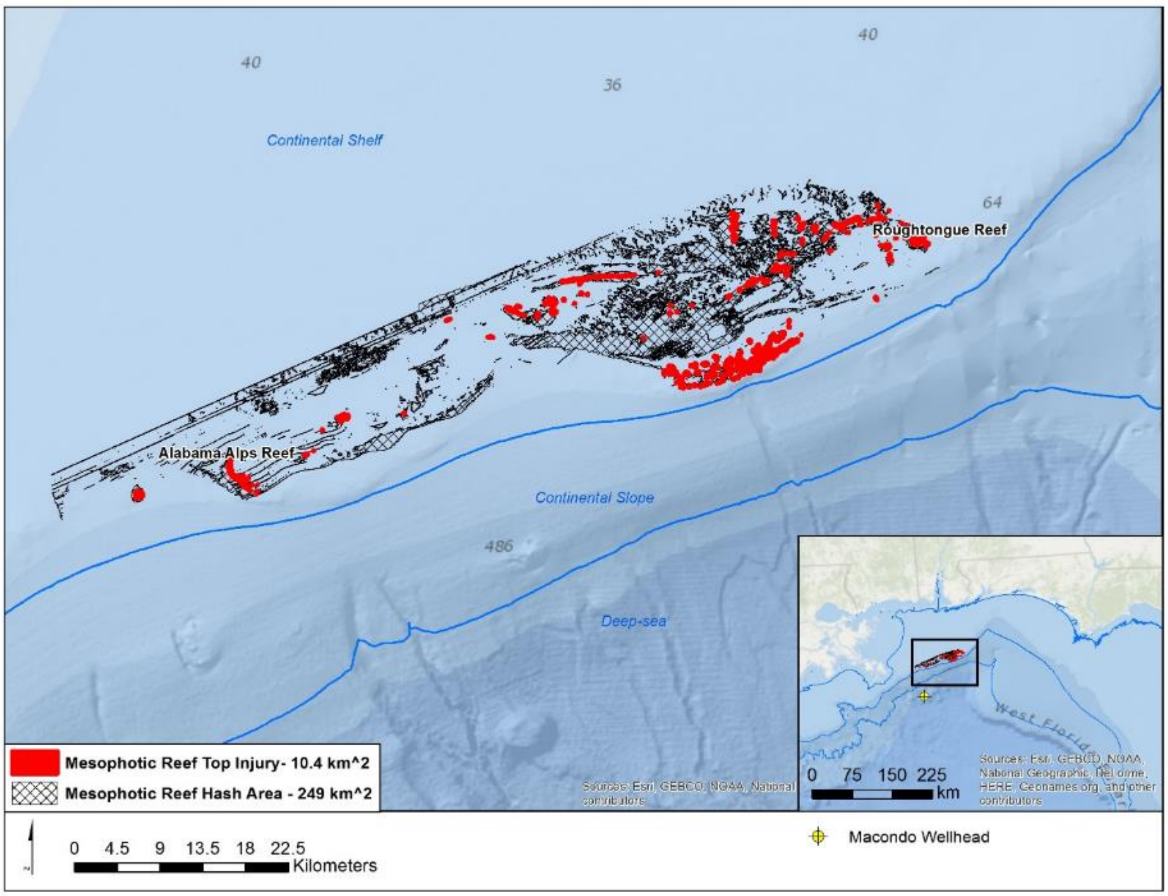


Figure 4.5-23. Footprint of the DWH oil spill impacts on mesophotic resources. Impacts to mesophotic injury are quantified at 10.4 square kilometers (red area). The mesophotic reef hash is a larger area of habitat surrounding the elevated reefs that is a destination for a variety of pelagic species that feed along the Pinnacles tract. This reef hash area is an area of uncertain impacts.

Table 4.5-4. Continental shelf exposure and injuries.

Continental Shelf					
Injury Zone	Adverse effects			Exposure/Injury Data	Supporting Documentation
	Contamination: Degradation of Habitat and Ecosystem Quality	Changes in Habitat and Ecosystem Health or Functionality	Mortality		
Mesophotic Reef Injury 10.4 km^2	X			SPMDs collected DWH fingerprinted PAHs.	Stout and Litman (2015)
			X	Significant decrease in abundance of planktivorous fish.	Sulak and Dixon (2015)
		X	X	Increase in prevalence of coral injuries relative to pre-spill and control sites.	Etnoyer et al. (2015); Silva et al. (2015)
Reef Hash Area 249 km^2	Larger area of habitat surrounding the elevated reefs that is a destination for a variety of species that feed along the Pinnacles tract. This reef hash area is an area of uncertain exposure and injury.				Sulak and Dixon (2015)

4.5.5.3.1 Ecological Implications of Pinnacles Mesophotic Reef Injuries

As with the footprint of habitat injury in the deep-sea quantified by the Trustees, injury to the Pinnacles reefs habitat was asserted based on documented injuries to several representative resources; specifically, the dominant mesophotic reef planktivorous fish and large sea fan corals. The ecological importance of planktivorous fish injury is tied to their role as prey. Large numbers of off-reef soft sediment shelf fishes, deep water fishes, and mobile pelagic water column fishes feed extensively on mesophotic reef invertebrates and fishes (Weaver et al. 2002)). Transfer of energy takes place from the reef ecosystem to the broader northern Gulf of Mexico ecosystem (Weaver et al. 2002). But reef fish are also understood to provide additional ecological services, which were adversely affected by the reduction in their population numbers (Hamner et al. 2007). Planktivorous reef fish produce fecal pellets that provide food for both particulate suspension- and deposit-feeding invertebrates and the microbial community occupying all components of the reef. Therefore, the loss of these fish potentially caused ecological impacts throughout the food web.

The ecological significance of loss of mesophotic reef corals is also best understood through the types of services that they provide, which have been adversely affected by their loss. For example, the pelagic larvae of sea fans serve as a re-population source for adjacent and distant reefs. Rising above the hard reef surface, the three-dimensional structure of soft corals also interrupts laminar bottom currents impinging upon the reefs, creating turbulence in the zone of topographically accelerated bottom currents (Gittings et al. 1992; MacDonald & Peccini 2001; Messing et al. 1990). This turbulence increases particulate residence time in the near-bottom water column, enhancing the availability of particulates as food for plankton-feeding fishes and other particulate-feeding invertebrates (Sulak & Dixon 2015). As with the soft corals of the deep sea, injury to tall mesophotic reef soft corals has reduced the amount of complex, tree-like, three-dimensional habitat that is important to fishes as refuge from predators, visual camouflage, and energetic refuge from strong bottom currents. Reef fish species that use prominent tall corals as landmarks to organize daily activity and social organization may have also been adversely affected. Tall soft corals also provide living habitat for micro-crustaceans and elevated feeding perches for other opportunistic megafaunal invertebrates, such as basket starfish that feed nocturnally upon small fishes, and provide food for a small number of fishes and invertebrates that graze upon coral polyps. Therefore, the documented injury to corals has both potential food web and other structure-related ecological significance.

The overall ecological importance of the Pinnacles reefs injuries to the larger northern Gulf of Mexico belies the small total area they comprise (Nash & Sulak 2015). Their ecological significance within the larger northern Gulf of Mexico is detailed in a series of papers and proceedings compiled by NOAA from a scientific forum on the "Islands in the Stream Concept," which evaluated potential approaches for conservation of these habitats (Ritchie & Keller 2008). Additionally, the diversity and trophic significance of these reefs as fish habitat is detailed in Weaver et al. (2002). In contrast to the open, soft-sediment plain of the outer continental shelf, these reefs represent unique, elevated, hard-bottom biotope that supports living three-dimensional habitat and high biological productivity and biodiversity. Furthermore, an area of reef hash substrate (areas of scattered rocks and rubble that include remnants of coral branches, mollusk shells, sea urchin tests, and other biogenic carbonate parts) surrounds the reef proper. This reef hash area provides its own set of resource services and habitat and protection for infauna and epifauna that are regularly foraged by many reef-attached species, such as groupers, as well as pelagic species. Although the Trustees did not directly assess impacts from the spill to this larger approximately 250-square-kilometer reef-hash area, it is likely that it was similarly exposed to DWH oil and dispersants. Therefore, it represents an area of uncertain exposure and injury around the approximately 10-square-kilometer mesophotic reef-top area within which the Trustees are asserting injury. The injury of these reefs undoubtedly represents a loss to a geographic area larger than the physical confines of the reefs themselves.

4.5.5.4 Injury Quantification Uncertainty

Within the more thoroughly studied and well characterized deep-sea and mesophotic benthic areas, the Trustees acknowledge that the full extent of spill-related losses is unknown, but likely greater than what was documented and summarized in this section. This likelihood is based on two factors: 1) the Trustees did not study everything in these areas and focused only on a subset of resources, and 2) the ecological interactions of resources, in some cases, are not fully characterized or understood in the deep-sea and mesophotic habitats.

Beyond the areas well-studied by the Trustees, uncertainty about spill-related adverse effects in the benthos increases with distance from the blown-out well. The overall magnitude of the spill and logistics associated with working on the sea floor meant that not all areas could be studied with the same level of intensity. For example, oil from any part of this surface slick could have sunk and exposed benthic resources below. Any sunken oil over this extremely large footprint would likely be very patchy, making it difficult for the Trustees to document exposure to and injury of benthic resources.

Figure 4.5-22 presents a benthic footprint for an area below a sea surface that was covered by heavy and persistent oil and repeatedly sprayed with large volumes of dispersants. Additionally, flocculent materials were documented as increased layers atop benthic sediments in this area. This footprint of 9,200 square kilometers (3,300 on the shelf, 3,300 on the slope, and 2,600 in the deep sea) falls generally between the areas of documented deep-sea and mesophotic reef injury and extends along an east-west trend consistent with topography and predominant currents. The Trustees documented some exposure and injuries of benthic resources within this area, but the concentrations of oil were low and injuries were considered to be patchy and localized.

4.5.5.5 Recovery

4.5.5.5.1 Deep Benthic Recovery

As of the writing of this document, spill-related contamination of the deep benthos zone has persisted for at least 4 years and may persist for much longer. Some recovery of benthic habitat may have already occurred, although recovery of different components of the benthic ecosystem will clearly take differing amounts of time based on the vastly different life cycles of species affected and ages of individuals killed. For example, sediment in close proximity to the wellhead still showed acute toxicity in samples collected in 2014 (Krasnec et al. 2015), but concentrations of oil compounds in these surface sediments declined between 2010 and 2014, suggesting reduced future exposures. Similarly, sediment sampling in 2011 already showed some shifts in faunal densities back toward baseline, but low sedimentation rates near the wellhead under natural conditions suggest it is unclear how long sediments may remain toxic to benthic marine resources. Other benthic parameters still showed evidence of persistent impacts. Similarly, northern Gulf of Mexico-wide populations of red crabs as of 2014 appeared to have returned to pre-spill levels, yet red crabs continue as of 2014 to be exposed to DWH oil (Douglas & Liu 2015). In contrast, deep-sea coral colonies, some of which were killed as a result of the spill, live in excess of 500 years, and exhibit very low recruitment rates, suggesting a significantly longer recovery time (Prouty et al. 2011; 2014).

4.5.5.5.2 Mesophotic Recovery

As is the case in the deep benthos, spill-related injuries to Pinnacles reefs have persisted for at least 4 years and may persist for much longer (Etnoyer et al. 2015). Some recovery of mesophotic reef fish may have already started to occur, as qualitative accounts from recent visits to the reefs have suggested an abundance of young, small, planktivorous fish. Planktivorous fish in the Pinnacles reefs have life spans ranging from 8 to 15 years, so while it might take on the order of a decade to fully restore the pre-spill population distribution, overall fish population numbers may be restored in less than a decade (Thurman 2004). However, recovery of different components of the mesophotic reef ecosystem will take differing amounts of time based on the different life cycles of species affected and ages of individuals killed. Although life spans for mesophotic corals in the Gulf of Mexico are not fully known, they are understood to have slower growth rates and longer life spans than their shallow-water counterparts. Researchers have documented coral ages between 23 and 100 years for Pacific black corals (Family: Antipathidae) at depths of 50–55 meters and approximately 100 years for gorgonian corals (Family: Plexauridae) at shallower depths of 20 meters (Grigg 1974; Roark et al. 2006).

Mesophotic reef sea fans therefore may take much longer to recover. Finally, the timeframe for recovery of unassessed organisms dependent upon the reefs is unknown.

4.5.6 Conclusions and Key Aspects of the Injury for Restoration Planning

Key Points

- In total, more than 770 square miles (2,000 square kilometers) of deep-sea benthic habitat (including soft-bottom and hardground) and 4 square miles (10 square kilometers) of mesophotic reef habitat on the continental shelf edge were injured as a result of the DWH oil spill. This area is more than 20 times the size of Manhattan or nearly two-thirds the size of the state of Rhode Island. This conclusion was based on a thorough foundation of documented pathway, exposure, and injury to benthic resources.
- There are potentially broader ecosystem impacts of quantified benthic resource injuries, based on the Trustees' understanding of the interconnectedness of the marine environment.
- Natural recovery of injured resources will take some time, and the pace may depend on the specific resource in question. Some resources, such as red crabs, may have already begun to recover, whereas deep-sea corals, with life spans in excess of 500 years, will certainly take much longer to recover.
- The Trustees identified a portfolio of restoration options to address these injuries, reflecting the range of substrate types across depths that have been shown to be injured.

Table 4.5-5 summarizes the key steps that the Trustees followed in their assessment of benthic resources, including documentation of pathway, exposure, and injury. It presents the quantified extent and degree of losses based on amounts and locations of benthic habitats and communities that were affected by the spill. The various types and amounts of documented losses are translated where possible into broad categories of impact based on the types, extent, and duration of change of community functionality across a seafloor habitat footprint and duration. These losses of public resources are expressed in units of area (square kilometers). In developing these estimates, the Trustees undertook and consulted numerous studies, used multiple lines of evidence, and relied on expert opinion to assert the quantified losses from the DWH oil spill experienced by northern Gulf of Mexico benthic marine resources. Potential restoration options are described as part of the restoration volume of this Programmatic Damage Assessment and Restoration Plan.

The Trustees recognize that integrating all benthic losses into a single quantitative unit representing habitat degradation and community loss has inherent uncertainty in the assumptions used. For example, some losses to species were absolute, such as death, whereas other losses to habitat represent degradation of quality through fouling, contamination, and loss of structure. Where possible, zones of injury were defined, as detailed above.

Specifically, as summarized in Table 4.5-5, the injury assessment showed that:

- A footprint of injury to benthic habitat was confirmed in the deep sea around the wellhead encompassing over 2,000 square kilometers.
- Based on the assessment of benthic natural resources over the past 5 years by the Trustees, more than 2,000 square kilometers of deep-sea benthic habitat (including soft-bottom and hardground) and 10 square kilometers of mesophotic reef habitat on the continental shelf edge were injured as a result of the DWH oil spill. This is greater than 20 times the size of Manhattan or nearly two-thirds the size of the state of Rhode Island.
- A significantly larger area of uncertain exposure and injury exists outside these areas (Figure 4.5-22). Approximately 8,500 square kilometers of potential exposure extends beyond and between the areas where the Trustees have quantified injury. Many pelagic resources, such as grouper, use both reef top and surrounding habitats for feeding.

The Trustees considered all of these aspects of the injury in restoration planning, and also considered ecosystem effects and recovery information.

Table 4.5-5. Summary of benthic losses.

<table>
<tr><th>Benthic Area</th><th>Pathway</th><th colspan="2">Exposure</th><th>Injury Determination</th><th>Quantification</th></tr>
<tr><td>Deep Benthos >800 m deep</td><td>Direct contact with deep-sea plume.
Contaminated marine snow.
Dissolved oil, droplet oil, and dispersant in water column.</td><td colspan="2">Oiled sediment.
Oil and dispersant on and around corals.
Contaminated marine snow.
Drilling muds and other debris.</td><td>Reduction of sediment faunal diversity.
Reduction of sediment faunal abundance.
Mortality of corals.
Contamination of benthic megafauna tissues.
TPAH50 exceedance of LC20 and LC50 modeled mortality.</td><td>Injury
Zone 1—28 km^2
Zone 2—195 km^2
Zone 3—793 km^2
Zone 4—1,275 km^2
Total
2,291 km^2
Additional uncertain exposure and injury
2,600 km^2</td></tr>
<tr><td>Continental Slope 200–800 m deep</td><td>Contaminated marine snow.
Dissolved and droplet oil and dispersant in water column.</td><td colspan="2">Contaminated marine snow.
Direct exposure to oil in the water column.</td><td>Overlap of surface dispersant application, surface oiled area, and floc detected on benthic sediment.
Additional injury to slope sediment suggested by independent academic research.
Additional injury to slope fish resources suggested by independent academic research.</td><td>Uncertain exposure and injury
3,300 km^2 with some confirmed exposure but predominantly uncertain exposure and uncertain injury.
Overall injury not quantified.</td></tr>
<tr><td rowspan="2">Continental Shelf ~10–200 m deep</td><td rowspan="2">Contaminated marine snow.
Entrainment of surface oil.
Dissolved and droplet oil and dispersant in water column.
Pathway of oil to nearshore limited to geographic areas with</td><td>Mesophotic reef resources</td><td>Contaminated marine snow.
Direct exposure to oil in the water column.</td><td>Reduction of fish populations.
Coral mortality and injury.</td><td>Mesophotic reef top injury
10.4 km^2 impacted.
Mesophotic reef hash potential injury
249 km^2</td></tr>
<tr><td>Shelf sediment resources</td><td>Exposure of sediment to oil minimal; TPAH50 levels very low where measured.</td><td>Overlap of surface dispersant application, surface oiled area, and floc detected on benthic sediment.
Additional injury to shelf sediment not determined.</td><td>Uncertain exposure and injury
3,300 km^2 of shelf with some confirmed exposure but predominantly uncertain exposure and uncertain injury.
Injury not quantified.</td></tr>
</table>

Benthic Area	Pathway	Exposure		Injury Determination	Quantification
	transiting surface slicks.	Soft coral hardground	Exposure suggested by independent academic research.	Injury to soft coral hardground suggested by independent academic research.	Injury not quantified.
		Nearshore hard coral reefs near Florida	Exposure not assessed.	Injury to nearshore coral reefs not determined.	Injury not quantified.
Nearshore Benthos	Covered in Section 4.6, Nearshore Marine Ecosystem				

4.5.6.1 Ecosystem Effects

As noted in the introduction to this section, the dividing lines that humans ascribe to habitats within the larger marine system are not absolute. In fact, certain biota are known to thrive at the edges of habitats or transition zones between habitat types. Nevertheless, the Trustees have identified and described the losses detailed herein, and pursuant to their requirements under the Oil Pollution Act have quantified the resource injuries listed above across two broad habitat types. However, as discussed in Section 4.5.5, Injury Quantification, these quantified injuries have the potential to adversely affect the larger marine ecosystem.

Because of the overall scope of the spill and logistical considerations, some uncertainty related to resource exposure and injury outside of these quantified areas exists. In some cases, the potential broader adverse implications are well understood. For example, given the role of the benthos in recycling nutrients and carbon up through the food web, resource injuries across vast areas of the sea floor, such as those observed, have the potential to lead to larger ecosystem perturbations up through the food web. These may or may not have been fully captured by the larger natural resource injury assessment. In other cases, as with deep-sea hardground habitats, the inhabitants and ecological functions are less well understood, and the larger ecosystem implications of observed injuries are also less well understood. Nevertheless, the Trustees relied on scientifically defensible data and information to describe and quantify benthic resource losses and understand that estimated resource injuries may not fully capture some of the broader ecosystem implications of the losses.

4.5.6.2 Recovery

As described in Section 4.5.5.5, the time required for natural recovery of benthic resources will likely depend on the specific resource in question. Further, there is uncertainty in recovery trajectories, particularly for some of the longer lived benthic resources. Some resources, such as red crabs, may have already begun to recover. By contrast, other resources, such as deep-sea corals with life spans in excess of 500 years, will certainly take much longer to recover.

4.5.6.3 Restoration Considerations

As described in Chapter 5 (Section 5.5.13), the Trustees have identified a portfolio of restoration to address these injuries, reflecting the range of substrate types across depths shown to be injured, while at the same time acknowledging that limited experience with restoration for some of these rare deep-sea benthic habitats will require the restoration to proceed with careful monitoring and adaptive management of restoration implementation.

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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* (DWH) oil (and response activities)?
- **Estuarine Coastal Wetlands Complex Injury Assessment (Section 4.6.4):** How were coastal wetlands and associated species affected by DWH oil and response activities? What was the magnitude of the injury?
- **Subtidal Oyster Assessment (Section 4.6.5):** How were subtidal oysters affected by DWH 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 DWH 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 DWH oil and response activities? What was the magnitude of the injury?
- **Submerged Aquatic Vegetation Assessment (Section 4.6.8):** How were submerged aquatic vegetation and associated species affected by DWH 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 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 (Nixon et al. 2015a).

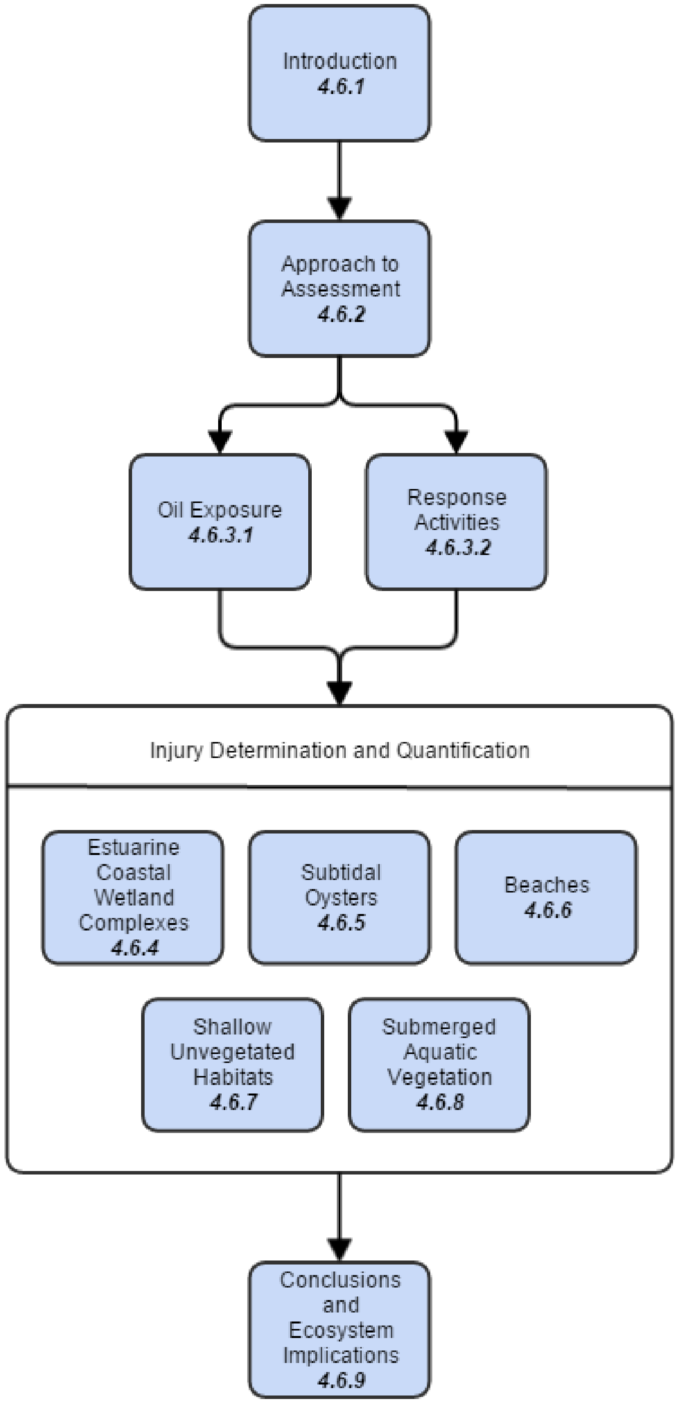
Oiling caused multiple injuries to marsh habitats, including reductions in aboveground biomass and total plant cover in mainland herbaceous salt marshes, reductions in periwinkle snail abundance, reductions in shrimp and flounder growth rates, reduced reproductive success in forage fish, reduced amphipod 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 Historical 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.

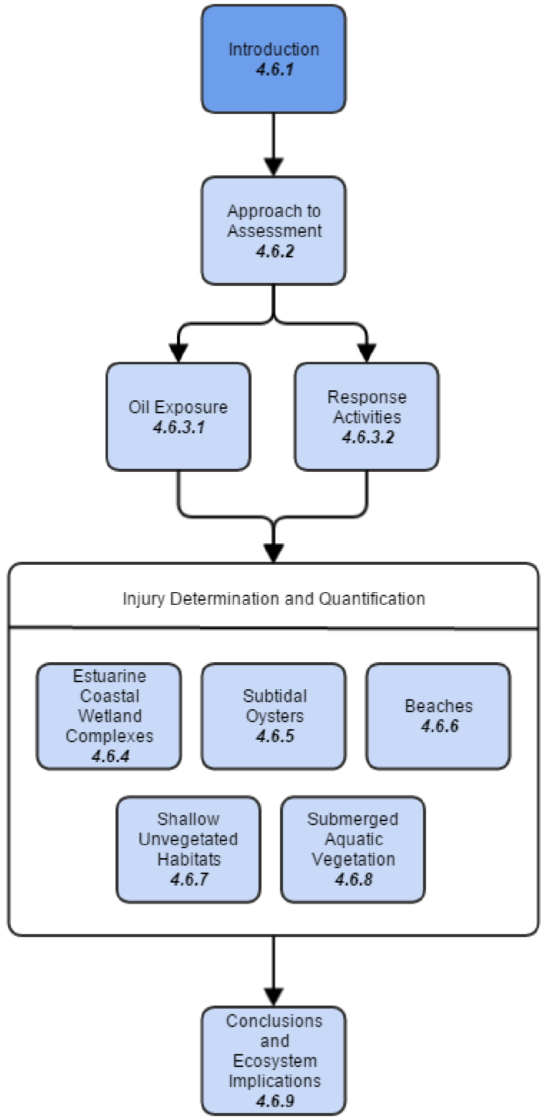
Addressing injuries to these marsh habitats will require special attention. Gulf salt marshes are productive because of their intricate complexity. Sinuous tidal channels that maximize edge habitat provide fauna access to flooded marsh surfaces for refuge and forage and promote rapid growth of juvenile 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.



4.6.1 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 DWH 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).

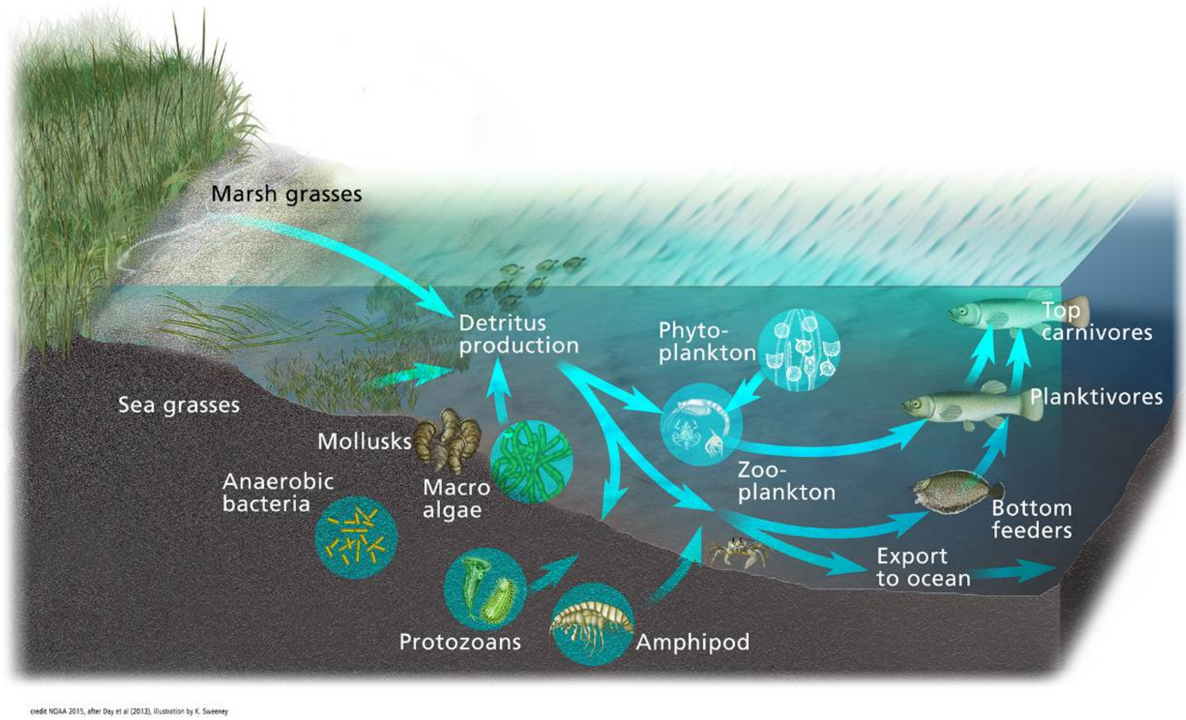
In addition, because of their unique ecological importance, many of the Gulf's habitats are federal trust resources and are protected as national parks, seashores, and wildlife refuges. These federal trust resources include various habitats (e.g., coastal wetlands, marsh, SAV, beaches, sand dunes) that 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; 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.

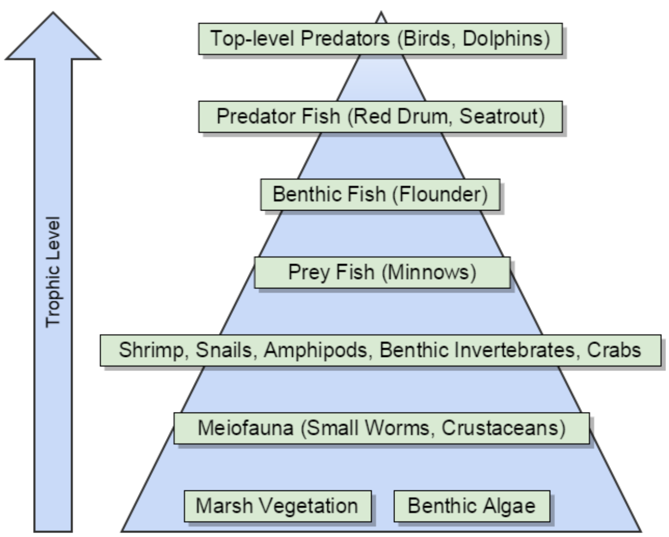


Figure 4.6-2. Simplified trophic pyramid for salt marsh species in the northern Gulf of Mexico. Primary producers such as marsh vegetation and benthic algae form the base of the nearshore food web, providing nutrients to other organisms, as well as habitat. Injuries to marsh vegetation initiate a cascade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

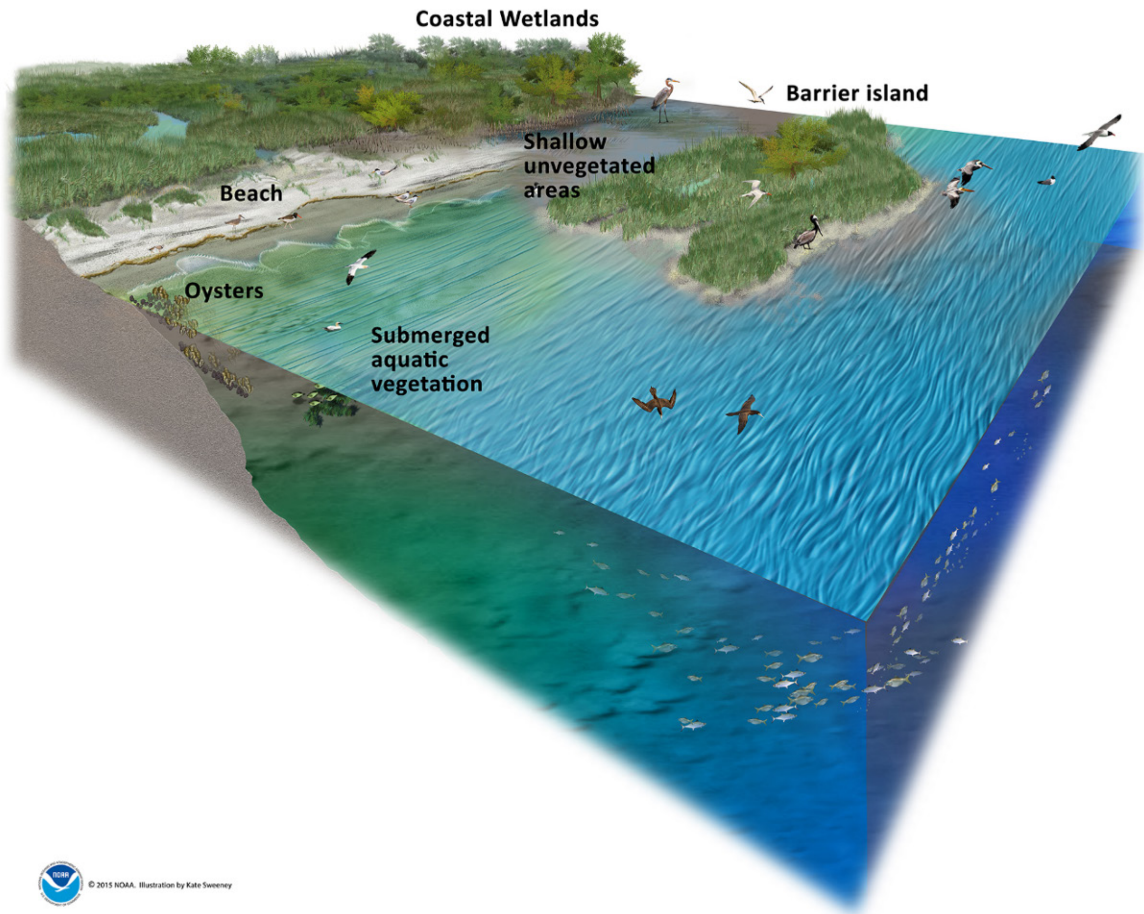
Another type of coastal wetland habitat in the northern Gulf of Mexico is the mangrove-salt marsh complex, which was evaluated in Louisiana. In this habitat, mangroves exist at the northern limit of their range in "stunted" form. Mangrove habitats are primarily composed of a mixture of black mangrove (*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 the 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 2015b).

4.6.1.2.2 Oyster Reefs

Oysters in the northern Gulf of Mexico form nearshore and subtidal reefs composed 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; 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, fish, 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.

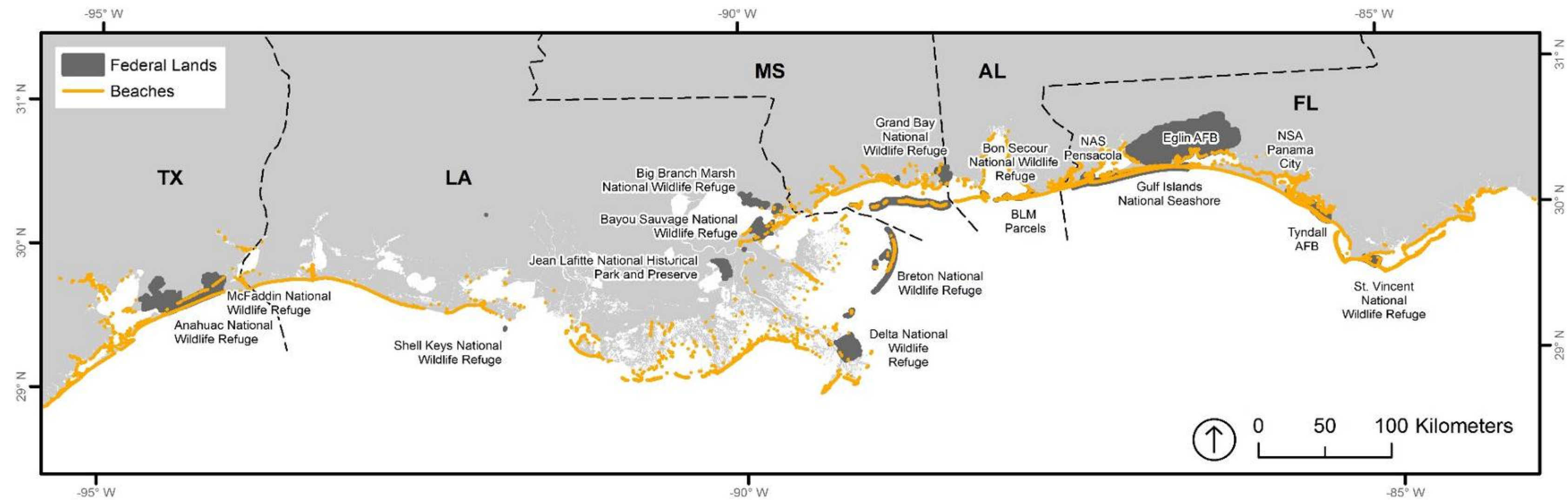
Northern Gulf of Mexico sand beaches and dunes are home to numerous organisms, including small crabs, clams, shrimp, and snails. These organisms live and feed on and within the sand and beach wrack (i.e., decomposing vegetation washed up on the shore by the surf). These small organisms in turn serve 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, 2003).



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

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).

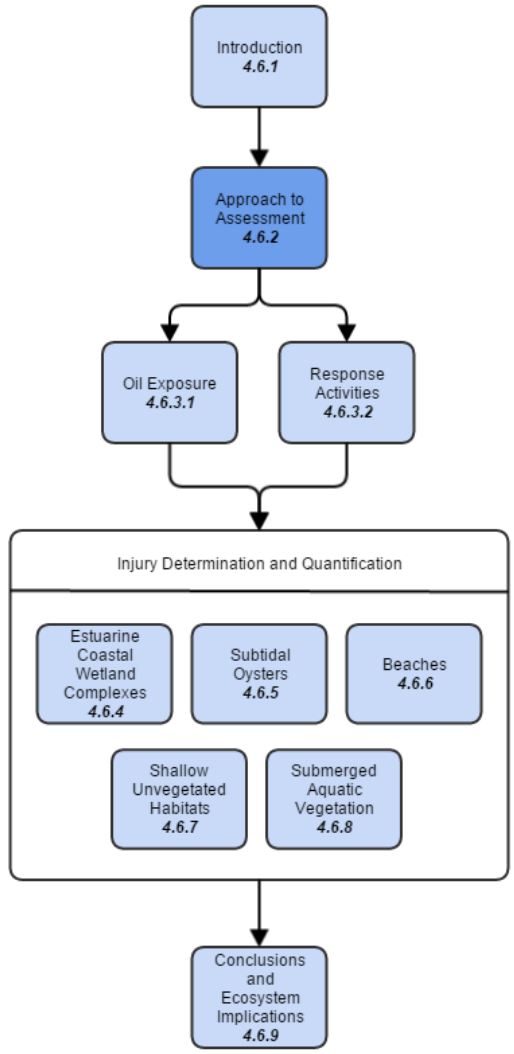
Freshwater SAV is a particularly important resource at the Barataria Preserve, a unit of the Jean Lafitte National Historical Park and Preserve in Louisiana (Poirrier et al. 2010). For several reasons, the seagrass beds inside the Chandeleur Islands are unique: they are the only existing marine seagrass 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 DWH 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 DWH 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) provide 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 DWH 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; boom were stranded 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 DWH response and the Natural Resource Damage Assessment (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 composed 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.

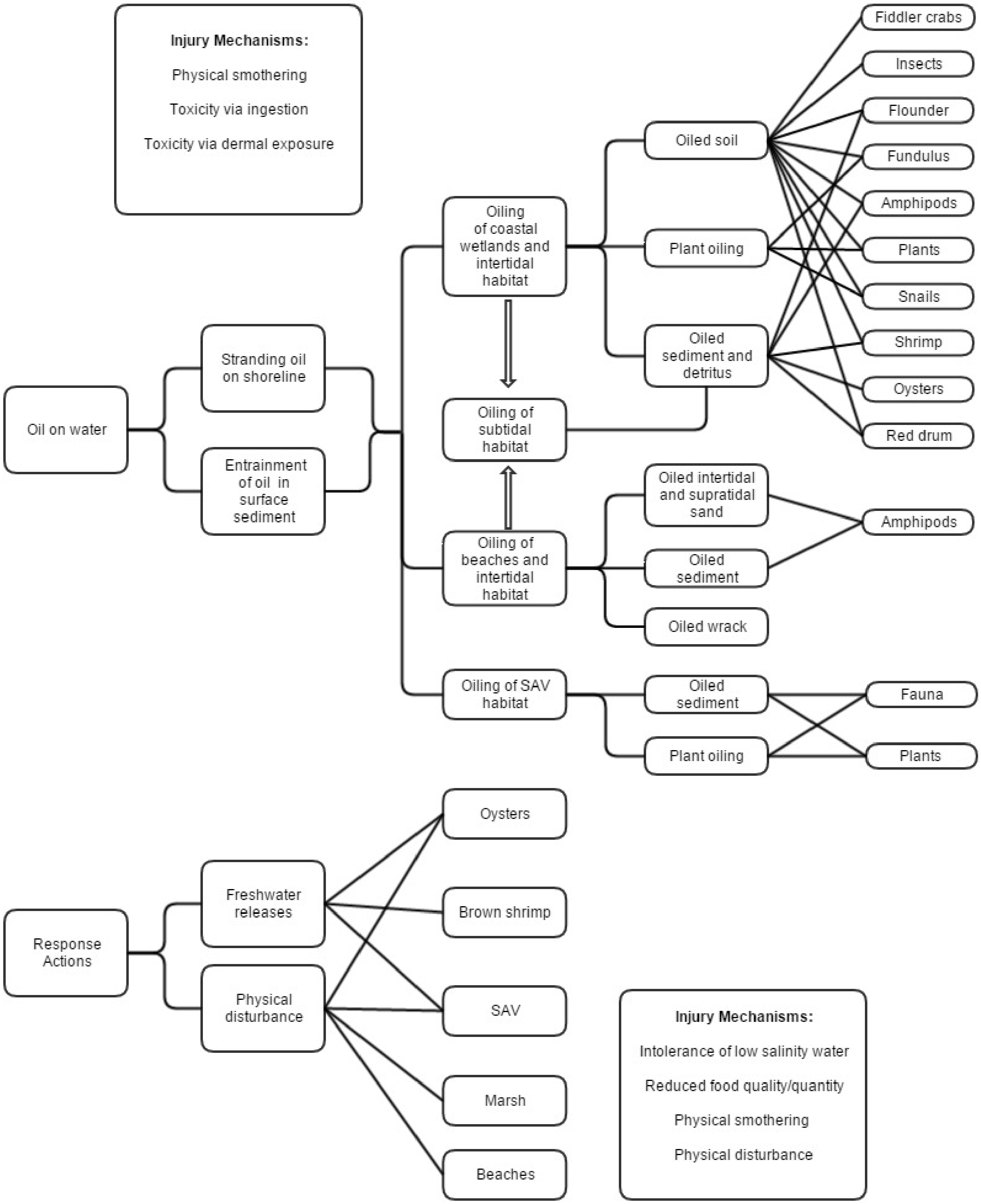


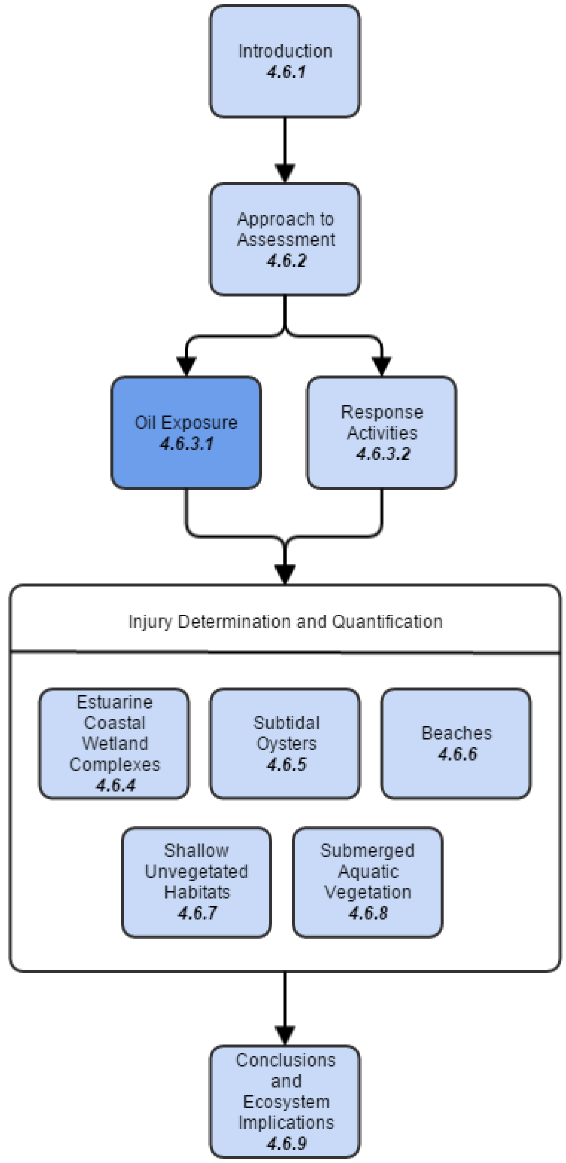
Figure 4.6-5. Pathways of exposure of representative species to oil and response actions and mechanisms of injury. The diagram illustrates the complexity of the interactions among the oil and response actions and the nearshore resources evaluated. Most resources were exposed via multiple pathways.

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 to 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.

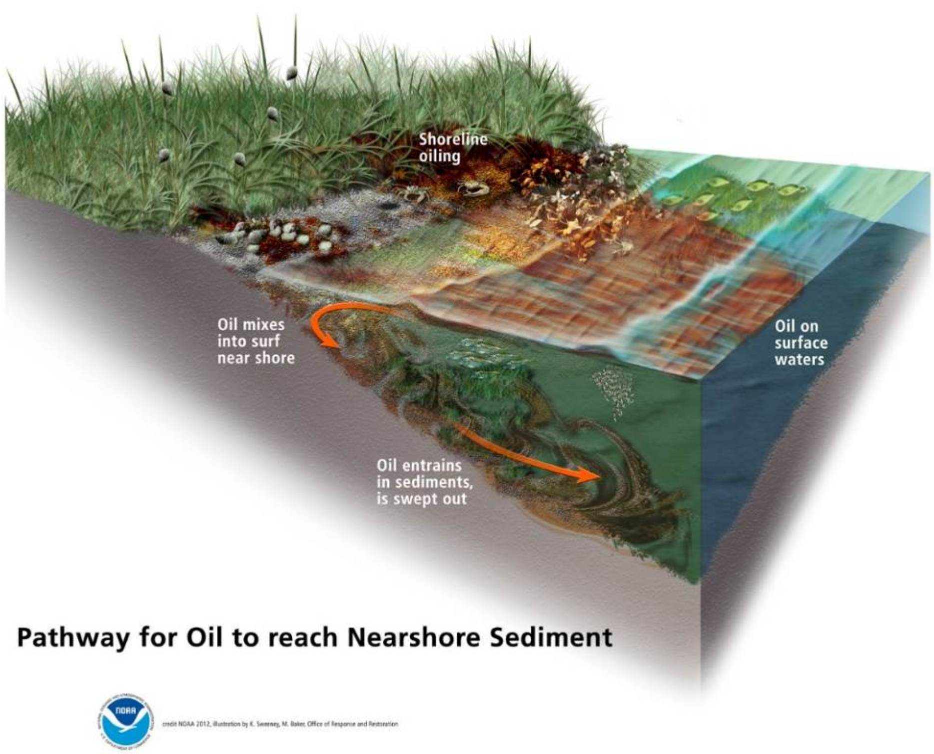


4.6.3.1.1 Pathways

As described in Section 4.2 (Natural Resource Exposure), some portion of the oil that reached the 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 U.S. 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 dataset was supplemented with other available

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 DWH 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.

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 (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) (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); 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) (Nixon et al. 2015b).

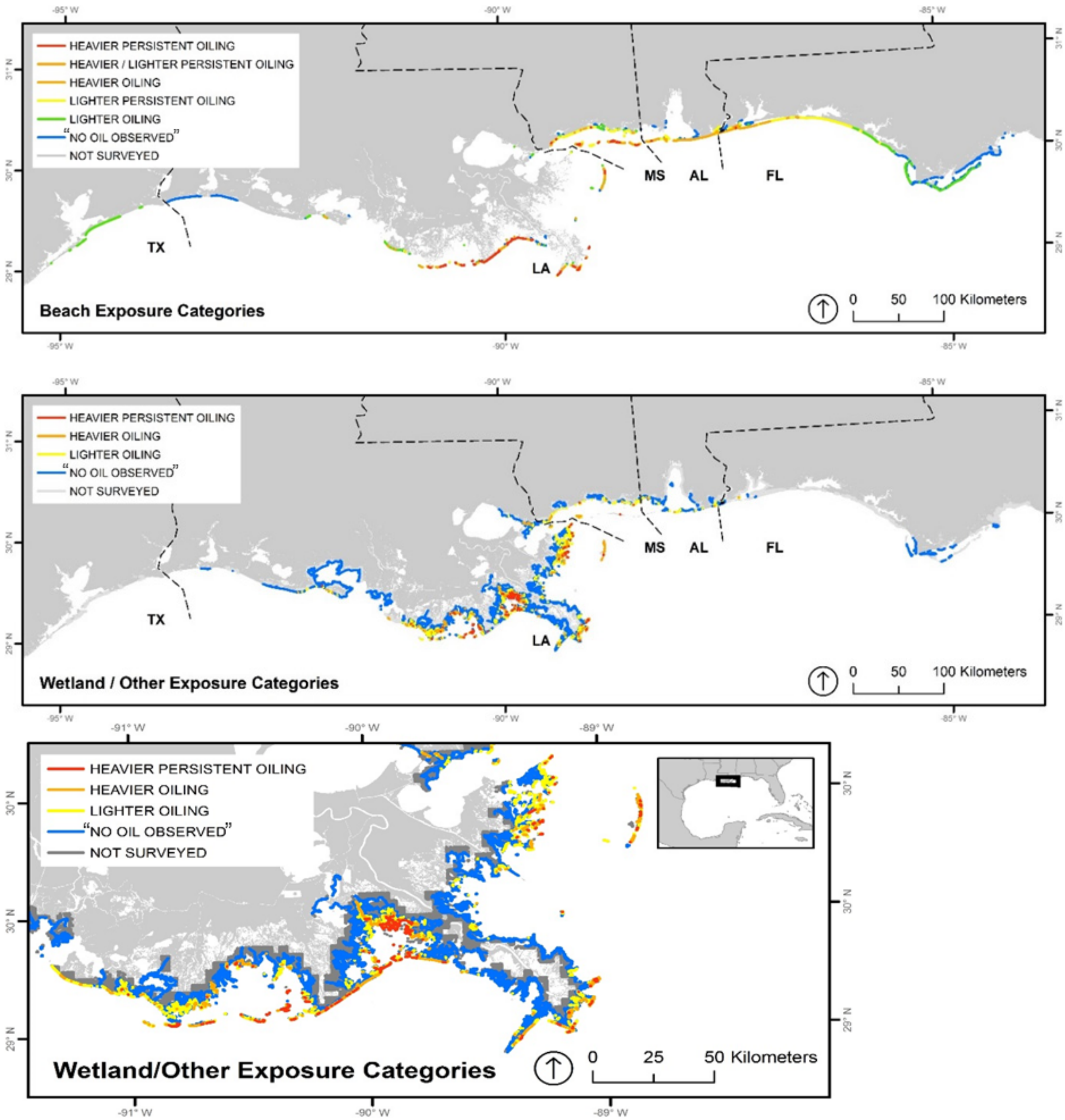
Why Was Oil Found at Locations Designated as "No Oil Observed"?

"No oil observed" is a shoreline category intended to describe areas where oiling was not observed during linear shoreline surveys. The 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[a]	Not surveyed	Not surveyed
"NO OIL OBSERVED"[a]	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

[a] "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: *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) (Nixon et al. 2015b).

Exposure Category		NO OIL OBSERVED[a]	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)

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

Table 4.6-3 provides additional detail by presenting lengths of shoreline oiling for different coastal wetland habitats (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 composed 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) (Nixon et al. 2015b).

Table 4.6-3. Lengths of shoreline oiling for coastal wetland habitat types by state and oiling category (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) (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 (Nixon et al. 2015b). As mentioned above, some shoreline locations characterized as “no oil observed” exhibited plant oiling; 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; 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 (Nixon et al. 2015a). This represents 1.6 percent of the 6,178 kilometers (3,839 miles) of 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; 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; 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 DWH spill were not sufficient to compute comparable ambient TPAH50 concentrations (Zhang et al. 2015a). Thus, a forensic-based approach 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

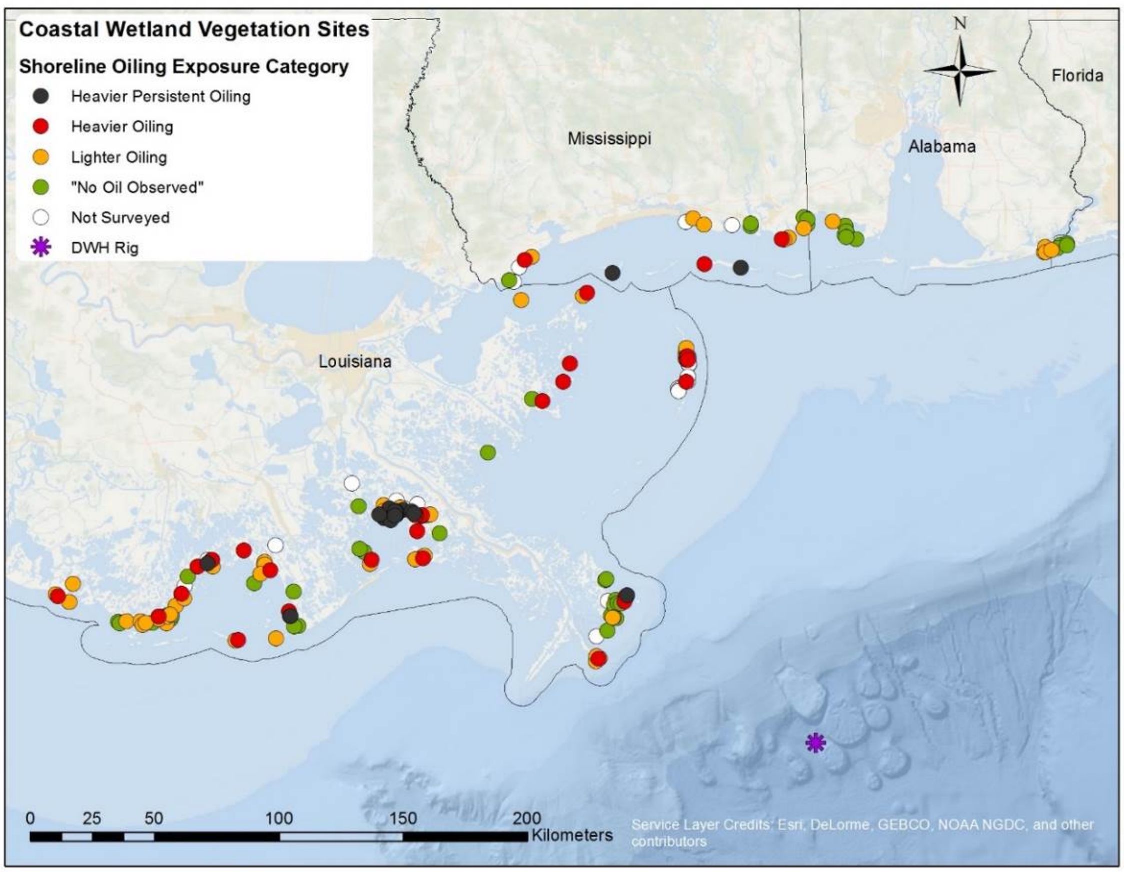
Table 4.6-6. Ambient TPAH50 concentrations in nearshore sediments adjacent to shorelines in the northern Gulf of Mexico. Sediment concentrations were highest adjacent to the Mississippi River Delta and lowest adjacent to the coastal wetlands of Louisiana's barrier islands (Zhang et al. 2015a).

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

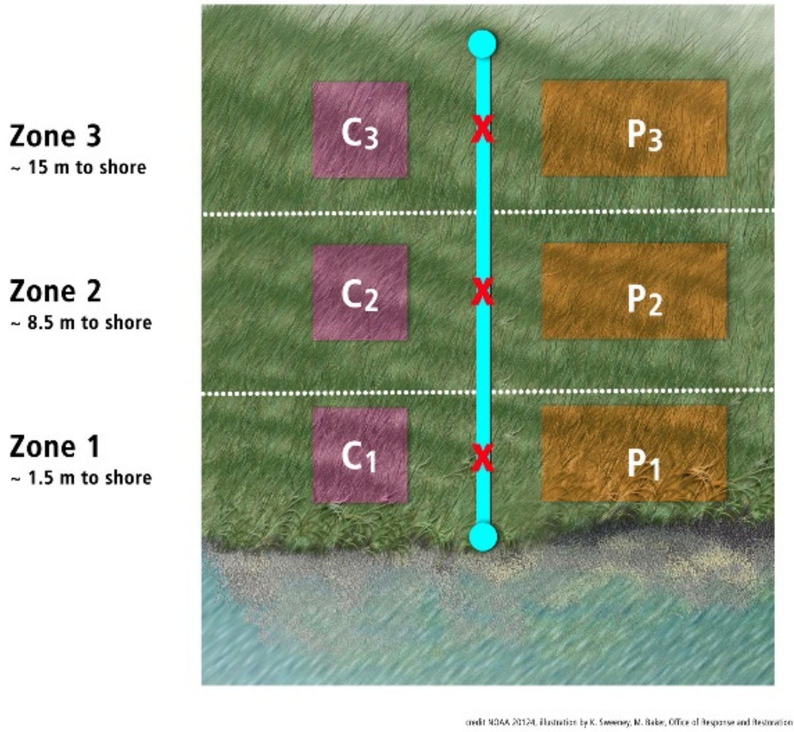
From 2010 to 2013, soil samples for TPAH50 analysis were collected in coastal wetlands across Louisiana, Mississippi, and Alabama (Figure 4.6-10).

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 2015a; 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 (Oehrig et al. 2015a). 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: *Oehrig et al. (2015a).*

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 to 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).

State	Habitat	Shoreline Exposure	Soil TPAH50 Concentrations (ppb)									
			Fall 2010		Spring 2011		Fall 2011		Fall 2012		Fall 2013	
			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

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[a] 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

[a] 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

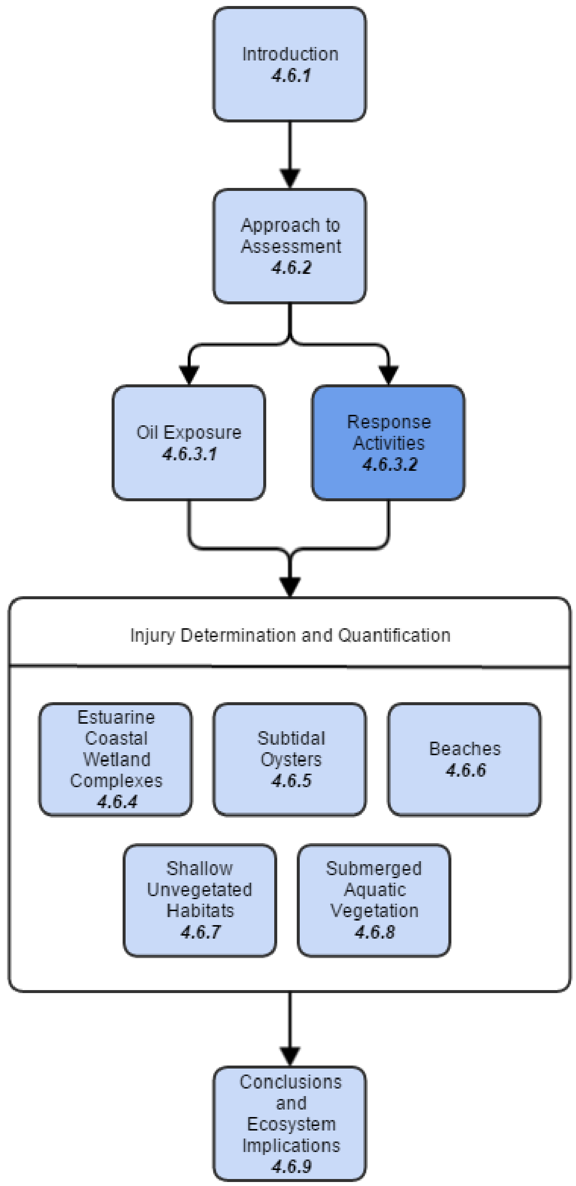
Table 4.6-10. Summary statistics of 2011 nearshore (within 500 meters) sediment TPAH50 concentrations along unvegetated shorelines. No relationship was observed between concentrations and degree of oiling (Zhang et al. 2015a).

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 eight 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 (Oehrig et al. 2015b).

Surface Water Oiling

The Trustees' assessment of water chemistry demonstrates that DWH 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 DWH 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:

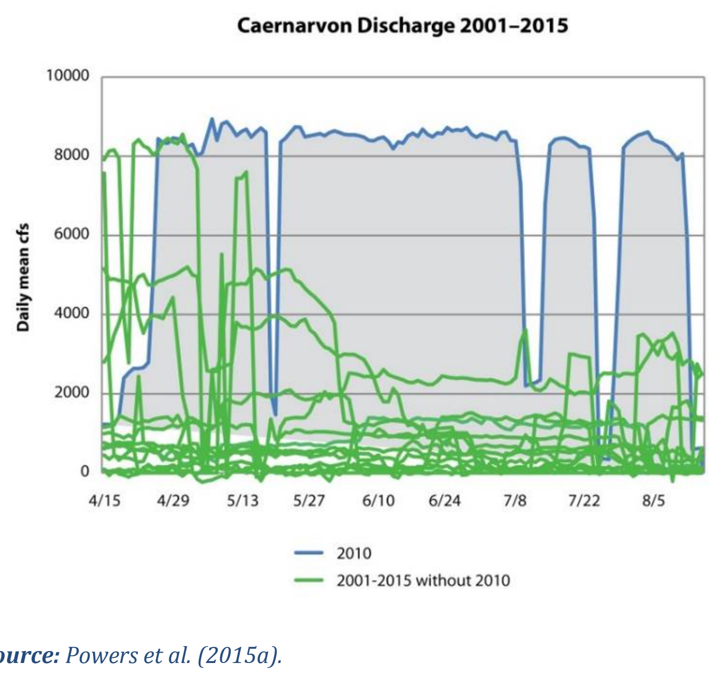
- **Manual treatment by response crews** using hand tools and vehicles to transport crews and wastes.
- **Augering** to search for buried oil.
- **Sifting** to separate oil from sand and remove it.
- **Tilling** to break up oil and expose it to air, with the expectation that this would accelerate biodegradation.
- **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; 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

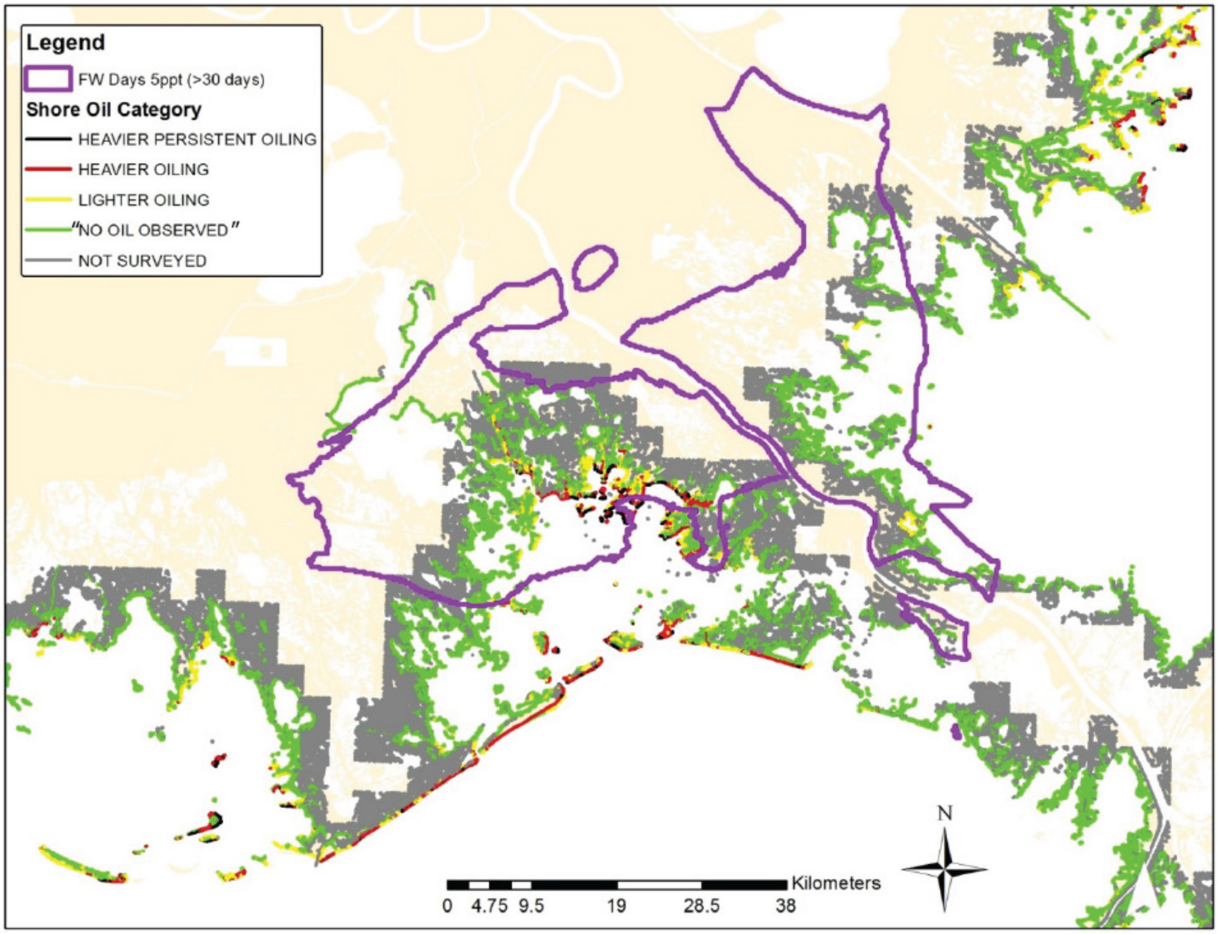
With oil approaching the shoreline, salinity control structures at nine separate locations in Louisiana (Davis Pond, Caernarvon, Bayou Lamoque, West Pointe a la Hache, Violet Siphon, White Ditch, Naomi Siphon, Ostrica Lock, and Bohemia) were opened as part of a series of response actions intended to 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 DWH 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 was significantly higher than discharge rates in the other years shown.



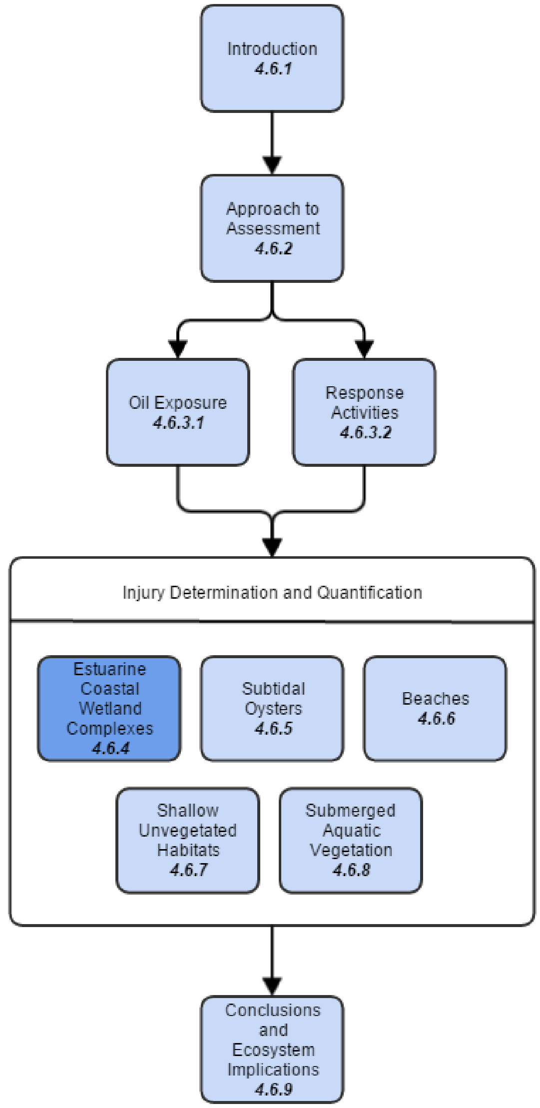
__Source__: Rouhani and Oehrig (2015b).

Figure 4.6-14. Spatial extent of impacts of the summer river water releases in response to the approaching DWH 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 DWH 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 DWH incident. It 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 DWH incident on coastal wetlands (Section 4.6.4.2), and a summary of this assessment's findings (Section 4.6.4.3). This 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 impaired 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.

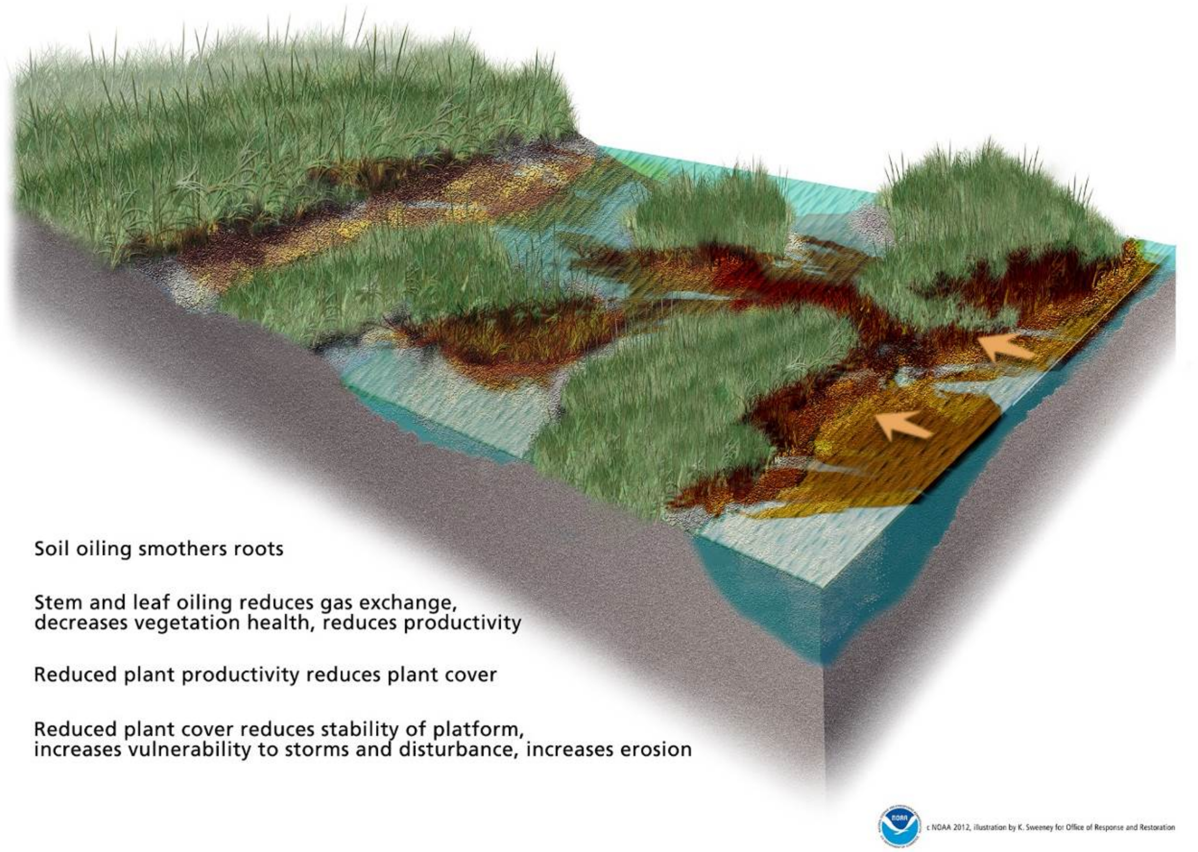
A major concern for Gulf of Mexico wetlands is whether sustainability and resistance to future disturbances have been reduced due to the DWH spill. These wetlands, particularly those in the Mississippi River Delta, remain viable only if their rate of elevation change (from both mineral sediment deposition and organic matter accumulation) 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 disturbances that result 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 DWH 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; 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 DWH 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.

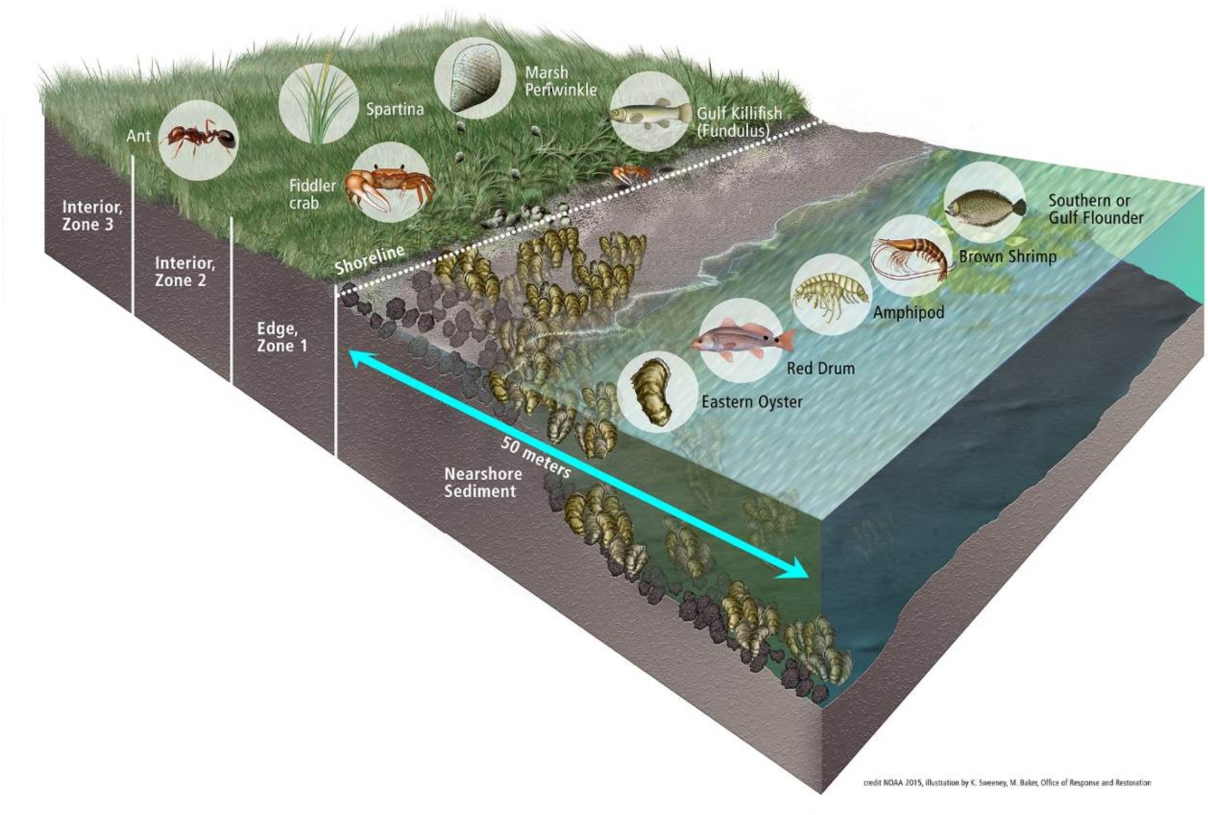
Pathway for oil to reach marsh surface



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 DWH 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 DWH 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 to 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 DWH 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 2015b; 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 2015b; 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

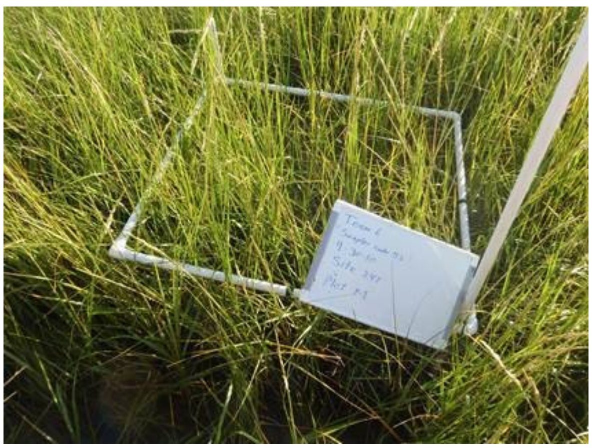
Numerous habitat types and regions suffered adverse effects to coastal wetland vegetation. Key measures of the health and productivity of the vegetative community included live aboveground biomass, vegetation cover, and the degree of chlorosis.

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 aboveground 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).

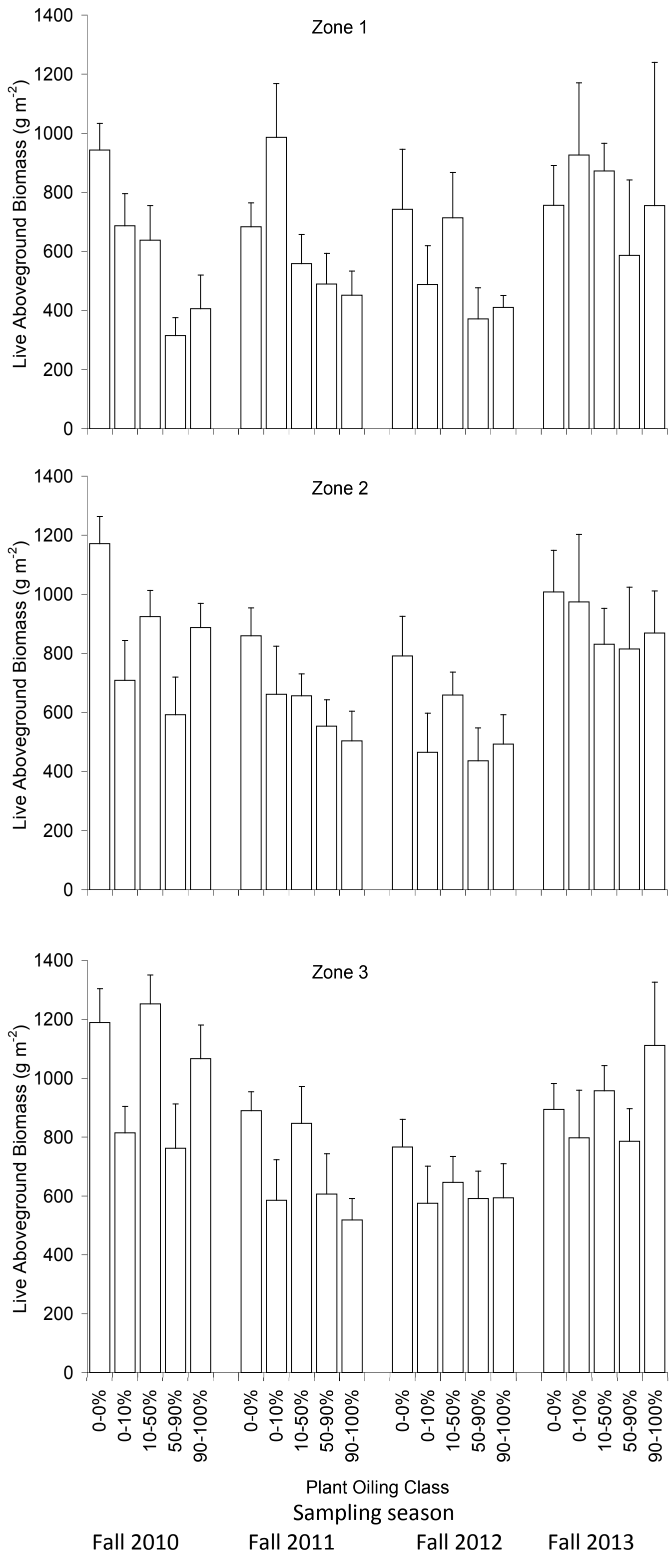
Studies of effects of DWH oiling on coastal wetlands were also conducted by researchers not involved in the NRDA. Results from these studies generally confirm the NRDA study findings, including reductions in aboveground biomass, plant cover, stem density rhizomes, and photosynthetic functioning in heavily 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 composed 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 impaired 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 fall 2011 (Willis & Hester 2015a). Further, the physiological health of mangroves, as indicated by visual estimation of chlorosis, was found to be diminished in fall 2011 due to plant oiling. This apparent delay in impacts reflects several factors. The initial (fall 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 vegetation 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 from quantifying 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 2015a). 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 fall 2011 (Hester & Willis 2015b). 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 DWH 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 (Oehrig et al. 2015a). 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; Oehrig et al. 2015a). 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 (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 (Nixon et al. 2015a).

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

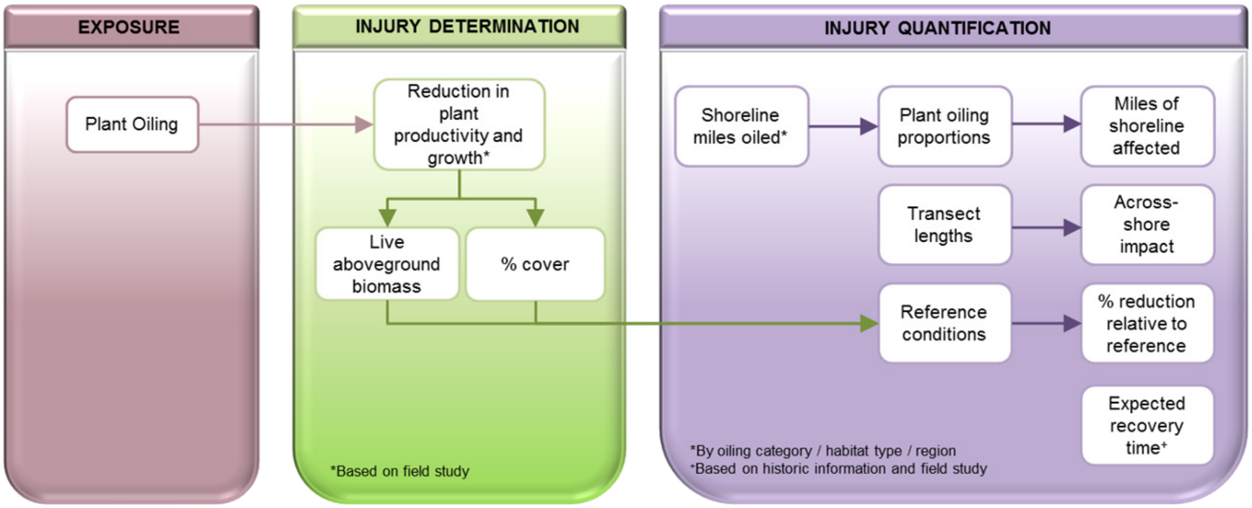


Figure 4.6-20. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to coastal wetland plants occurred. Plant aboveground biomass and percent cover at oiled sites were compared to reference sites to develop 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; 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 (Oehrig et al. 2015a). 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 382 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.

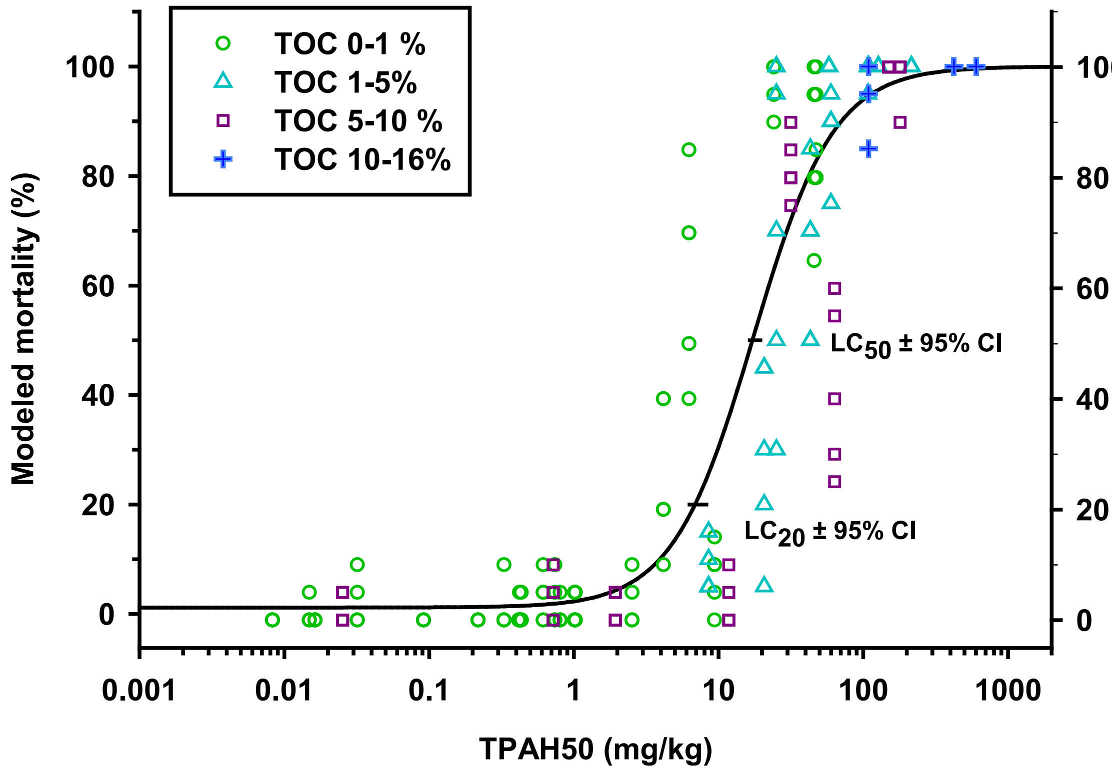
shrimp, *Fundulus*, and flounder (Fry et al. 2003; McTigue & Zimmerman 1998). The Trustees evaluated injury to *L. plumulosus* as a representative of sensitive animals that burrow in soils and sediment of mainland herbaceous and back-barrier salt marshes, mangroves, and Delta *Phragmites* habitats.

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 36 to 95 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 95 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 2010 and fall 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.

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, and 2) 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).



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 7.2 parts per million, and 50 percent die at a concentration of 17.9parts per million.

The quantity of amphipods removed from the marsh ecosystem is estimated at 382 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 the following: 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 7.2 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

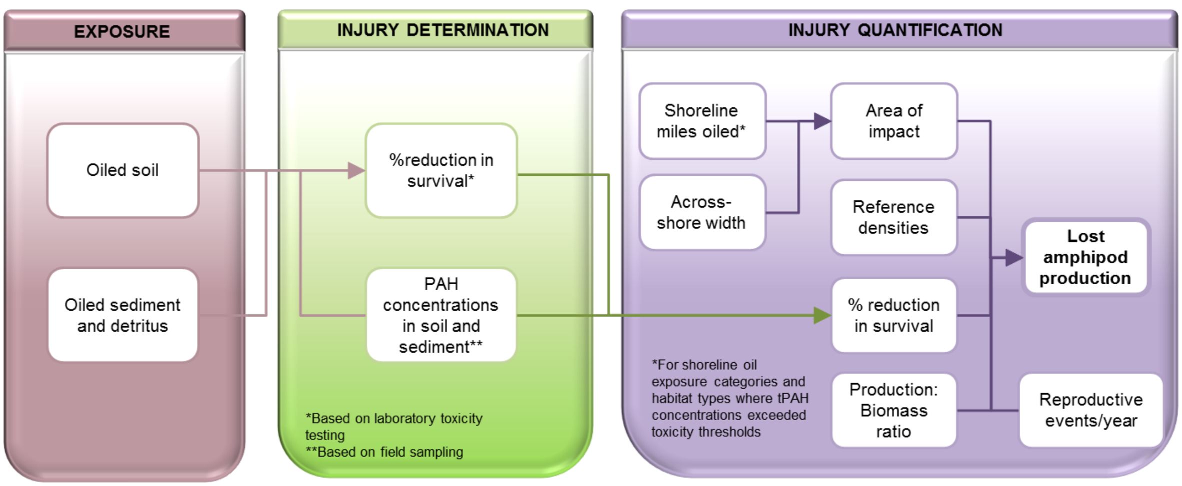


Figure 4.6-22. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to amphipods occurred. Concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-21) to determine the reduction 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-sized range of snails is not expected until at least 2021.

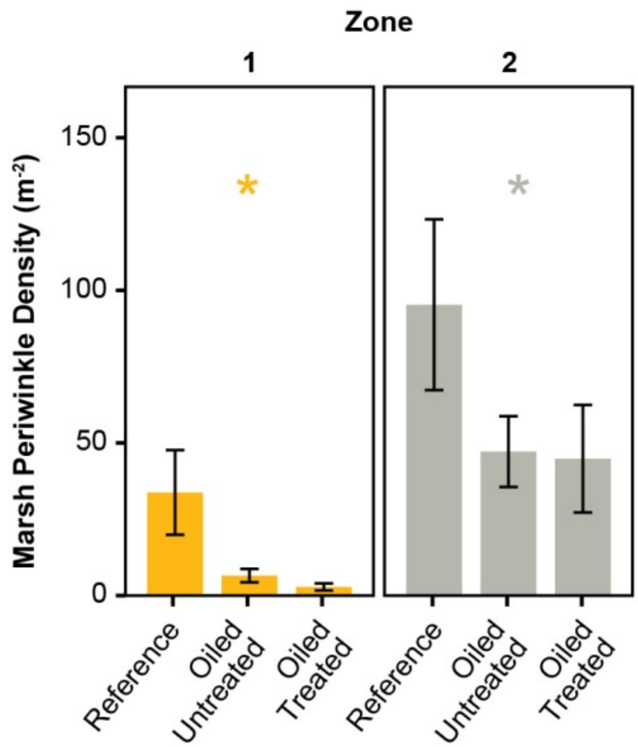
Marsh periwinkles (*Littoraria irrorata)*, a salt marsh snail, are widely distributed, abundant, and conspicuous grazers on algae and fungi that grow on the stems and leaves of marsh plants and on soils (Montague et al. 1981; Subrahmanyam et al. 1976). Through their grazing activities, marsh periwinkles 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; 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 .

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 1 to 2 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).

Other studies of salt marsh habitats oiled by the DWH 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 4.6-24. 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 bind 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).

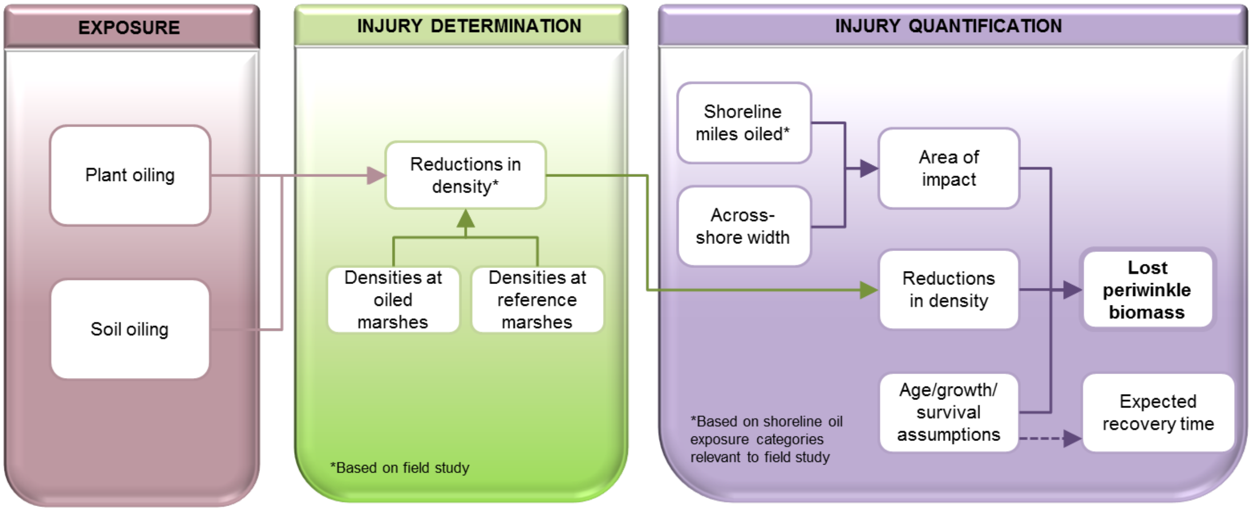
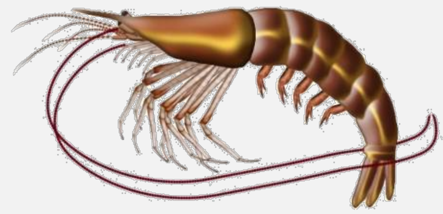


Figure 4.6-24. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to periwinkles occurred. Measured reductions in snail numbers in each zone (compared to reference sites) and the length and width of each 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 heavy 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, and 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



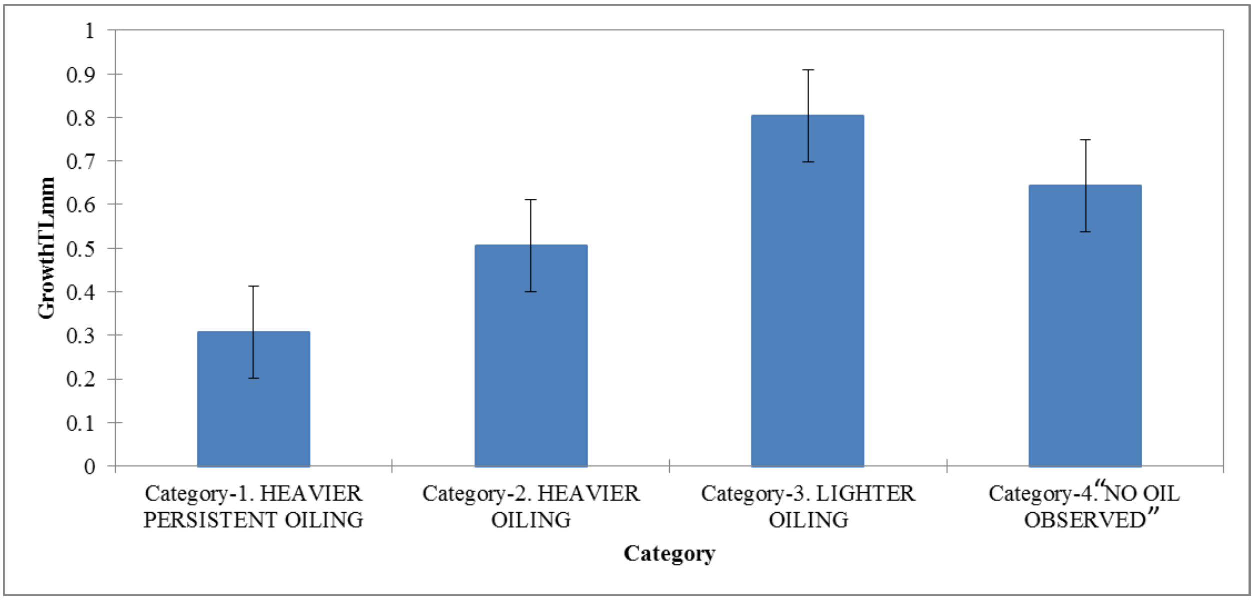
- White shrimp and brown shrimp are important components of the Gulf of Mexico ecosystem and food web and support a robust commercial fishery.
- Juvenile penaeid shrimp growth was dramatically affected by oiling.
- The total loss of shrimp production over 2010 and 2011 due to oiling is estimated at 2,089 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 1991).

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 (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 4.6-25). 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 4.6-25. 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 freshwater 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 DWH (i.e., "Oil Spill" scenario). The results of Rose et al. (2014) suggest that the effects demonstrated for brown shrimp likely extend to other fauna.

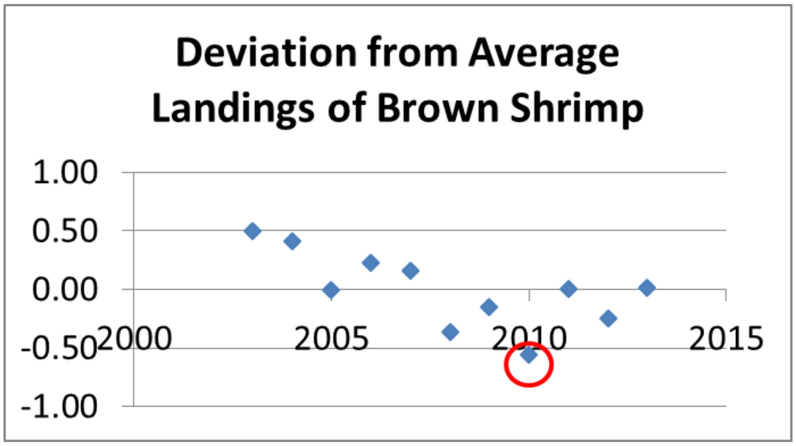
Injury Quantification

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: 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 4.6-26).

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 4.6-26. 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 4.6-27.

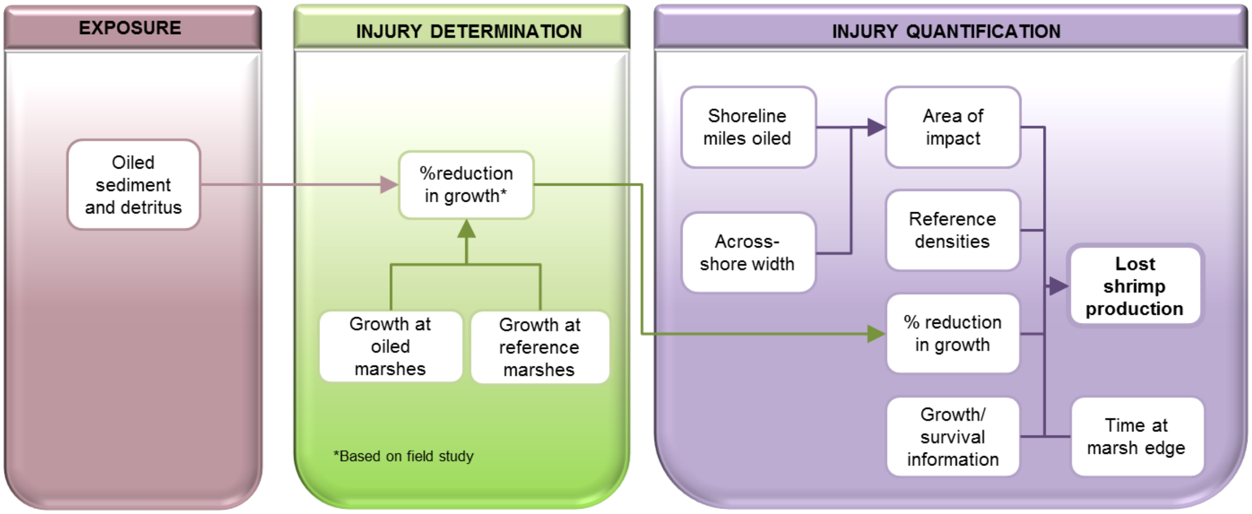


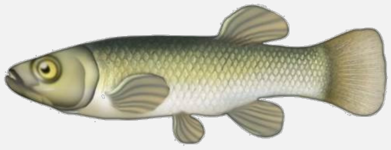
Figure 4.6-27. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to shrimp occurred. Measured reductions in shrimp growth in each shoreline oiling category (compared to unoiled reference sites), the length of each shoreline oiling category, and the width of the area used by juvenile shrimp are used to calculate reduction in shrimp growth. Literature values for baseline densities, growth, and survival are used to quantify the weight of lost shrimp over the time that effects were observed.

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 DWH 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 1991; 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.

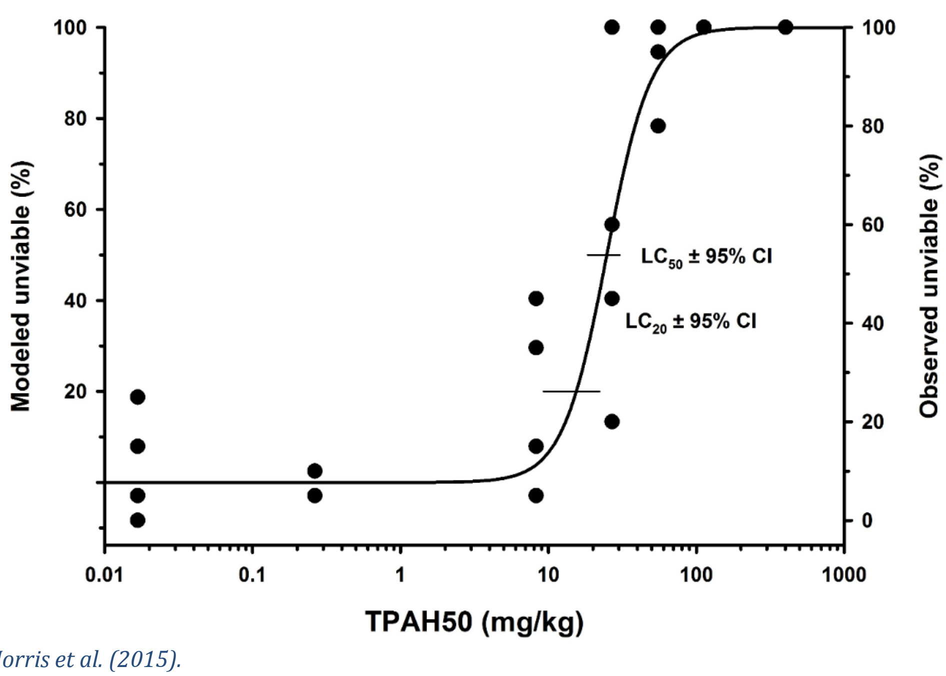


- Marsh soil conditions at heavier persistently oiled Louisiana mainland herbaceous salt marsh reduced successful hatching of *Fundulus* eggs by 68 to 99 percent when compared to conditions where no oil was observed.
- An estimated total of 84.7 MT wet weight of *Fundulus* was lost where soils exceeded 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 4.6-28). 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 2010 and fall 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 4.6-28. 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 DWH 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 with 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 4.6-29.

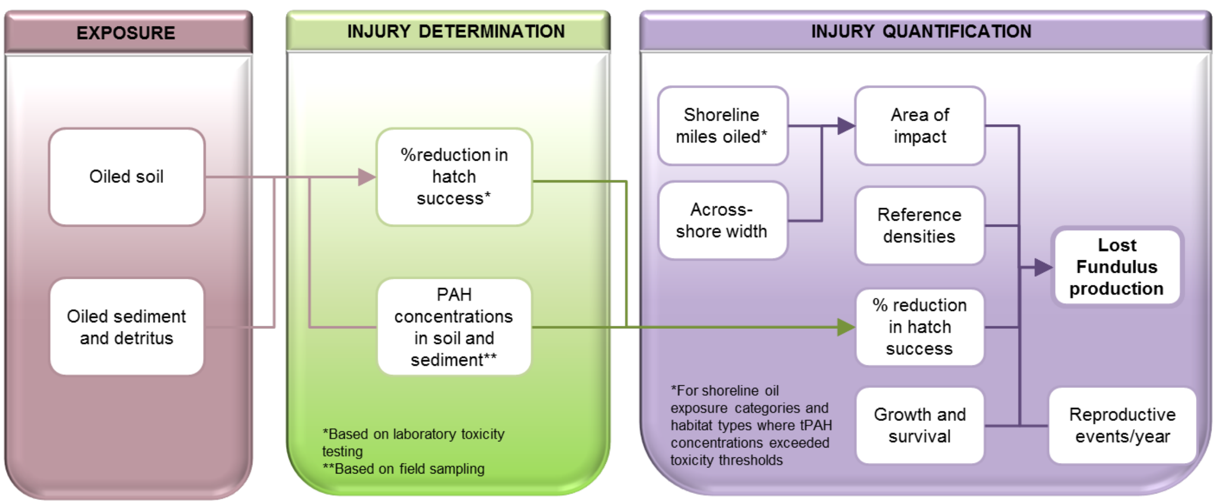


Figure 4.6-29. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to *Fundulus* occurred. Concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-28) to determine the reduction 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

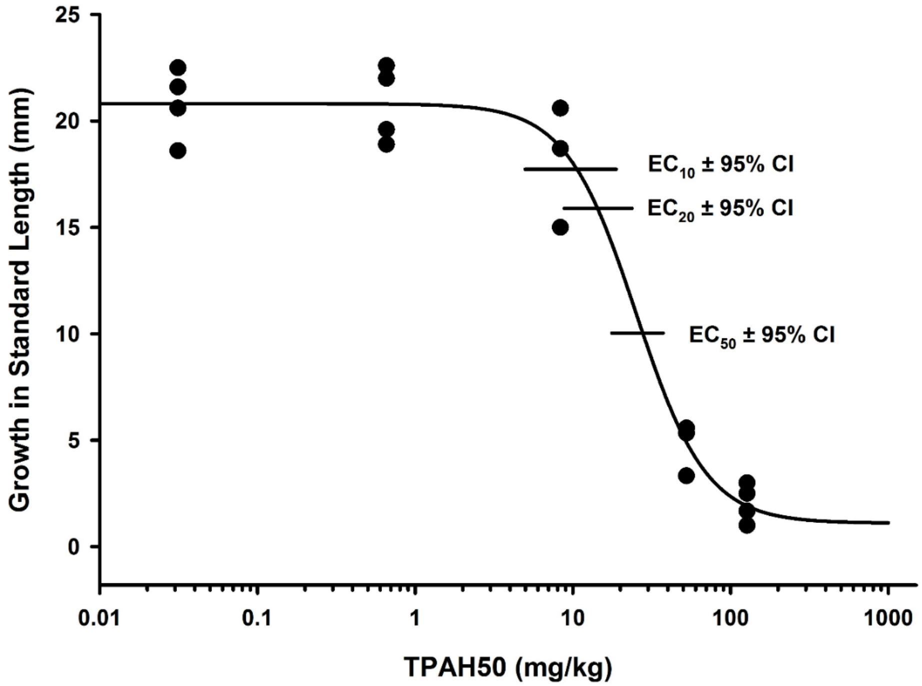
(*Paralichthys lethostigma*) use the surfaces of flooded shallow salt marsh, brackish marsh, mangrove, delta *Phragmites*, and other coastal habitats throughout the northern Gulf of Mexico (Burke 1995). Their close association with sediment makes them vulnerable to PAH in marsh soils and submerged sediments. 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 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 (Oehrig et al. 2015a). 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 4.6-30) 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 fall 2010 and fall 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 DWH oil contaminated sediments (Brown-Peterson et al. 2015).



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

Figure 4.6-30. Relationship between TPAH50 concentration in marsh soil and growth of juvenile southern flounder. As they were 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 lifespans than female flounder (4 years versus 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 4.6-31.

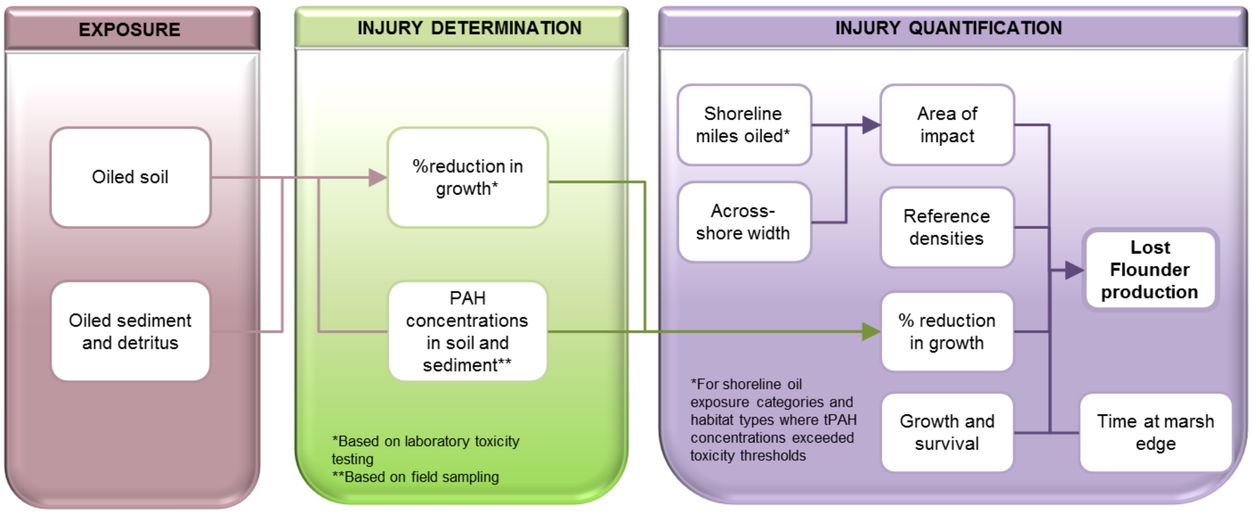


Figure 4.6-31. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to flounder occurred. Concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-30) to determine the reduction 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 12.8 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 12.8 parts per million (Powers & Scyphers 2015).

Table 4.6-15. Oiling over 39 miles (62 kilometers) 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

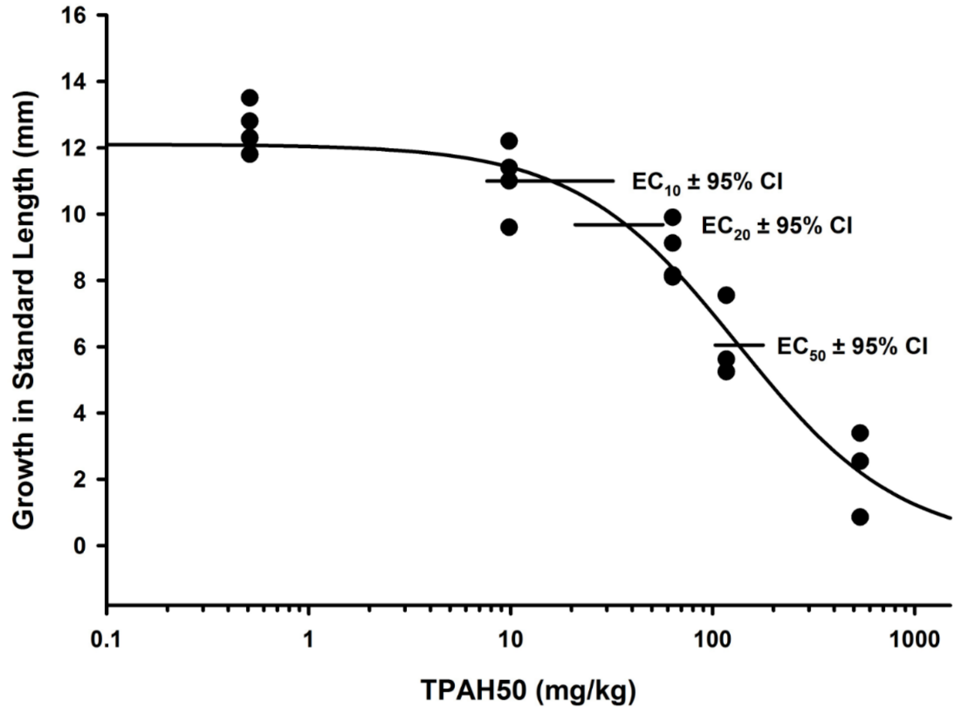
Key Points

- 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 563 MT wet weight of red drum was lost where TPAH50 concentrations in marsh soils exceeded 37 parts per million. This effect occurred over 39 miles (62 kilometers) of oiled shoreline in 2010-2012. Total PAH concentrations fell below 37 parts per million in marsh soils in 2013.

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 nearshore 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, marine worms, and (later) small fish; they are eaten by birds of prey and larger fish.

Injury Determination

Oil stranded on marsh shorelines beginning in early summer 2010. By fall 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 4.6-32) (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 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 4.6-32. Relationship between TPAH50 concentration in marsh soil and growth of juvenile red drum. As they were 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 4.6-33.

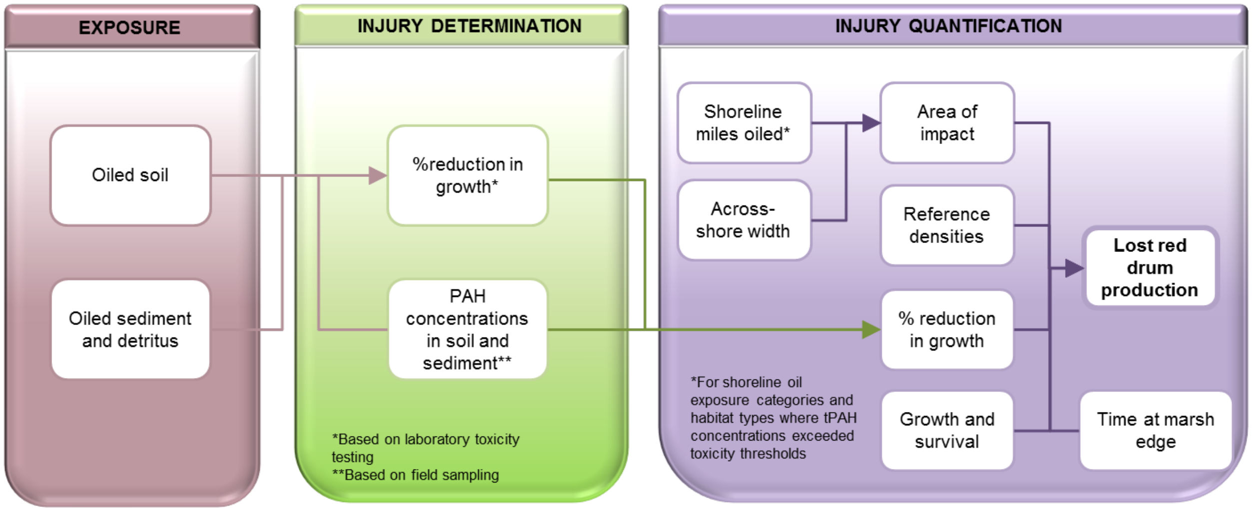


Figure 4.6-33. Illustration of information used to demonstrate exposure and to determine how (injury determination) and to what extent (injury quantification) injury to red drum occurred. TPAH50 concentrations in each marsh zone were compared to the toxicity curve (Figure 4.6-32) to determine the reduction 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.

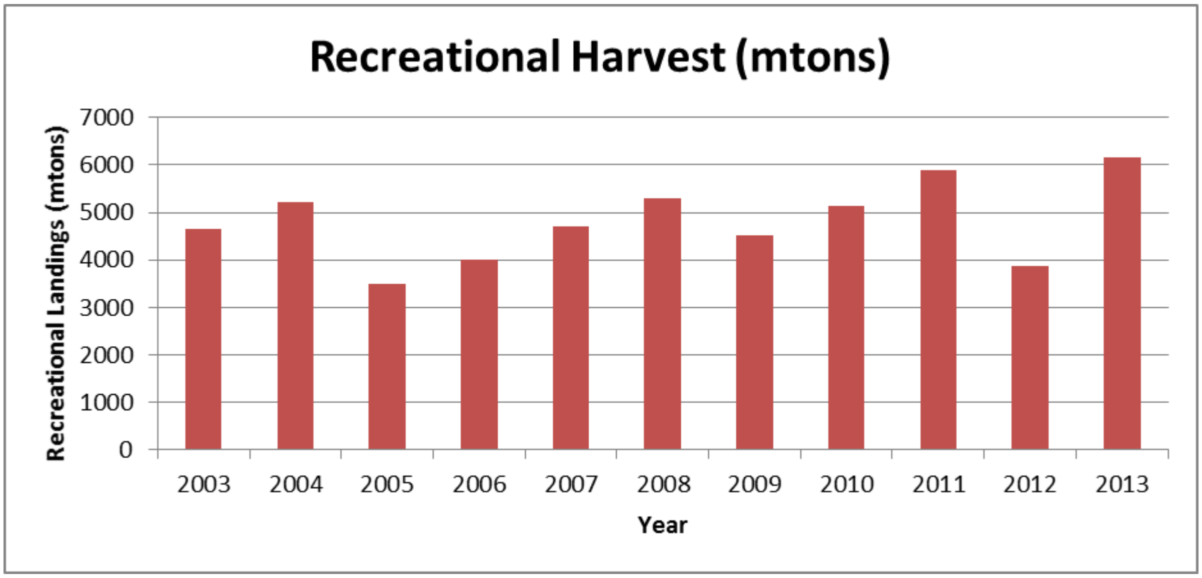
An estimated total of 563 MT wet weight of red drum was lost due to oiling of mainland herbaceous salt marsh in Louisiana where TPAH50 concentrations in marsh soils exceed 37 parts per million. This effect occurred over 39 miles (62 kilometers) of oiled mainland herbaceous shoreline in Louisiana between 2010 and 2012 (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); uncertainties in baseline densities of drum; and assumptions about growth, survival, and reproduction (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 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 of 2 to 3 years. For fish that were juveniles in fall 2010, this would be expected to occur in 2012. Notably, recreational landings of red drum in 2012 were anomalously low (Figure 4.6-34).

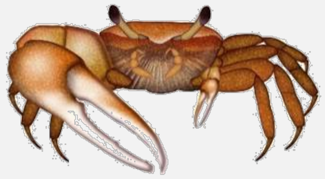


Source: *NOAA Fisheries Recreational Landings.*

Figure 4.6-34. Recreational harvest of red drum since 2003. Note that the harvest for 2012 is lower than immediately preceding years. Two- to 3-year old adult fish caught in 2012 would have been present in oiled marshes as juveniles during 2010.

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, the 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, the 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 DWH 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 by adding hard substrate adjacent to marsh edges (Meyer et al. 1997; Piazza et al. 2005; Scyphers et al. 2011).

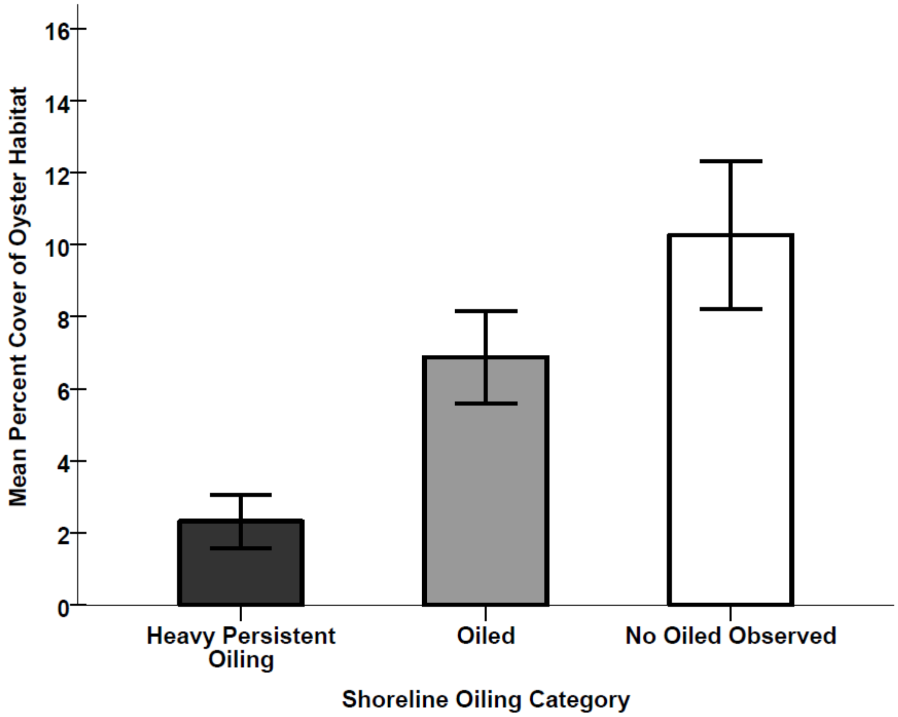
Injury Determination

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 4.6-35). 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 4.6-36). The injury resulting from the DWH oil spill in summer 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 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).

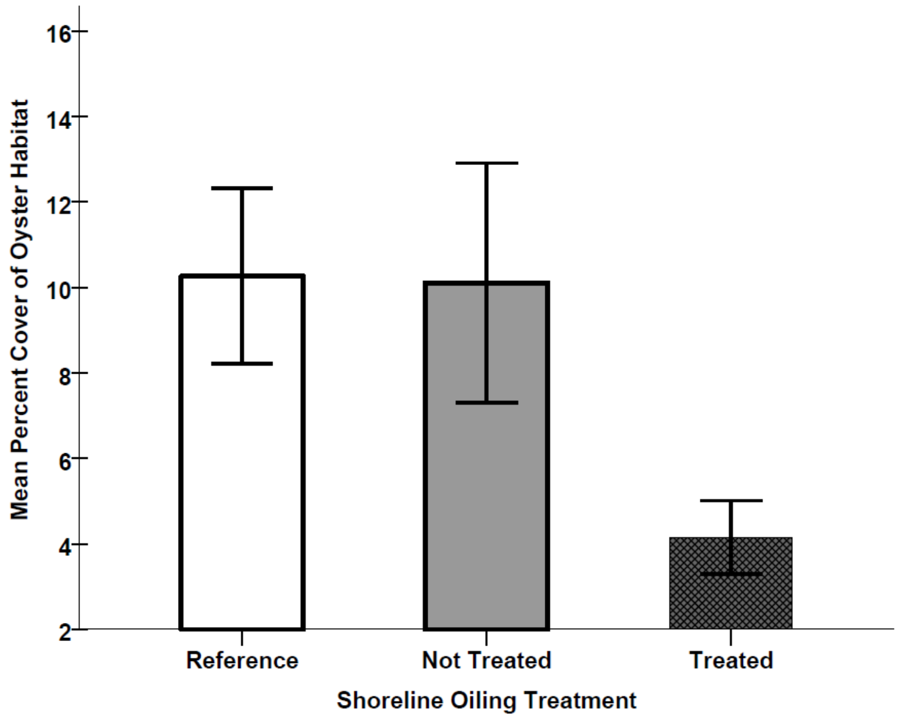
These findings are the result of NRDA field studies conducted in 2012 and 2013 to determine whether percent cover of nearshore oyster habitat and oyster abundance differed as a function of shoreline oiling and cleanup activities (Powers et al. 2015b). Overall, 187 sites from Terrebonne Bay, Louisiana, 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 Nixon et al. (2015b) were reduced to three: heavy persistent oiling, oiled, and reference ("no oil observed").[4] The heavier and lighter oiling 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).

[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 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 4.6-35. 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.



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

Figure 4.6-36. 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 4.6-37. 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.

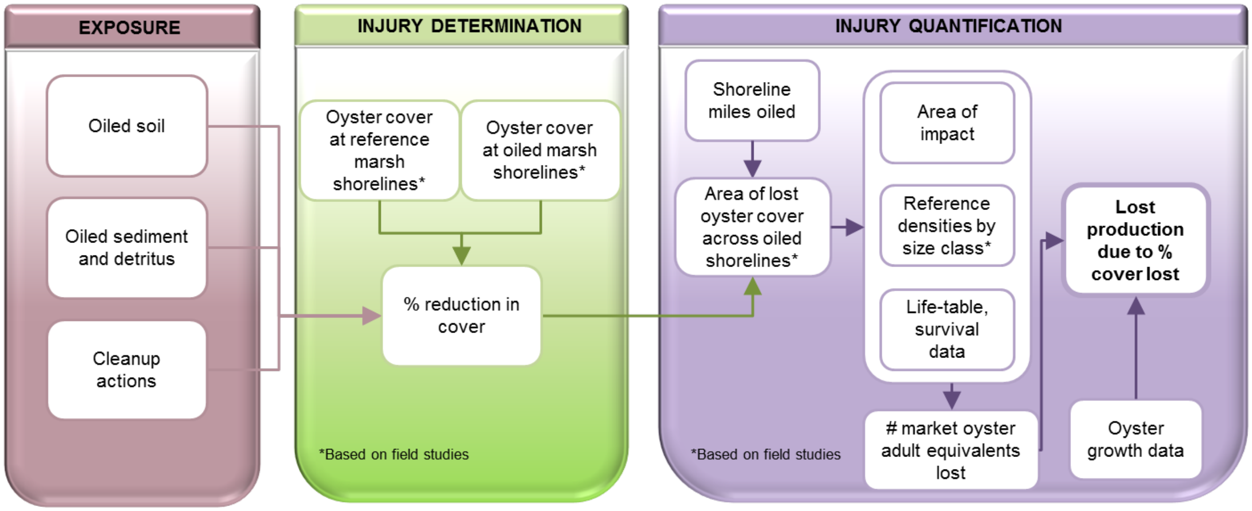


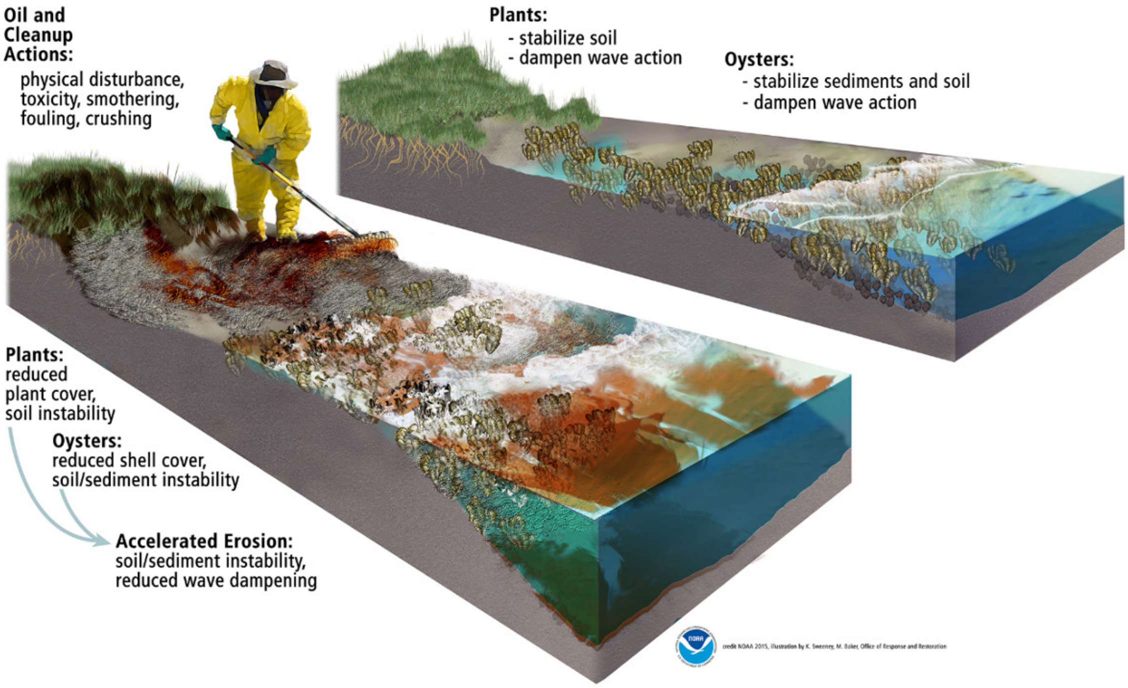
Figure 4.6-37. Illustration of information used to determine how injury to nearshore oysters occurred. Oyster cover in nearshore environments was mapped at oiled sites and unoiled (or reference) sites during field studies and extrapolated to the entire vegetated shoreline to determine the total amount 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 DWH 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 DWH 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 4.6-38 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.*

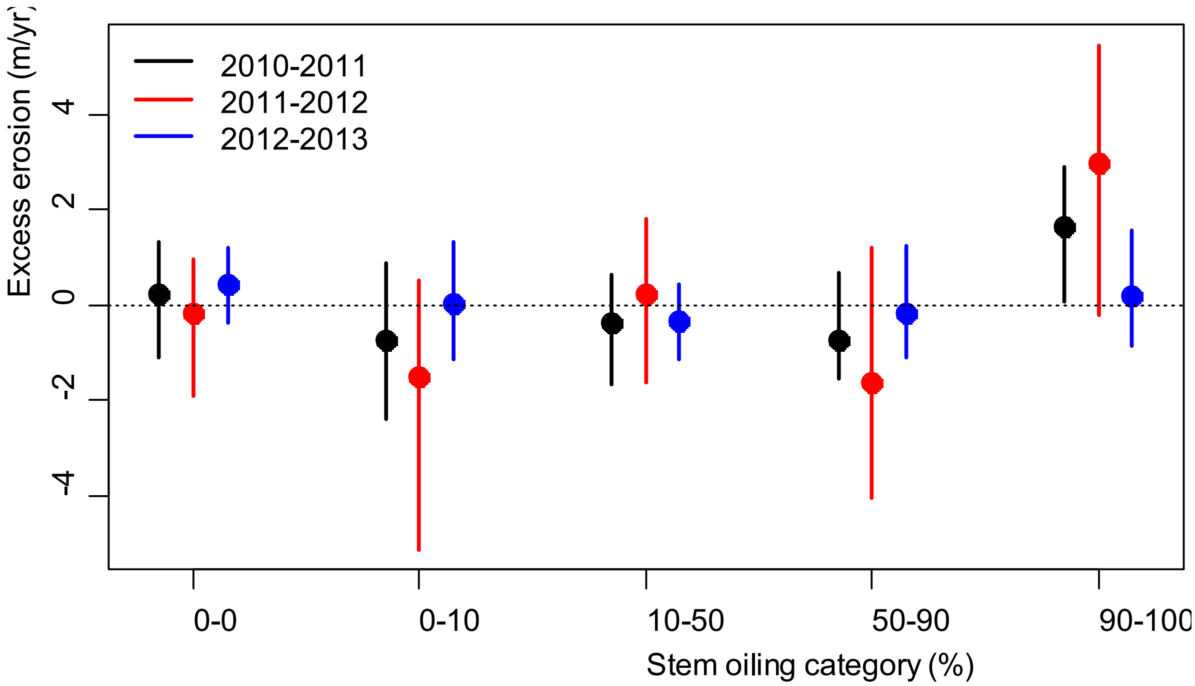
Figure 4.6-38. Trustees explored relationships between shoreline oiling, cleanup actions, plant oiling, oyster cover, and erosion. Mechanisms that could enhance erosion as a result of spill impacts include a loss of soil stability (which could be affected by loss of plant cover, physical trampling, or 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 meters/year from fall 2010 to spring 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 meters/year, and the control sites retreated at a rate of 0.94 meters/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 4.6-39) 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 meters/year 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), demonstrates that marsh oiling resulted in increased rates of erosion and loss of Louisiana coastal salt marsh.

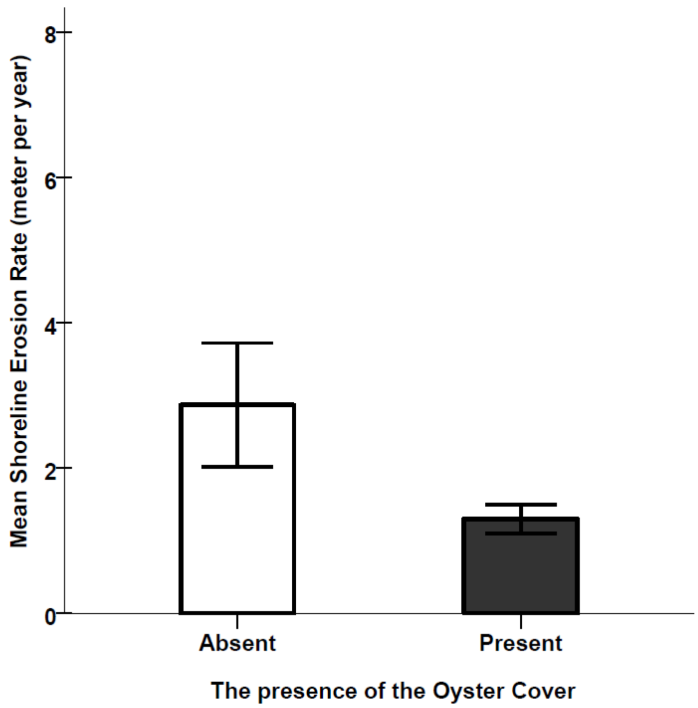


Source: *Silliman et al. (2015).*

Figure 4.6-39. 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 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; 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/year (Figure 4.6-40). 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 4.6-40. 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 heavy 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 DWH 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

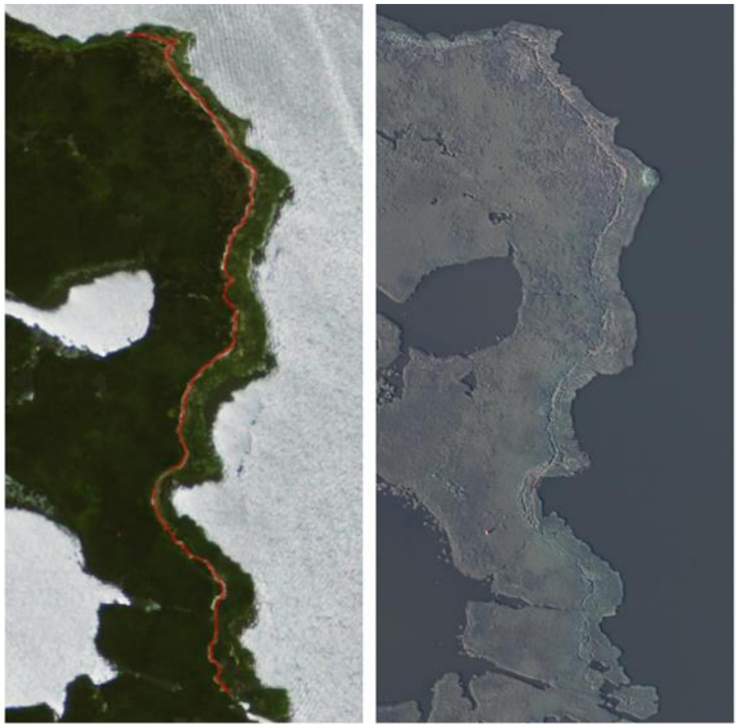
As described in Chapter 2 (Incident Description), several different types of response activities took place in the marsh. The Trustees evaluated the impacts of such activities on accelerated erosion (see Section 4.6.4.6). Here, the Trustees analyze marsh impacts as a result of boom that was deployed 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 4.6-41 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 4.6-42 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; this boom stranding potentially damaged nests, crushed eggs and chicks, and limited the re-occupation of these areas for nesting post-storm (Figure 4.6-41). 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 4.6-41. Stranded boom on wetlands in St. Bernard and Plaquemines Parish, July and August 2010. Vegetation was crushed under the boom.

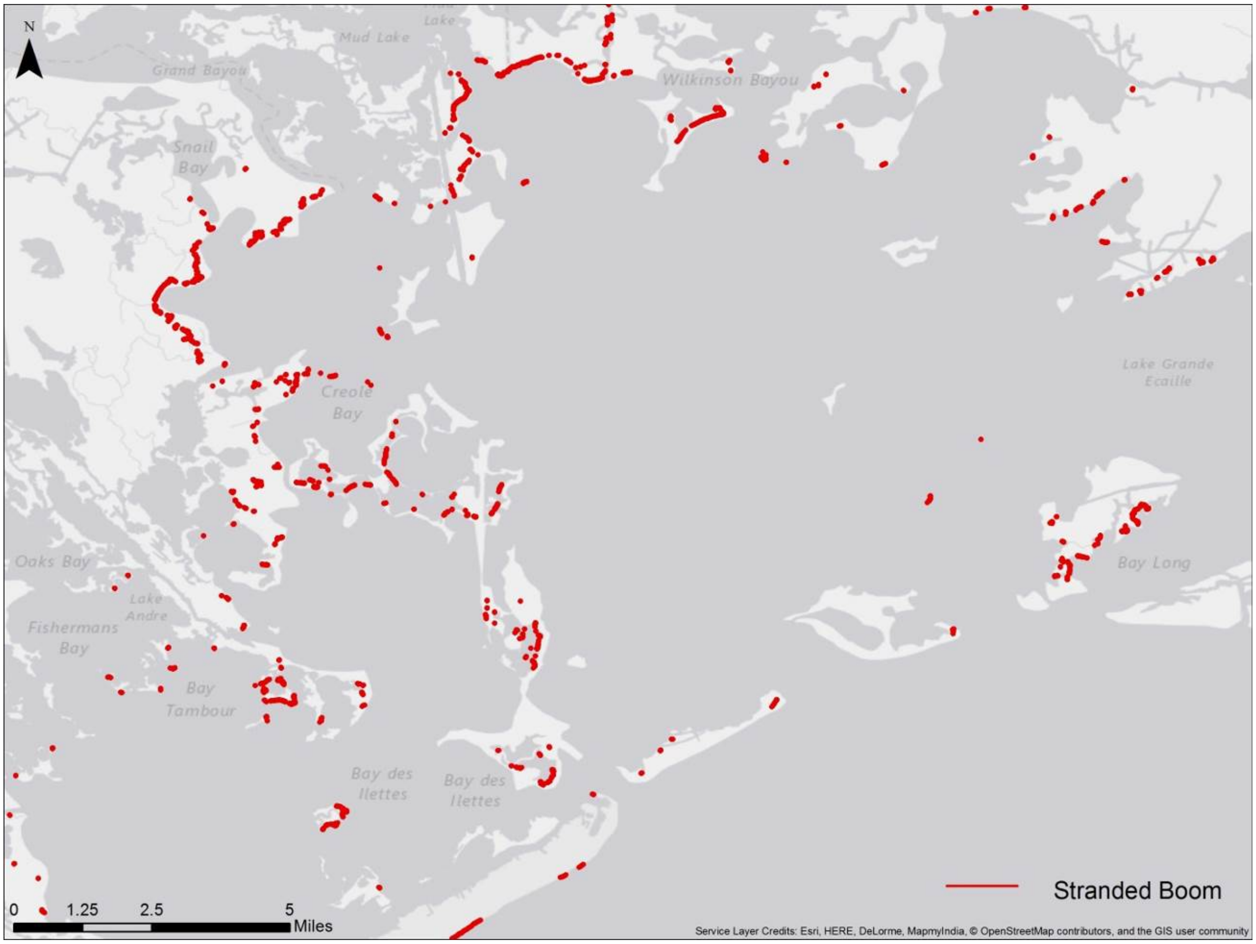


Sources: Left image: NAIP (2010); data available from the U.S. Geological Survey. Right image: ERMA (2015).

Figure 4.6-42. 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

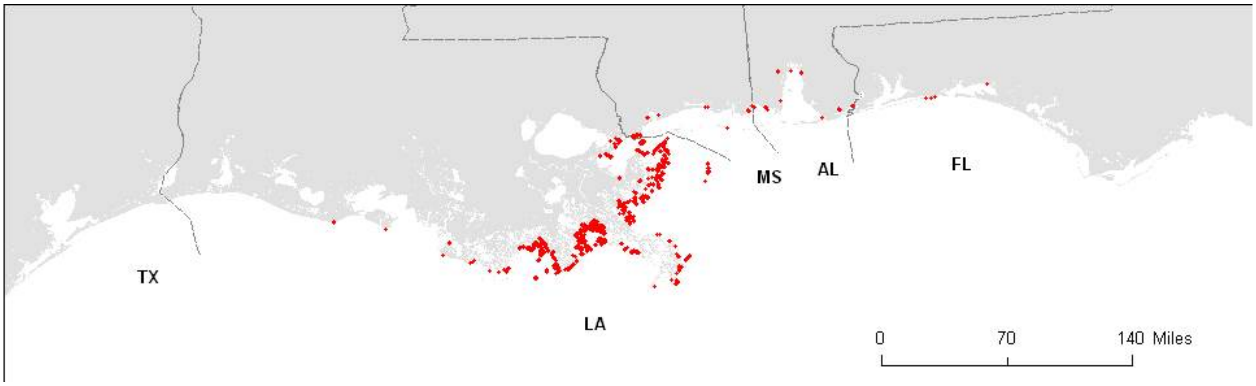
As an illustrative example of the extent of boom stranded in wetlands, the Trustees digitized stranded boom in Barataria Bay from a set of aerial images collected during overflights conducted in late July 2010 (ERMA 2015). For just this one snapshot in time, approximately 19 miles of boom were identified as stranded in Barataria Bay wetland habitat (Figure 4.6-43).



Source: *ERMA (2015).*

Figure 4.6-43. 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 upward of 497 miles (800 kilometers) of boom were stranded in wetlands (Figure 4.6-44) (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 4.6-44. 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* aboveground 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	95%	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

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
***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)	2010–2012	3 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 life cycle. 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.

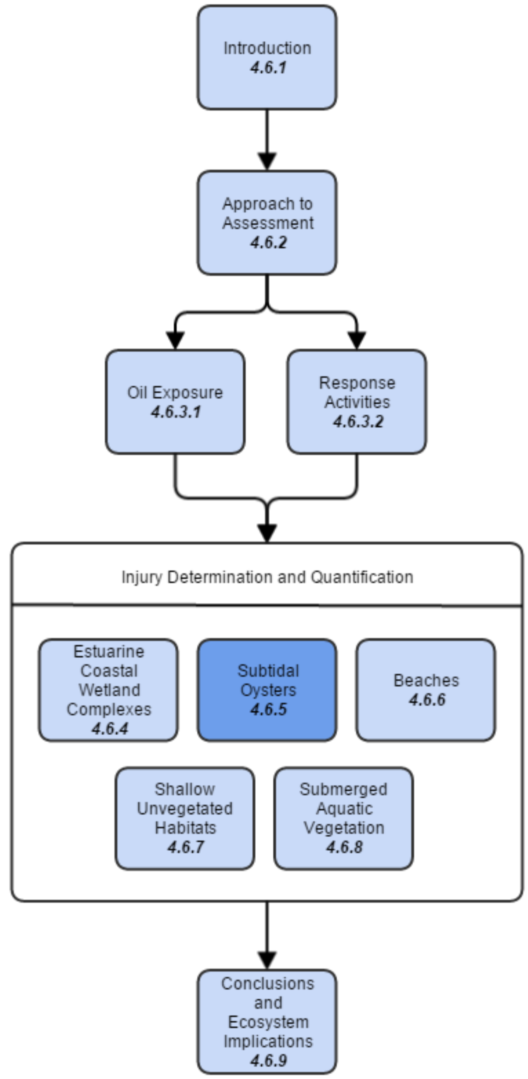
Mainland herbaceous salt marshes across Louisiana, Mississippi, and Alabama were impacted. Louisiana salt marshes experienced reductions in live aboveground biomass and live plant cover ranging from 11 to 53 percent compared to reference conditions over a total of 350 to 721 miles (563 to 1,161 kilometers) (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 (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 95 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; 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) (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 SAV (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 DWH-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 DWH 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 fresh water from natural events, such as storms, the opening of the salinity control structures exposed oysters to low salinities for much longer timeframes 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 summer 2010 (Powers et al. 2015a).

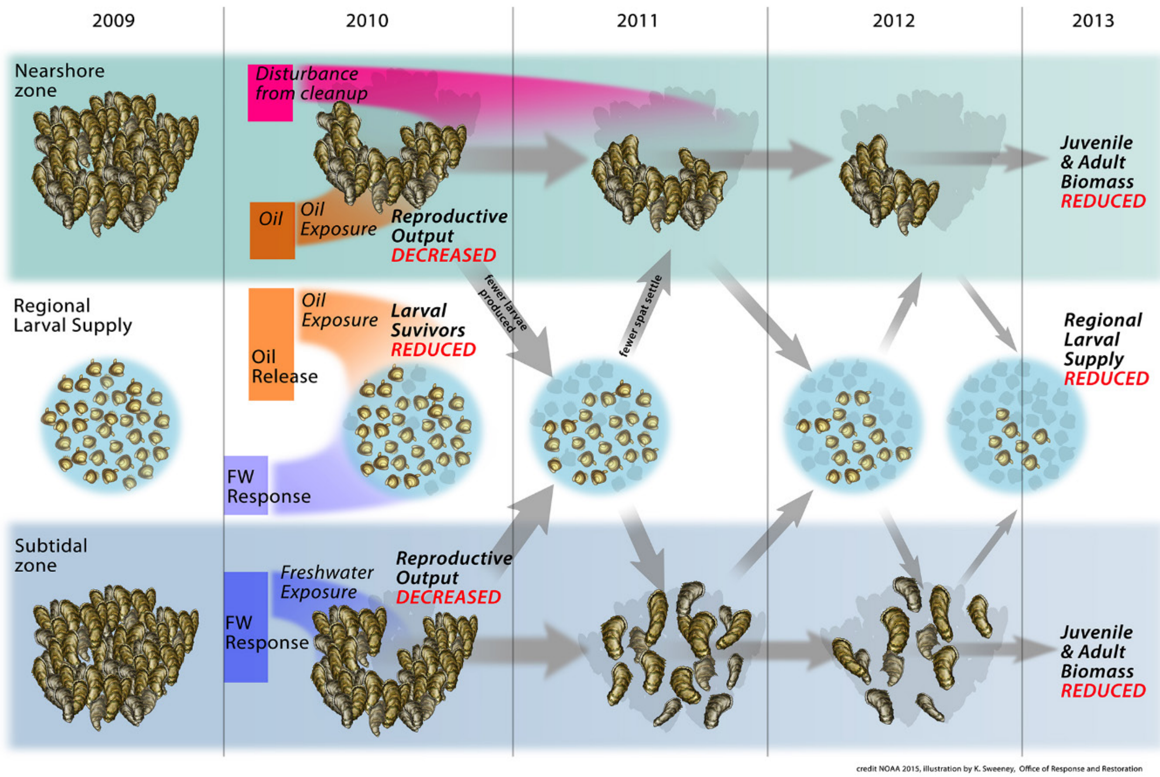
The relationship between oil exposure and abundance of subtidal oysters was examined by evaluating relationships between abundance of each age class (adult, juvenile, and spat) and oyster tissue PAH concentrations, oiling (as measured in terms of co-located sediment TPAH50), and oil-on-water (days, frequencies,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

Exposure to DWH-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 fresh water and areas less exposed, using data collected under the DWH 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 Louisiana Department of Wildlife and Fisheries (LDWF) Nestier tray studies to identify areas likely to have experienced decreased survival due to fresh water 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.*

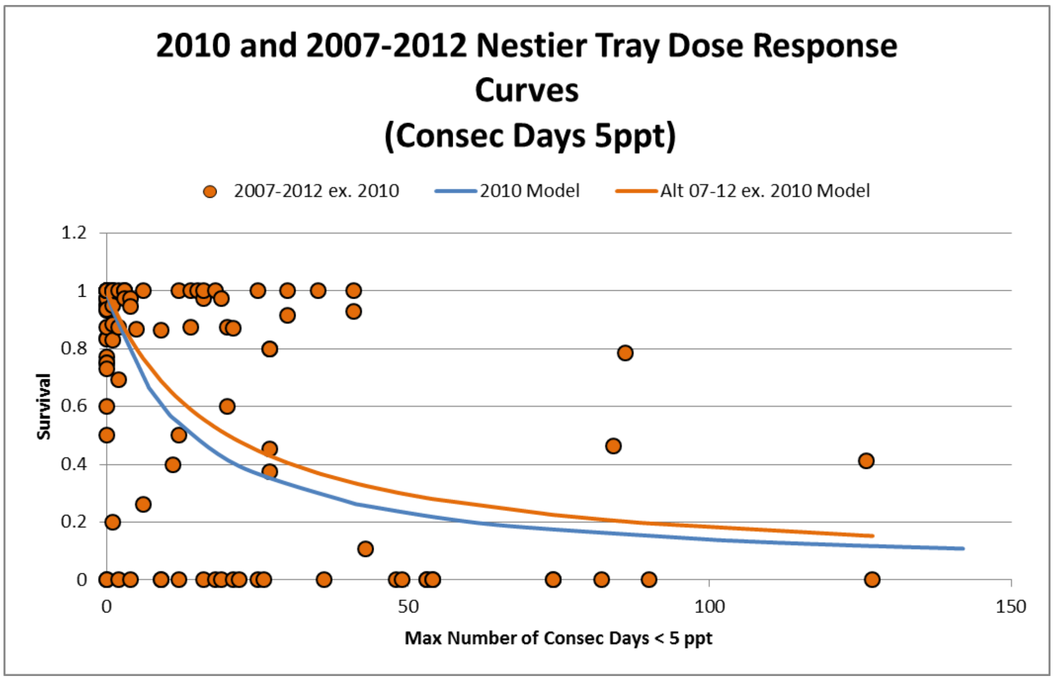
Figure 4.6-45. This figure illustrates: 1) how the DWH spill harmed oyster populations in the Gulf of Mexico and 2) how the connections between oyster populations in the nearshore and subtidal zones and the regional larval supply could result in ongoing suppression of the oyster population following 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 4.6-45, oysters could be affected in many ways: by river water released as part of DWH 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 late spring and summer 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 DWH 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 summer 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 4.6-46) (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).

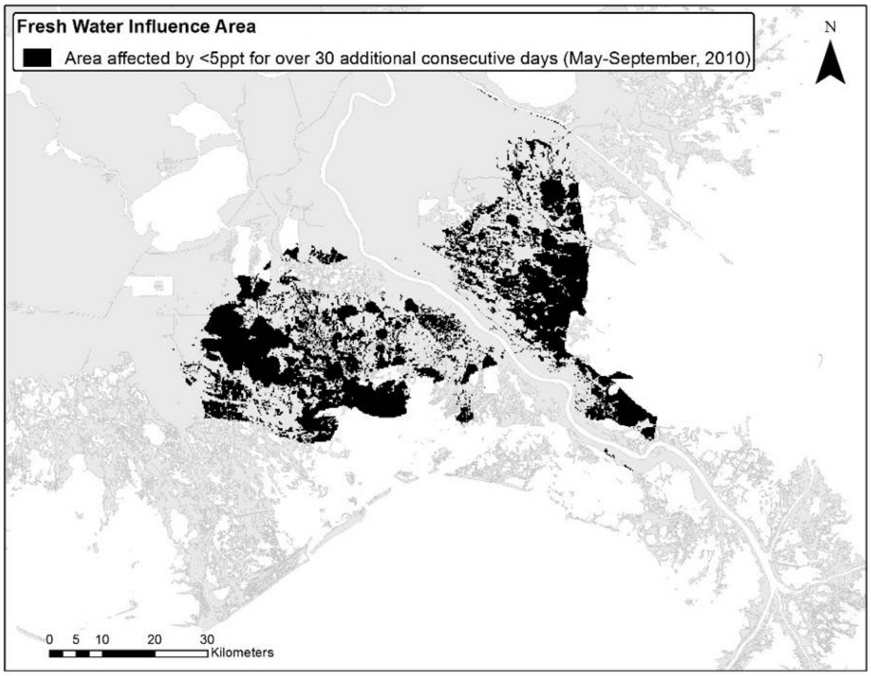
Figure 4.6-46. This graph shows the relationship between exposure to fresh water and oyster survival. As the number of consecutive days with salinity below 5 parts per thousand increases (horizontal axis), survival of oysters drops (vertical axis) and more oysters die. For example, a survival of 0.400 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 DWH spill could also have potentially harmed oysters (Morris et al. 2015), confirmation of such exposure is limited (Oehrig et al. 2015b). 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 DWH 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 fresh water (Figure 4.6-46) (Powers et al. 2015a; Rouhani & Oehrig 2015a, 2015b).

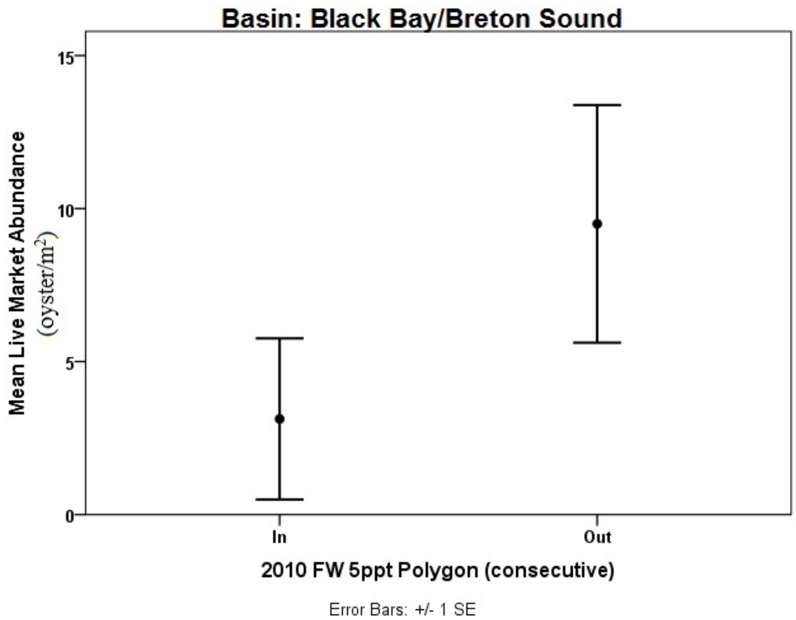
The area of freshwater impact was determined by interpolating thousands of salinity values throughout the estuary and comparing 2010 salinities to those in the years prior to the spill (2006 to 2009) (McDonald et al. 2015; Rouhani & Oehrig 2015b) (Figure 4.6-47). For each 200-square-meter grid cell in 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 freshwater 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 4.6-47 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 4.6-47. 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 4.6-48). All approaches to estimating DWH-related subtidal oyster injury used field-collected abundance data, either from Louisiana's oyster fisheries-independent monitoring program or from the DWH NRDA. All approaches also showed that the fresh water 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 4.6-48. 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 4.6-47, 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).

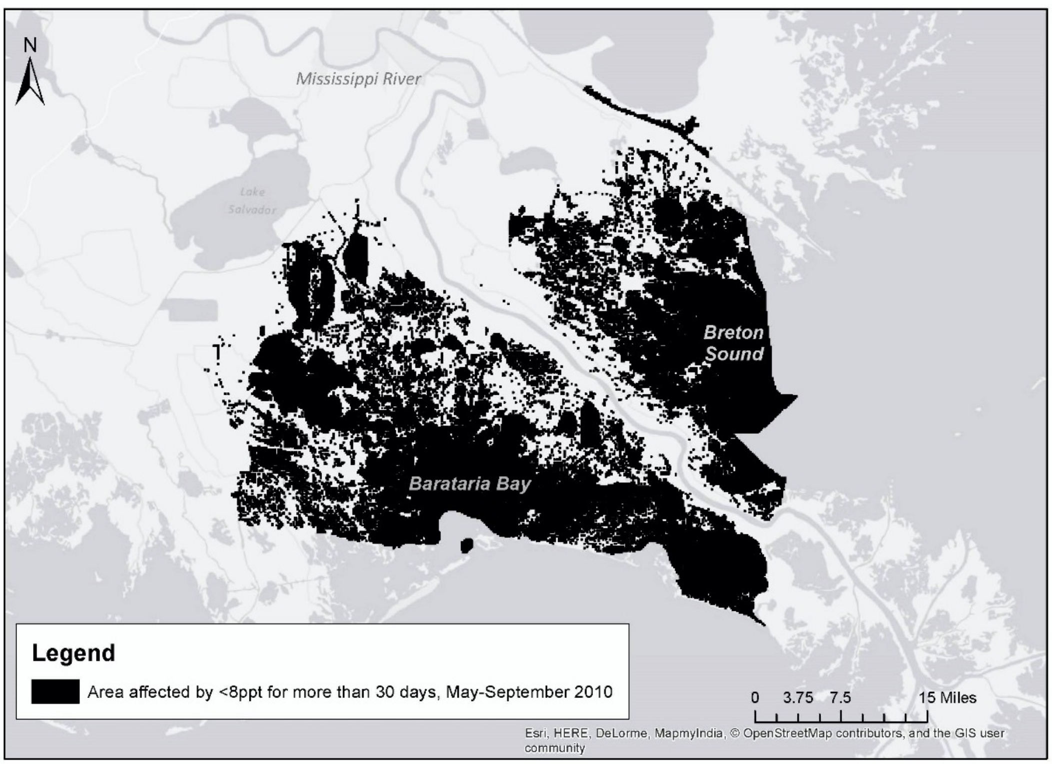
In addition to the NRDA analysis of injury to subtidal oyster abundance, the Trustees (Grabowski et al. 2015b) evaluated fisheries-independent data on oyster abundance collected by Gulf state monitoring programs to evaluate trends over time (i.e., before and after the spill). The Trustees found significant 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.

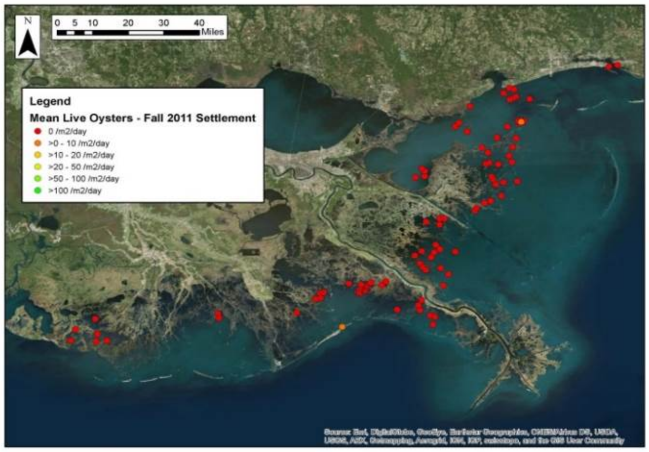
Extended periods of failure of any part of the reproductive cycle can lead to sedimentation of existing reefs, removing substrate for settlement and reducing oyster cover over time.

With reduced numbers of juvenile and adult oysters in subtidal areas in 2010, fewer larvae were produced in 2011 and beyond (Grabowski et al. 2015a). In addition to the area where salinity dropped low enough to kill oysters in 2010, Figure 4.6-49 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 4.6-50) (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 4.6-49. 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 DWH spill.



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

Figure 4.6-50. This map shows measurements of oyster spat settlement in the northern Gulf of Mexico in fall 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 late spring and summer 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 DWH 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 DWH-related mortality in 2010 (subtidal) and 2010–2011 (nearshore). While it is true that 2011 was a high flow year for the Mississippi River, which 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 4.6-51 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).

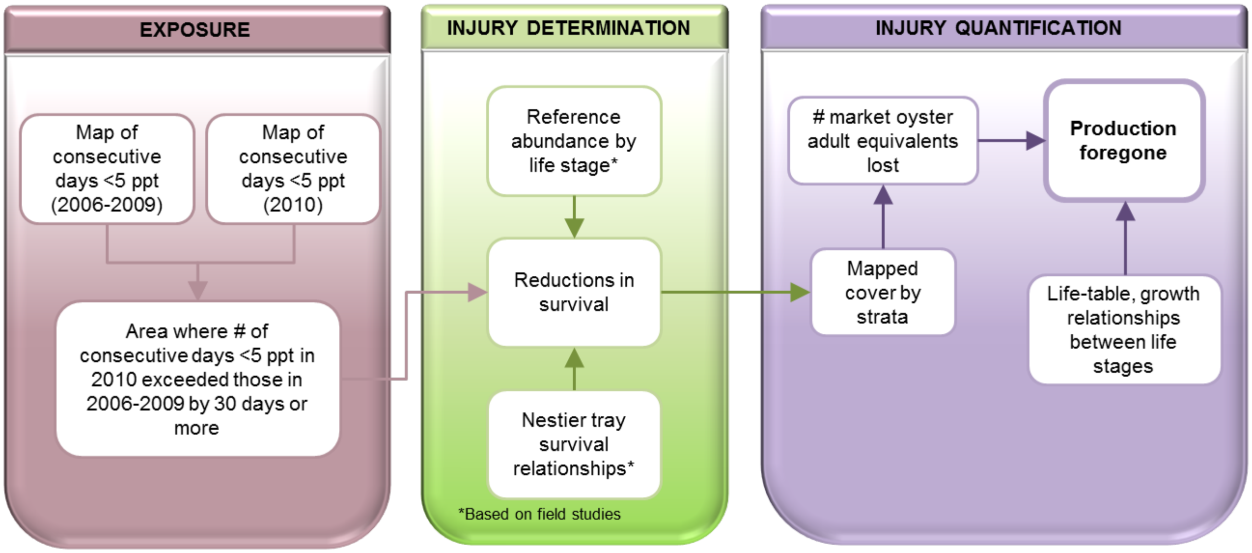


Figure 4.6-51. This diagram illustrates the analysis used to determine and quantify injury to oysters living in subtidal areas. Left box: Thousands of salinity and temperature measurements were gathered and used to identify areas where salinity was unusually low in 2010 compared to prior years. Center 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 4.6-46). 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 fresh water and areas less exposed, using data collected under the DWH NRDA in 2010.
- The second approach ("Fisheries Temporal") quantified injury by comparing abundance in 2010 to those in prior years, both for the impact area and the entire basin, 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 fresh water and expected oyster death, derived from annual LDWF Nestier tray studies (Figure 4.6-46), to identify areas likely to have experienced decreased survival due to fresh water from the Davis Pond and Caernarvon structures.

Figure 4.6-51 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 fresh water 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. 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 fresh water, along with the eggs not produced by surviving oysters (Figure 4.6-52). These foregone eggs were converted to foregone adult equivalent oysters 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 & 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 fresh water 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; 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 4.6-52. It should be noted that while the reproductive output of subtidal oyster populations may gradually recover over 7 years without intervention (because oyster shell cover is still present in these areas), the reproductive

[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.

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).

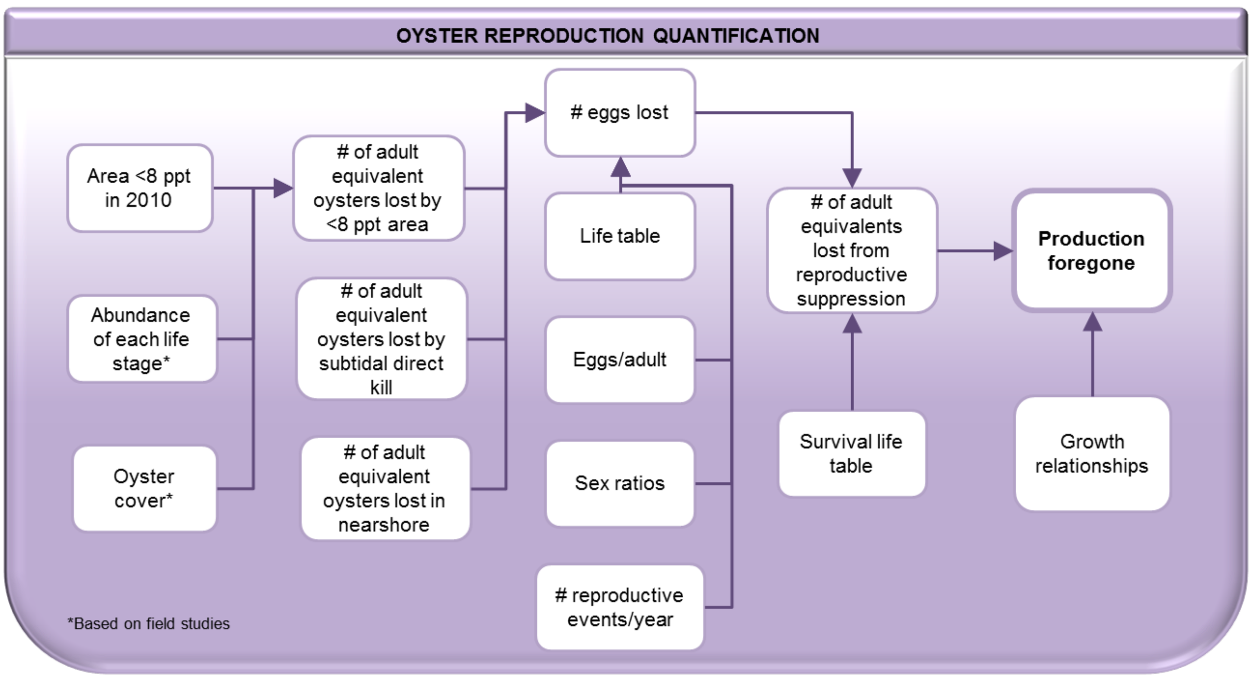


Figure 4.6-52. This diagram illustrates the calculation of lost oysters and production from injury to oyster reproduction. Oysters living in areas where salinity was lower than 8 parts per thousand in 2010 would not have reproduced, subtidal oysters killed by river water releases did not reproduce, and 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 fresh water show that the DWH 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 last few 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 DWH 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 DWH 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:

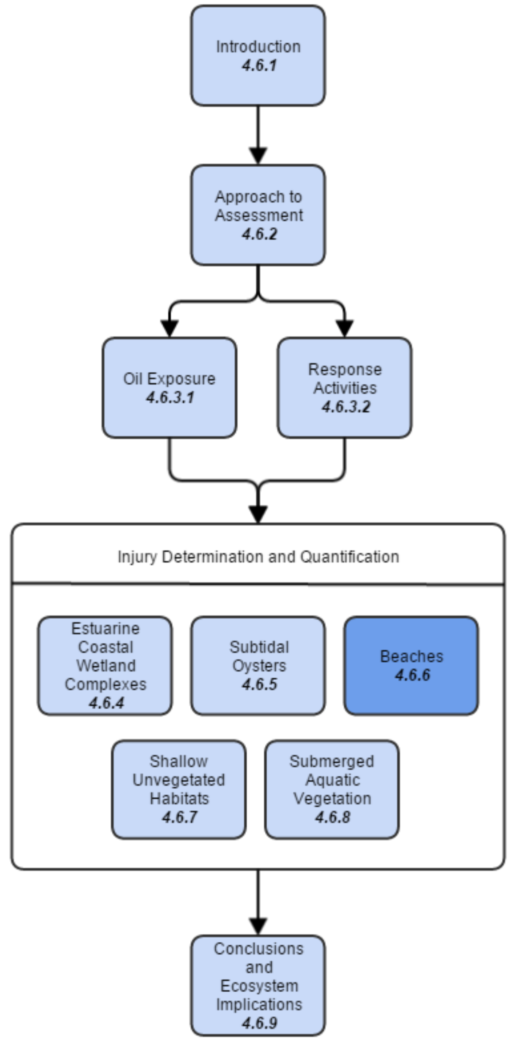
- As a result of the oil spill and response activities, between 4 and 8.3 billion subtidal oysters (adult equivalents) are estimated to have been lost.
- The abundance of subtidal oysters in coastal Louisiana was dramatically reduced by summer river water releases conducted as part of response actions to the DWH 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 DWH oil and response activities?
- **Injury Determination** (Section **4.6.6.3**): How did exposure to DWH 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 DWH 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 DWH 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 4.6-53):

- **Step 1 (box 1 in Figure 4.6-53).** The Trustees first characterized the exposure of sand beaches to DWH oil and response activities. This characterization took into consideration the repeated oiling of beaches that occurred over the duration of the DWH 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 USCG'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 4.6-53)**. The Trustees then determined the nature and extent of injuries to sand beach habitat caused by the oil and response activities. The Trustees conducted an exhaustive review of existing literature to

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 DWH 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.

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 4.6-53).** 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).

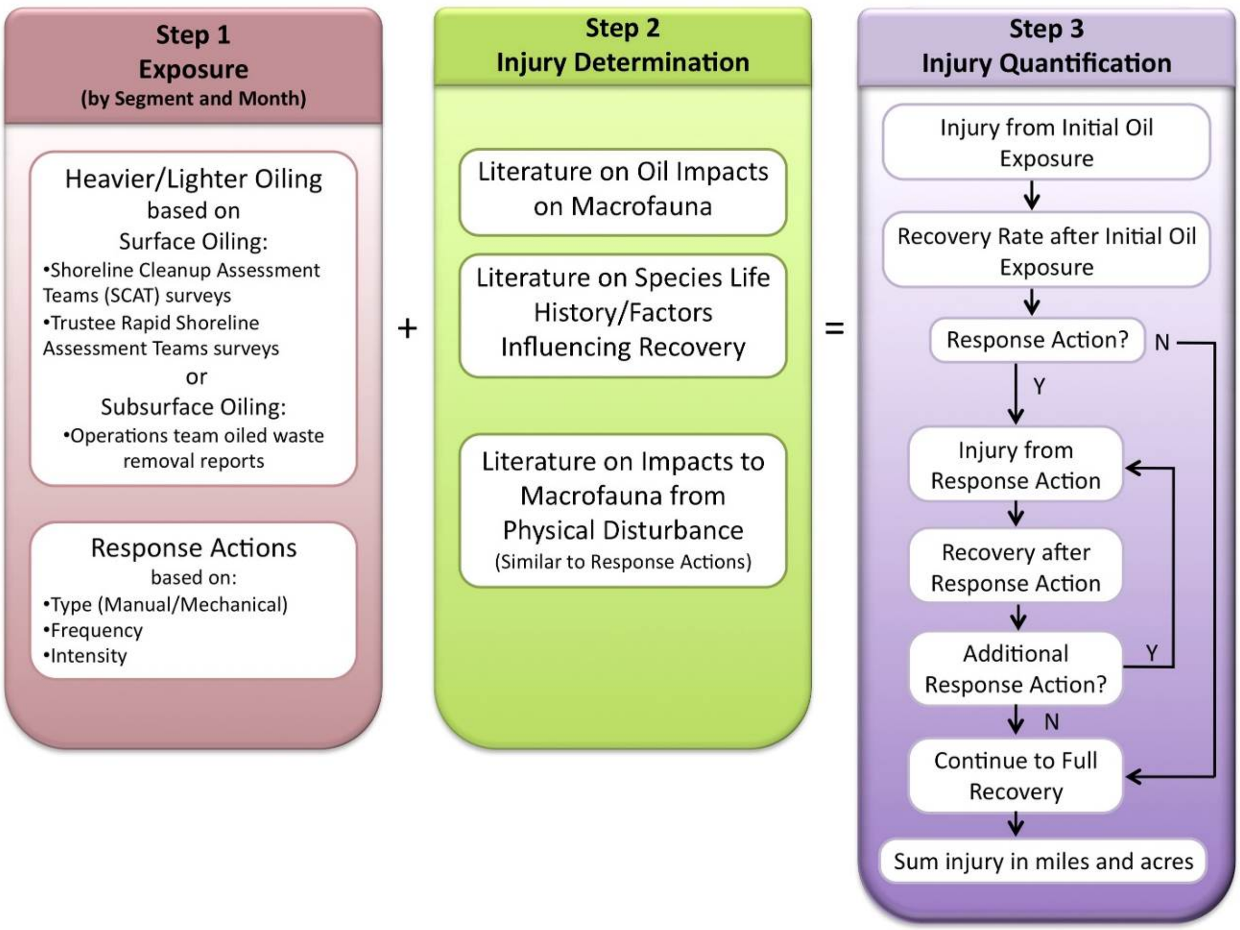


Figure 4.6-53. Approach to the sand beach exposure and injury assessment. The approach addresses impacts resulting from both oiling and response activities during the DWH spill. The geographic extent of beach oiling and response actions was compiled (step 1, evaluating exposure). Literature information on effects of oil and response actions to beach fauna was compiled (step 2, determining injury). The length of shoreline injured to different degrees was quantified using observations of beach oiling and response actions and the literature information on effects (step 3, quantifying injury).

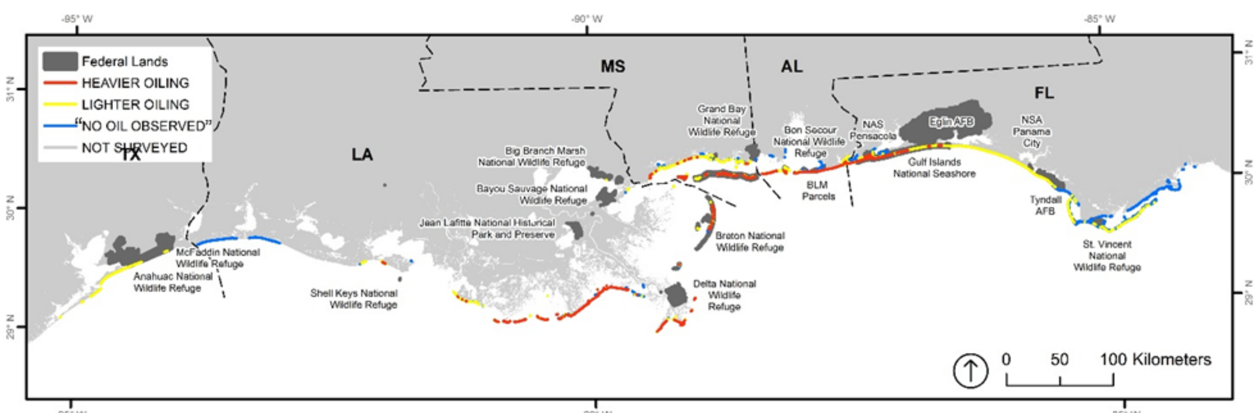
4.6.6.2 Exposure to Oil and Response Activities

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 DWH 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 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 4.6-54 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 the U.S. Department of the Interior (DOI) and the U.S. Department of Defense (DOD) within each state. Figure 4.6-55 shows the same information graphically for oiled shoreline lengths.



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

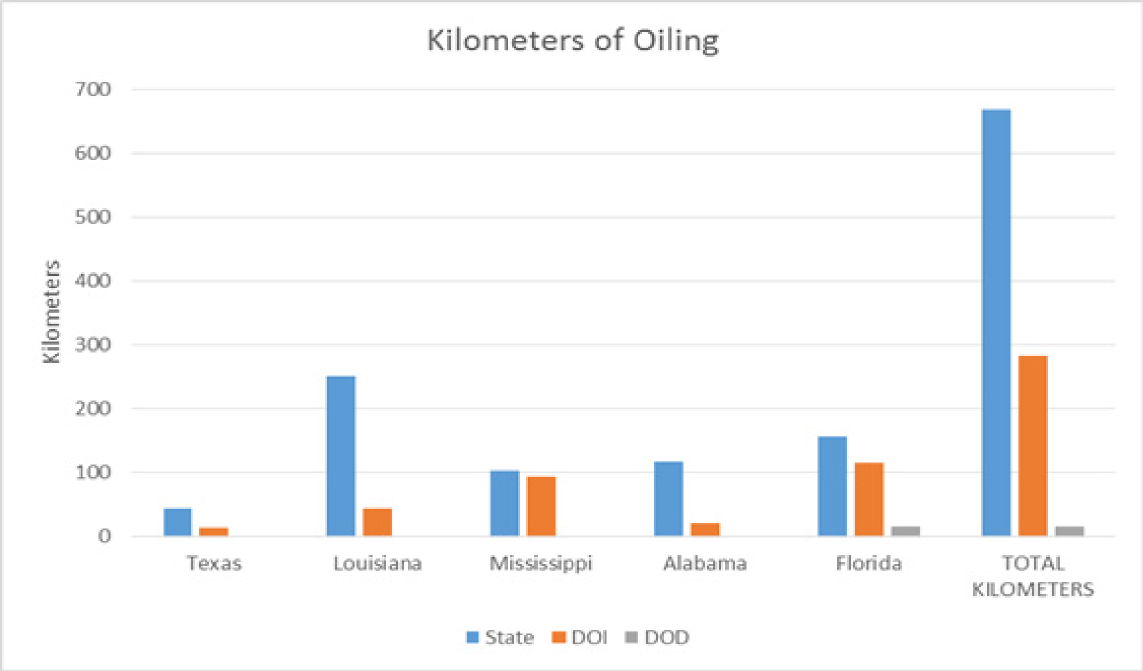
Figure 4.6-54. Map of the northern Gulf of Mexico, showing beaches that were oiled during the DWH spill. For the sand beach injury assessment, oiling exposure was grouped into two categories: lighter and heavier oiling. This map illustrates the extensive spatial extent of sand beach habitat that was oiled as a result of the DWH oil spill, from Texas to Florida.

4.6.6 Beach Assessment

Table 4.6-18. Length and area of oiling by state lands and DOI and 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 4.6-55. Kilometers of shoreline oiling in each state and on federal lands (DOI/DOD).

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 DWH 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 4.6-56). 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.

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.

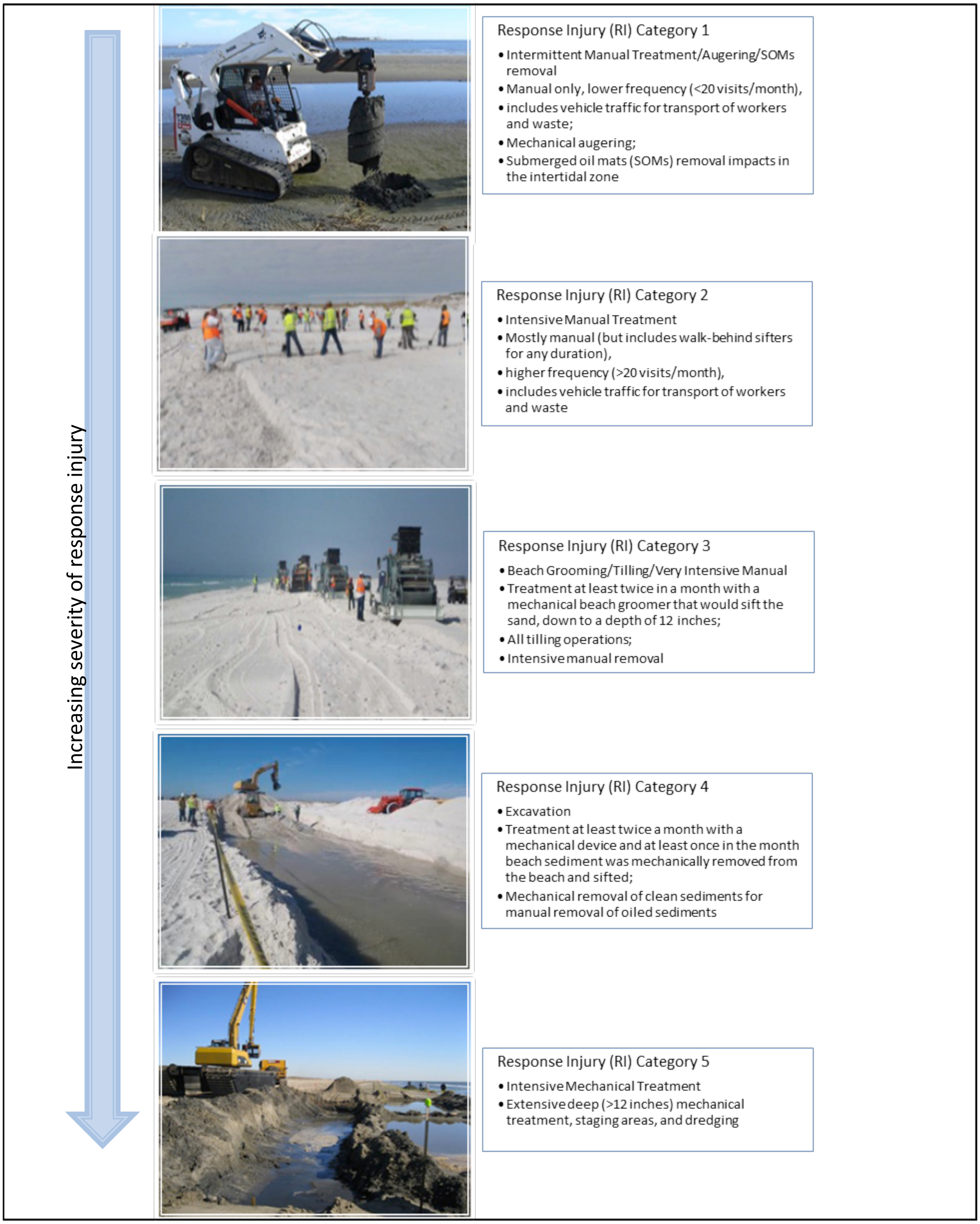
Cleanup did not occur at all oiled sand beaches for several reasons, such as:

- 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).

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



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

Figure 4.6-56. Types and descriptions of the five RI categories for sand beach impact assessment.

Table 4.6-20. Length and area of shoreline that were assigned an RI category (including placement of barriers) by state lands and DOI and DOD lands, by state. The top table presents shoreline lengths in miles and acres; 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 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.

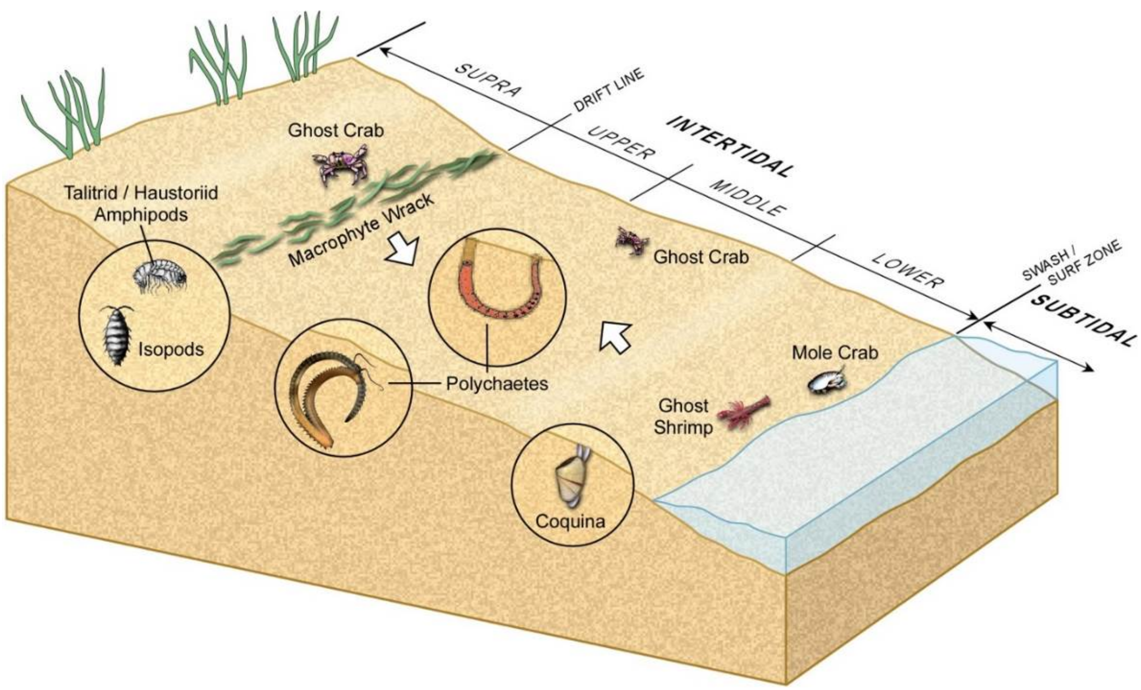
4.6.6.3.1 Macrofauna

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 their 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; they were used in this assessment to gauge the level of injury caused by the DWH oil and spill response activities.

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 4.6-59). 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).

The Trustees focused on macrofauna in their determination of injury because these small organisms are a key component of sand beach ecosystems and are a 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.



Source: Michel et al. (2015).

Figure 4.6-57. 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.

2. How oil and different degrees of disturbances similar to the response activities affect these communities as they start to recover from the DWH 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; 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 DWH 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 DWH 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 (Nixon et al. 2015b), 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 generally examine much smaller spills in terms of the volume spilled. Smaller spills therefore affected shorter lengths of coastline; as a result, studies of earlier spills can only provide a minimum measure of likely impacts from the DWH 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 with 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 with unoiled locations, and locations with lighter oiling had a 40 percent decrease.

The Trustees also reviewed 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 an 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; that research was used to estimate the degree of invertebrate community disturbance and to assign each activity to an RI category.

Table 4.6-21. Expected impacts from different types of response-related activities that were conducted during the DWH 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 tenfold 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. Alteration 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, which 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 life cycles 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 timeframes 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 replacing 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 distance 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; this will slow their recovery rate.

Further discussion of sand beach macrofauna recovery timeframes subsequent to the DWH 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 DWH 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, nest destruction, and egg and chick mortality (see text box below). These types of activities are comparable to certain DWH 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

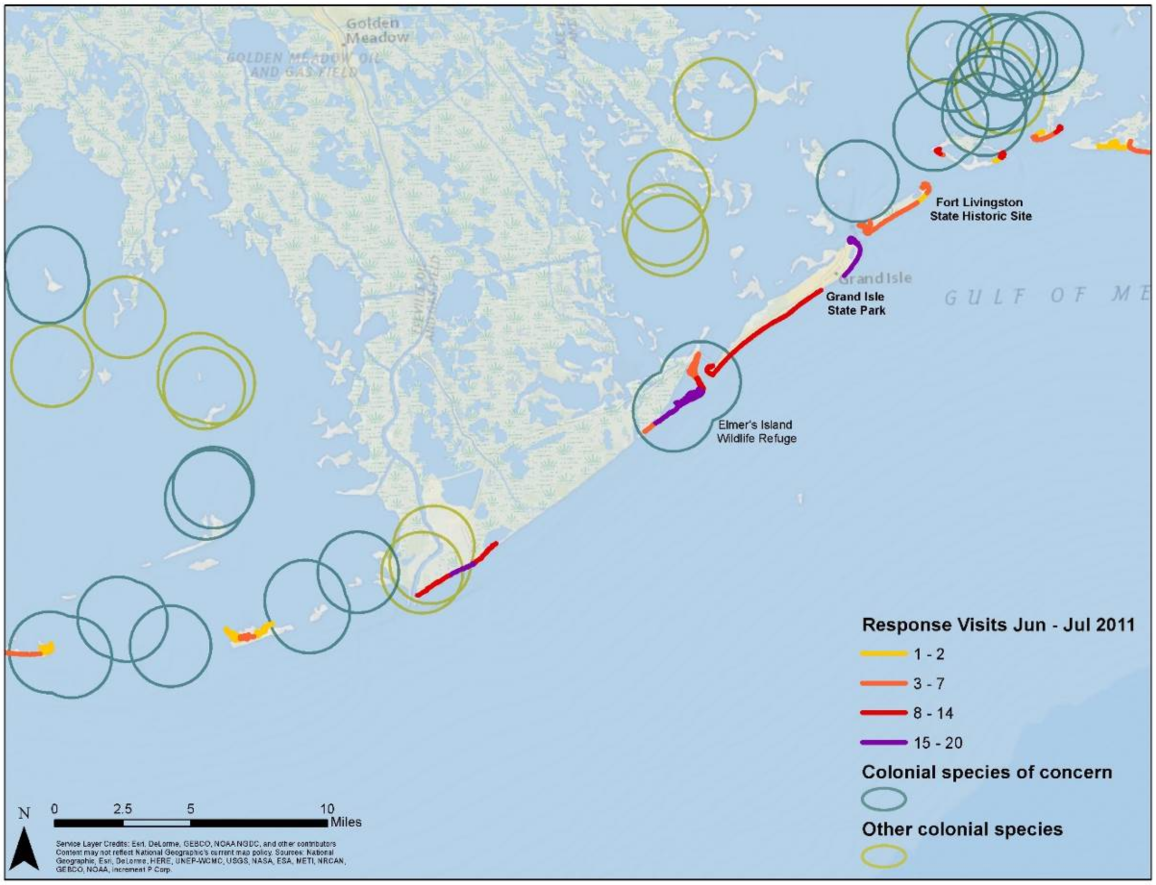
Further, even small numbers of response activity visits on a given beach could have pronounced negative effects on the reproduction of some species, if the activities took place during nesting season (April to July). After a nest failure, certain bird species—American oystercatchers, snowy plover, and 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 4.6-60 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 to 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; 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 4.6-58. 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 to 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 4.6-56 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 DWH spill not occurred. Step 3 in Figure 4.6-53 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 RI 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 fall 2010/2011. After that, buried oil at selected Alabama and Florida beaches was 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 2 months (July and August) in summer 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. RI was not assessed for Texas due to the short duration of operations.

Table 4.6-22. Length and area of RI by state, including federal lands (DOI, DOD). Injury quantification as presented in Michel et al. (2015). For DOD lands, RI 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; 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	1,291
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	2,694
TOTAL	**208**	**1,067**	**74**	**349**	**113**	**516**	**66**	**762**	**460**	**2,694**
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	1,162
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**	**1,677**	**232**	**2,460**
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 DWH 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 DWH 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 DWH 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:

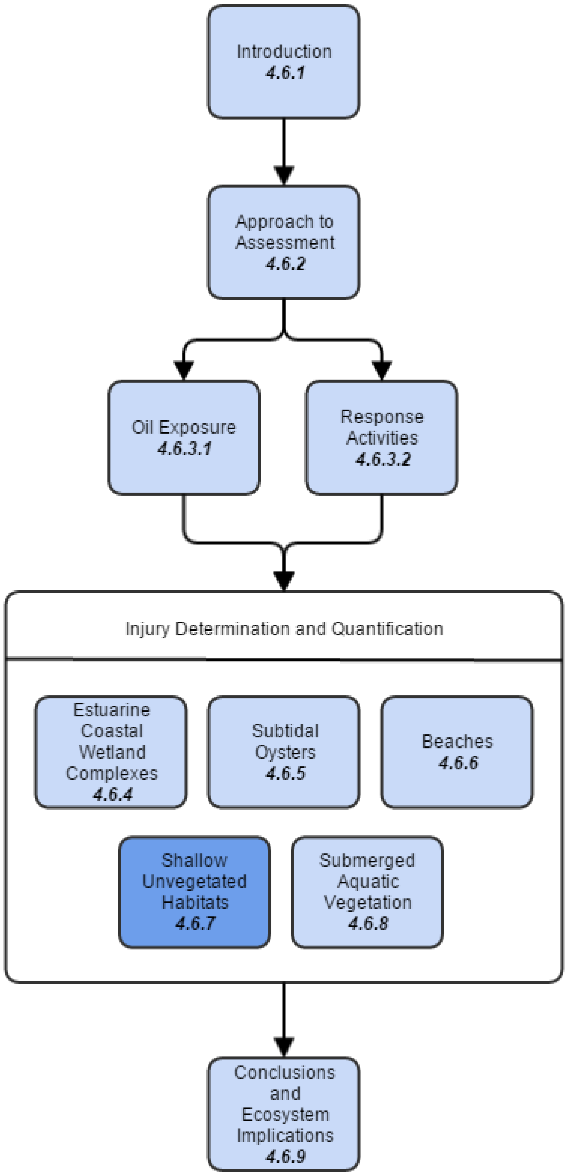
- 965 kilometers (600 miles) of sand beach and dune habitat along shorelines and back-barrier islands across the northern Gulf of Mexico were injured as a result of the DWH oil spill.
- Injuries resulted from a combination of the direct effects of oil and ancillary adverse impacts of response activities 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 2003). 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 2003). 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 2003). 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 4.6-61).

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 fresh water (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 DWH 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 DWH 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.

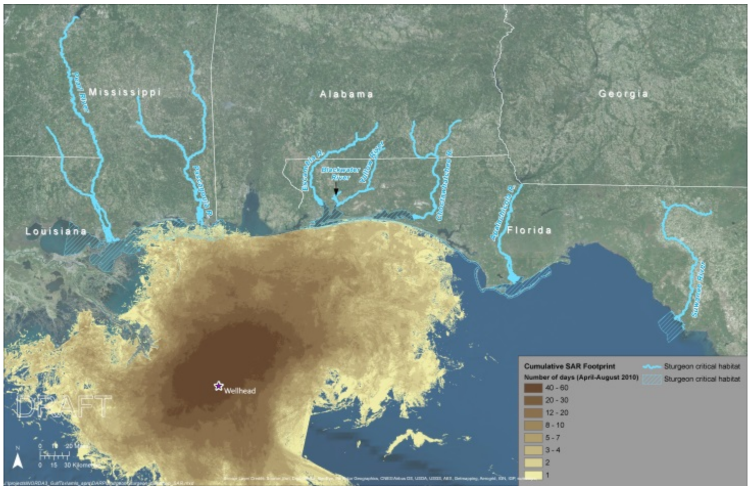
4.6.7.2 Pathways for Oil and Response Actions to Affect Gulf Sturgeon

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 2003). 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 4.6-61). 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 summer 2010 provided temporary refuge from exposure to oil associated with the DWH 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 DWH 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 4.6-62) (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.

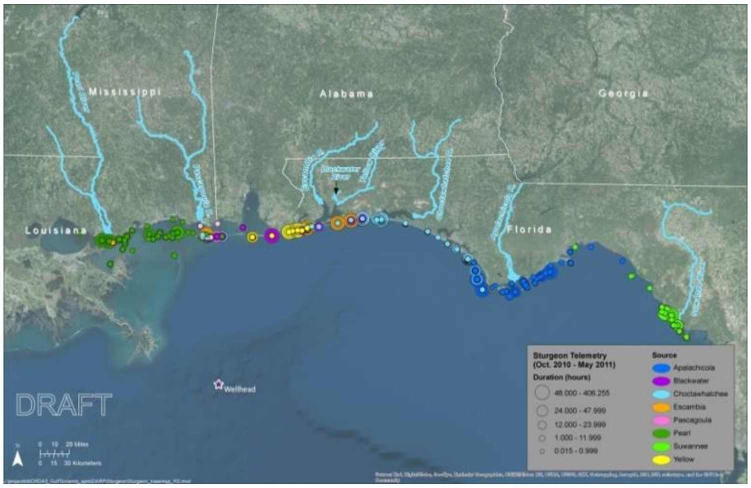
To determine exposure in the absence of quantitative, comprehensive substrate analyses or chemical body burdens for benthic macroinvertebrate prey items, the Trustees relied on available oil distribution data from before and during the residency time for Gulf sturgeon in the northern Gulf of Mexico (FWS 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 4.6-63) (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.



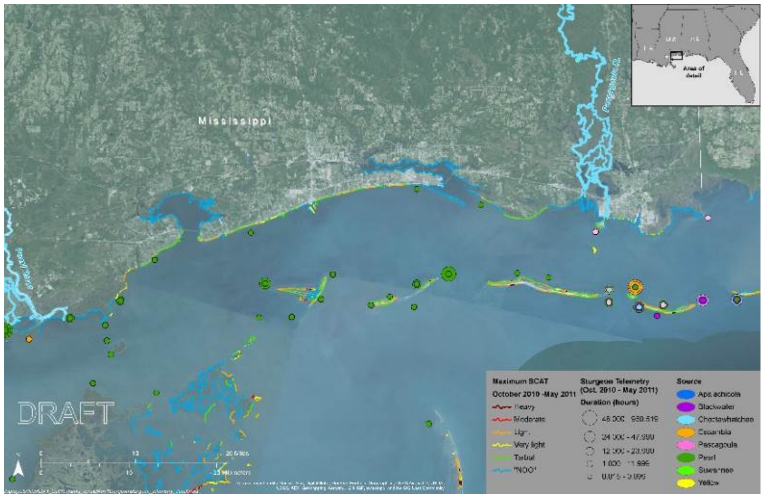
Source: *FWS (2015).*

Figure 4.6-59. 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 4.6-60. Distribution and residence time of Gulf sturgeon from fall 2010 to spring 2011, by source river.



Source: *FWS (2015).*

Figure 4.6-61. 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 DWH 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 DWH 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 DWH 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 DWH 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 6 out of every 10 (63 percent) fish from these six populations (Pearl, Pascagoula, Escambia, Blackwater, Yellow, Choctawhatchee) were potentially exposed to oil from the DWH 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 DWH 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 2003). 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 2003). During the spring months, Gulf sturgeon migrate up rivers to reach their spawning grounds (FWS & NOAA 2003).

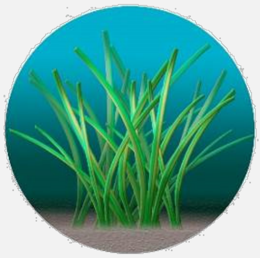
The Trustees considered all aspects of the Gulf sturgeon injury assessment in planning restoration for this endangered species. Key points that informed the Trustees' restoration planning include:

- 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 DWH oil.
- Between 1,100 and 3,600 Gulf sturgeon were estimated to be exposed to DWH oil in the nearshore areas of the northern Gulf of Mexico in fall 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).

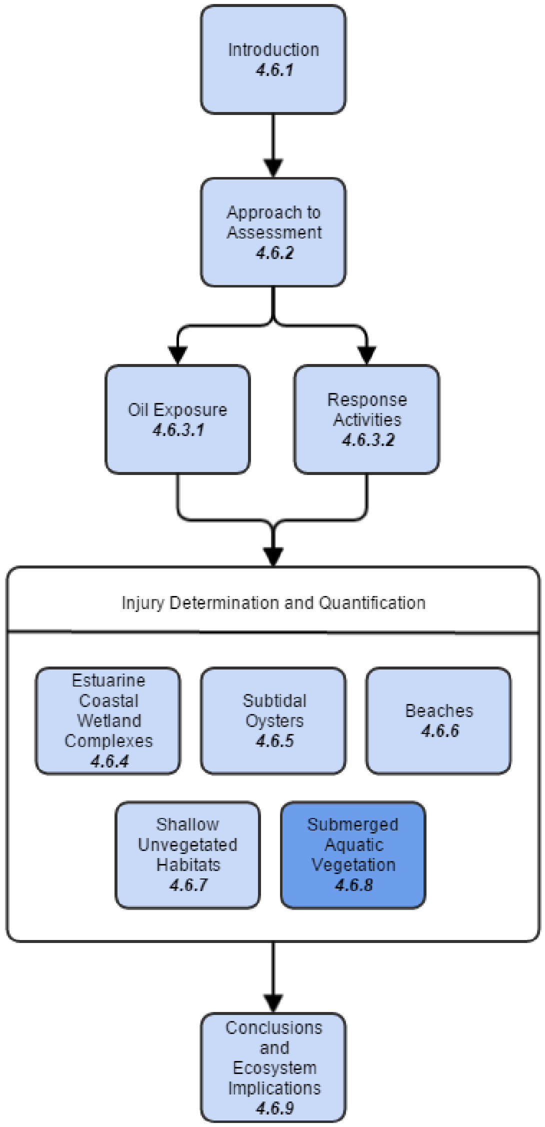
As described in Chapter 5 (Section 5.5.7), the Trustees have identified restoration approaches for this threatened species that emphasize spawning habitat and reproductive success.

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 fresh water 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 Historical Park and Preserve.

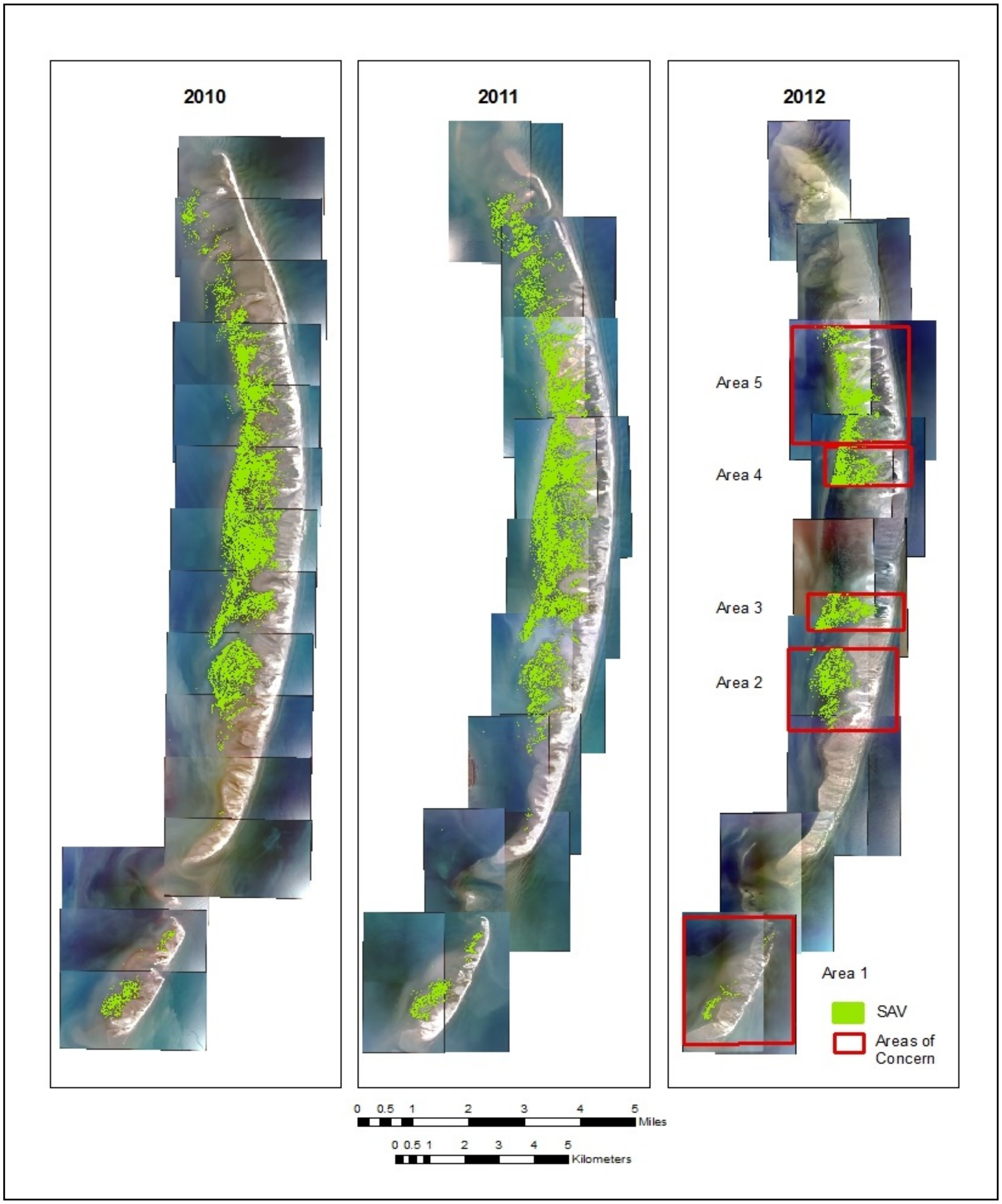
4.6.8.2 Injury Determination

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 4.6-64) (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 DWH 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; SAV tissue TPAH50 concentrations 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 4.6-62. 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 boom 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, boom, 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).

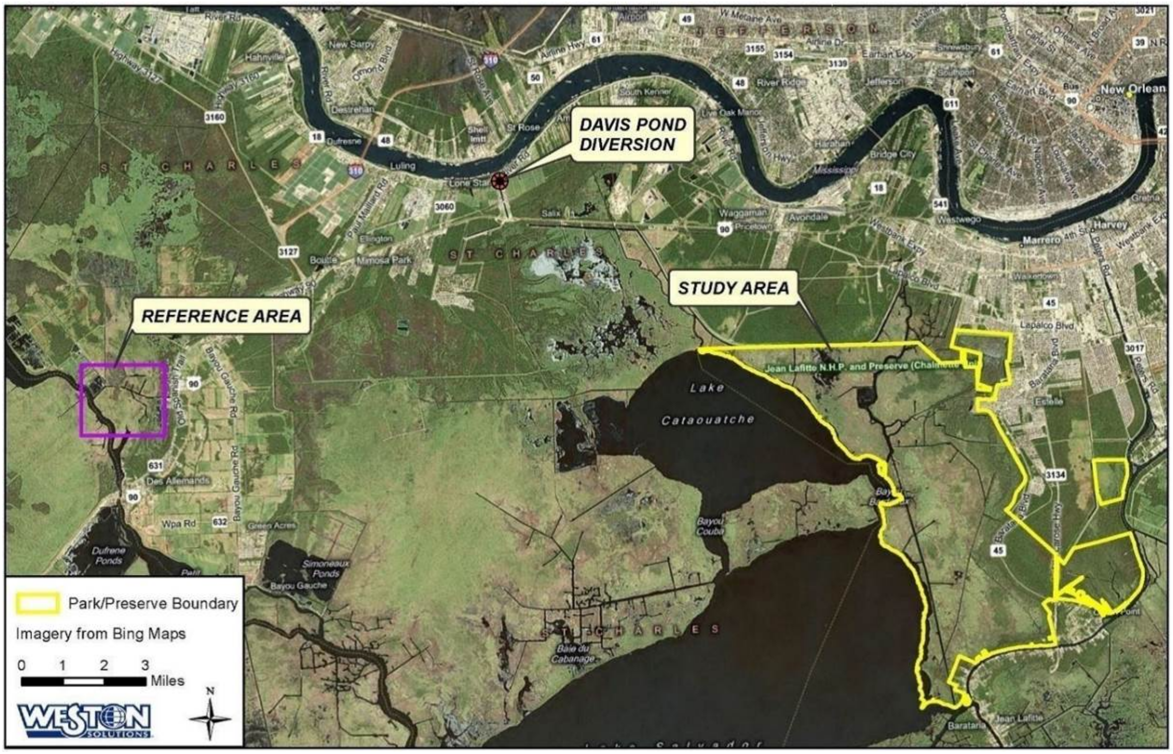
4.6.8.2.3 Freshwater Injury in Jean Lafitte National Historical Park and Preserve

In response to the DWH oil spill, Mississippi River water flows through the Davis Pond structure to Lake Cataouatche were increased during summer 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 2011 and 2012 (Figure 4.6-65) 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 five stations were sampled in Bayou des Allemands. Data were collected on water quality parameters,

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

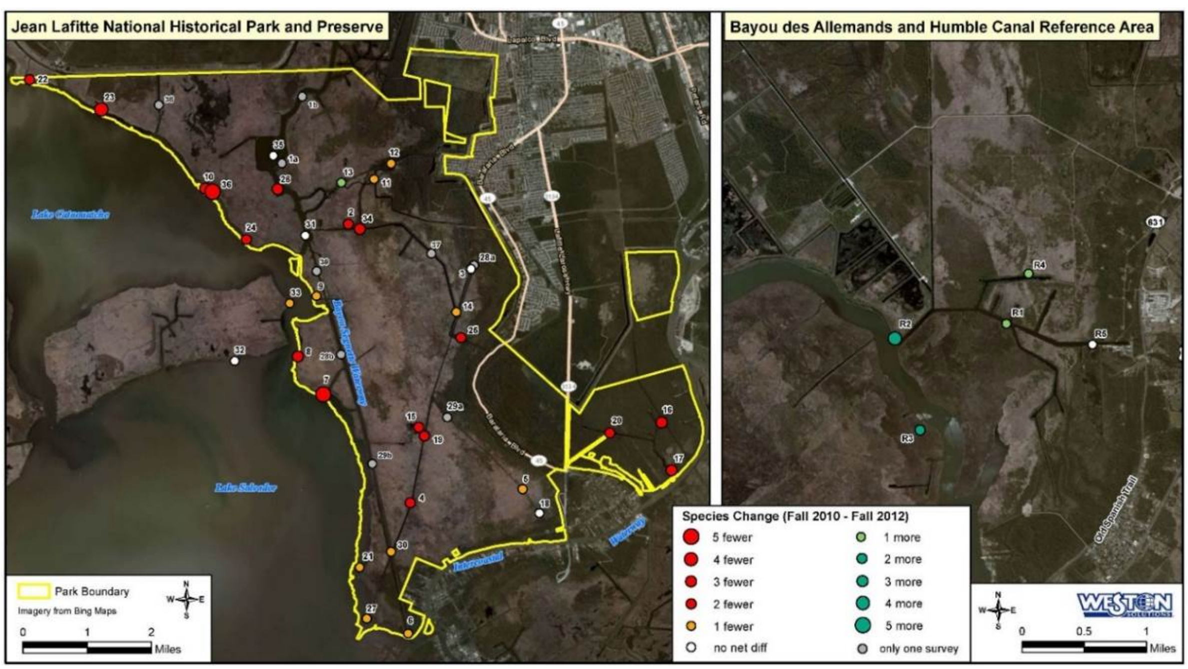
sediment and water nutrient levels, SAV community structure, and floating aquatic species abundance. Details of this study are provided in Weston Solutions Inc. (2015).

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 4.6-66) (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 4.6-67). 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.



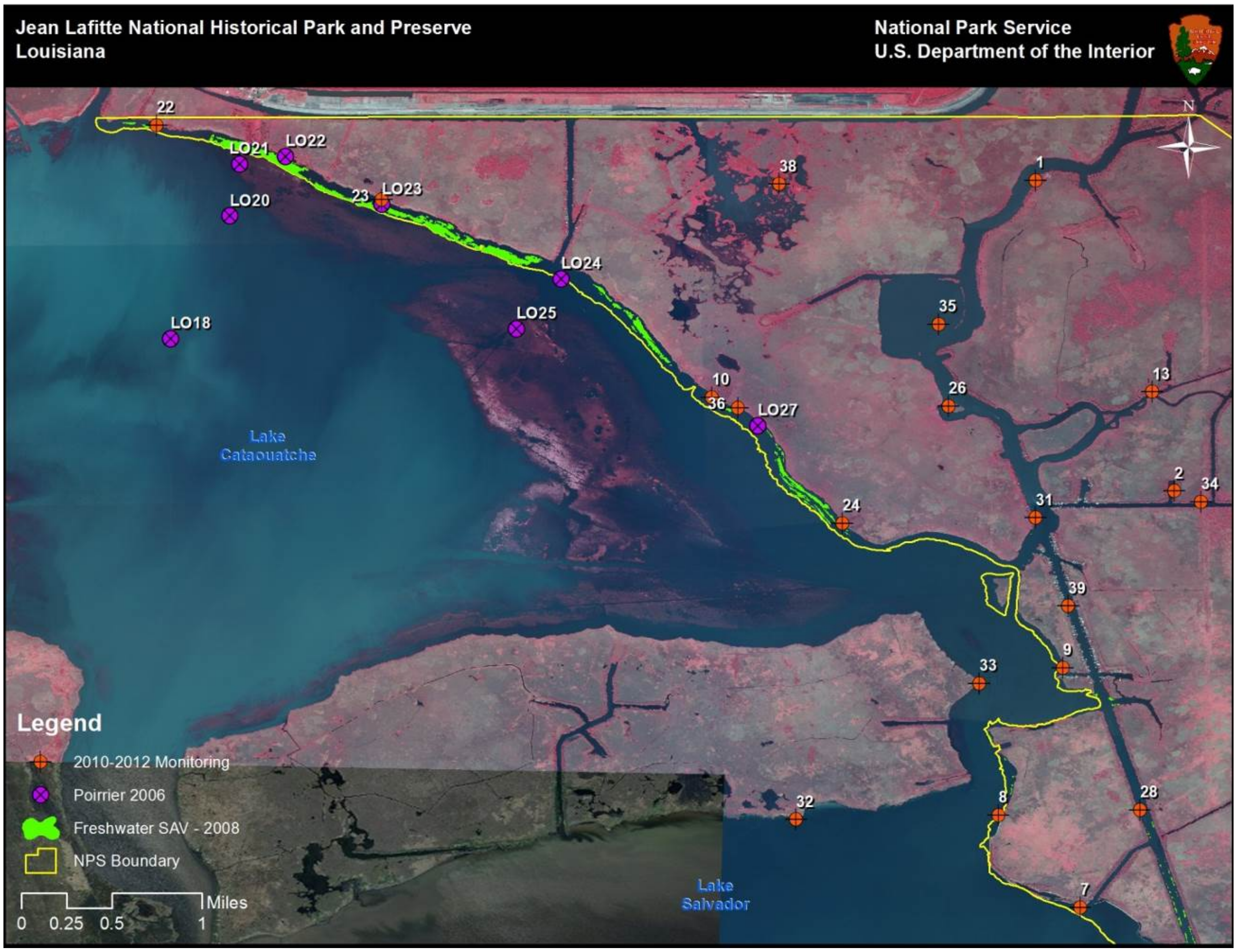
Source: Weston Solutions Inc. (2015).

Figure 4.6-63. Jean Lafitte National Historical Park and Preserve study area and stations.



Source: Weston Solutions Inc. (2015).

Figure 4.6-64. 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 4.6-65. 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 <

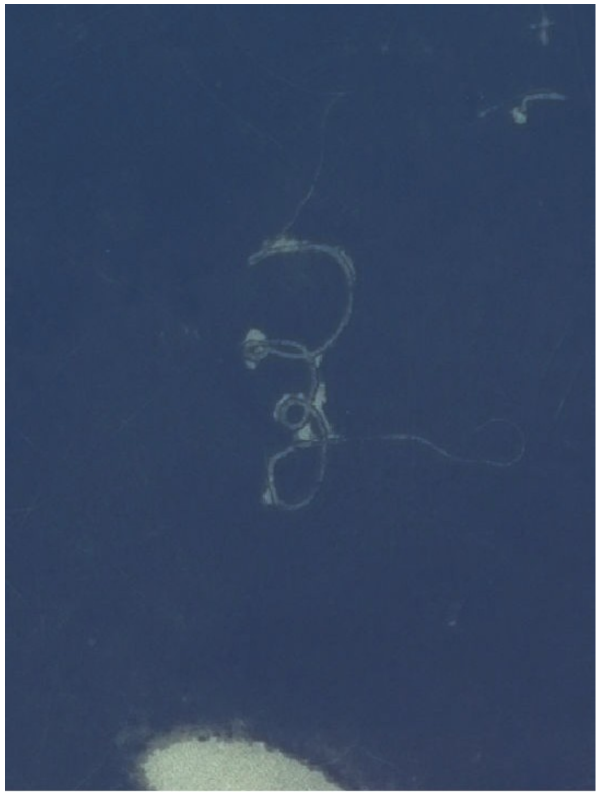
100 square meters were assumed to recover in 1 year. Overall, 51 acres of seagrass had persistent loss greater than 0.0247 acres (100 square meters); of these, 34 acres (14 hectares) were identified as having recovery times predicted to exceed 1 year. Predicted recovery times for these 34 acres (14 hectares) 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 4.6-68.



Source: *NOAA.*

Figure 4.6-66. 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 both the platform elevation and 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 DWH oil spill caused two types of injuries: propeller scars from response vessels and blowholes from vessels attempting to power off a shallow seagrass bed (Meehan 2015). The area and depth of propeller scars and blowholes 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:

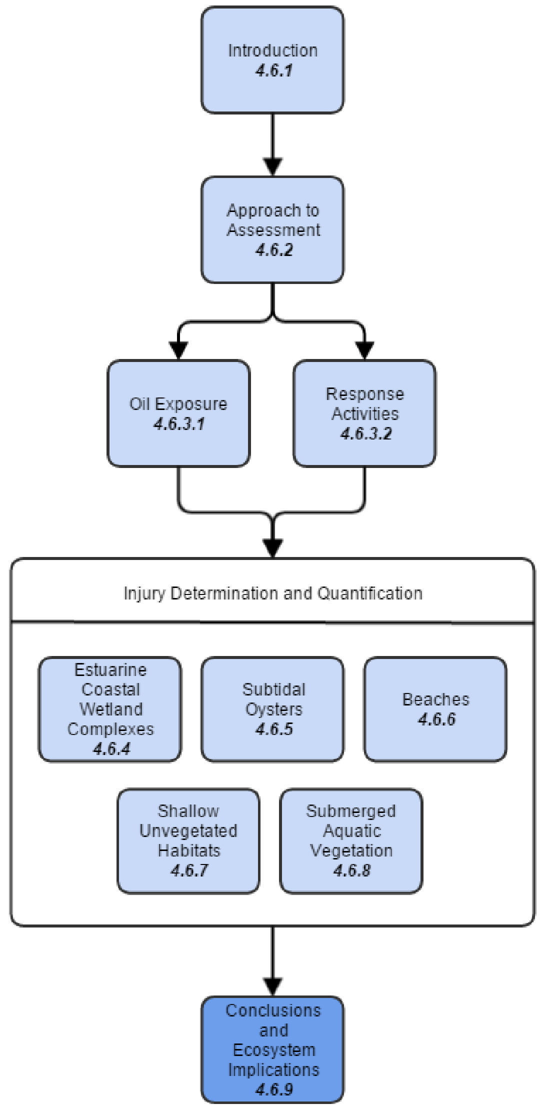
- SAV in the Chandeleur Islands, Louisiana, was injured as a result of oiling. The spatial distribution of seagrasses decreased from 2010 to 2012 along the shallow shelf west of the Chandeleur Islands. A total of 112 acres (45 hectares) of seagrass beds were identified as persistent loss (defined as 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 included propeller scars and blowholes 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 Historical Park and Preserve, Louisiana, was injured as a result of freshwater releases. Increased amounts of fresh water 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 November 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 DWH 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 DWH 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	-	-	-
SAV	-	-	-	20	-	-	-	-	-	.05

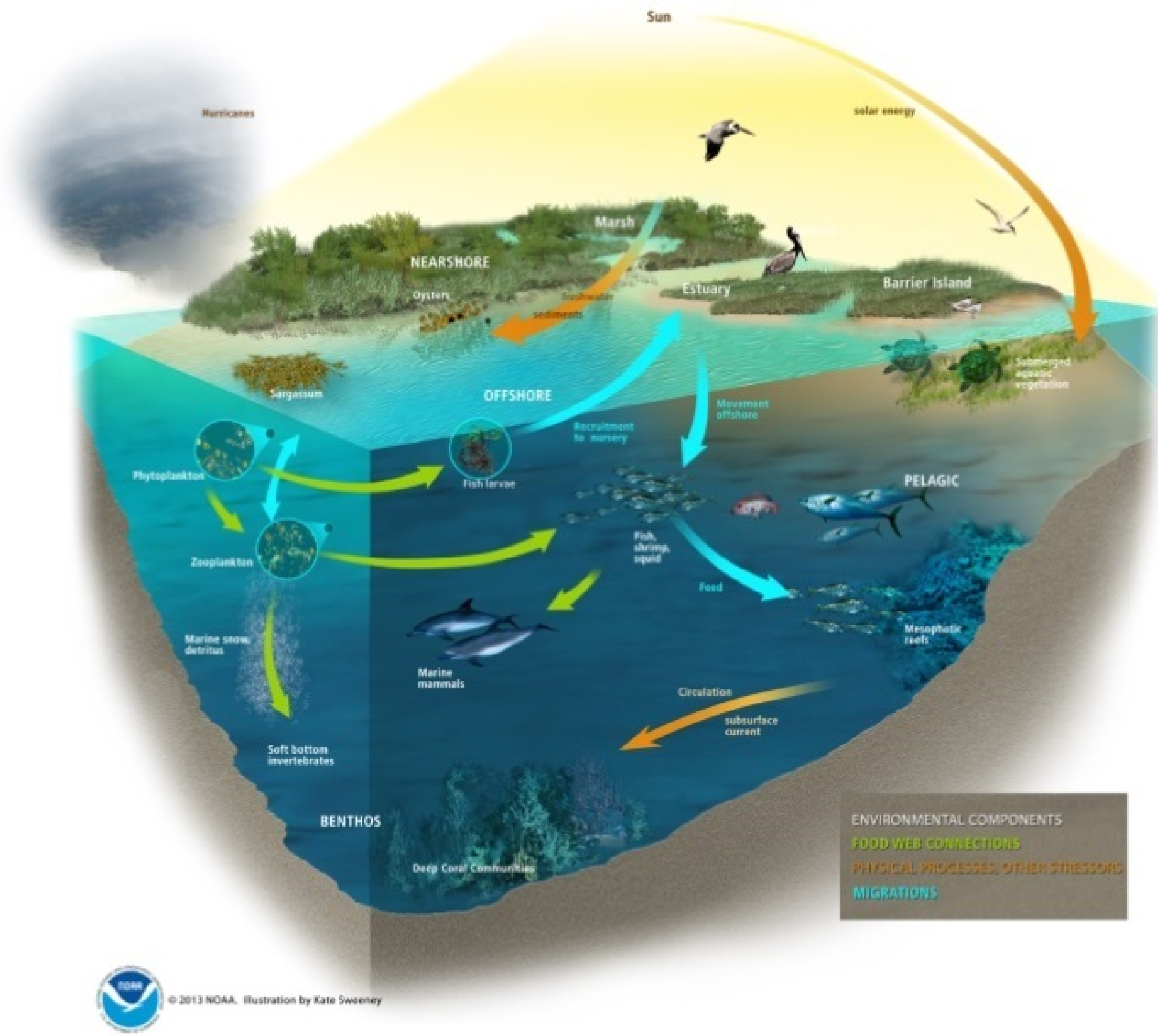
- Habitat was not present or was not measured by the specified metric.

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 4.6-69). 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 4.6-67. 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 aboveground 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 life cycles. 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 4.6-38). 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 DWH 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 life cycle 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); 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 diverse ecological services and benefits to a large variety of species.

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4.7 Birds

What Is in This Section?

- **Executive Summary**
- **Introduction and Importance of the Resource (Section 4.7.1):** Why do we care about birds and their habitats?
- **Approach to the Assessment (Section 4.7.2):** How did the Trustees assess injury to birds?
- **Exposure (Section 4.7.3):** How, and to what extent, were birds and their habitats exposed to *Deepwater Horizon* (DWH) oil?
- **Injury Determination (Section 4.7.4):** How did exposure to DWH oil affect birds?
- **Injury Quantification (Section 4.7.5):** What was the magnitude of injury to birds?
- **Conclusions and Key Aspects of the Injury for Restoration Planning (Section 4.7.6):** What are the Trustees' conclusions about injury to birds, ecosystem effects, and restoration considerations?
- **References (Section 4.7.7)**

Executive Summary

The Trustees documented large-scale and pervasive bird injuries in the northern Gulf of Mexico as a result of the DWH oil spill. This chapter describes the work conducted by the Trustees to determine and quantify injuries to birds resulting from the DWH spill.

Birds are highly valued and ecologically important components of the northern Gulf of Mexico ecosystem. This region supports a diversity of coastal bird species throughout the year, as nesting grounds during the summer, as a stopover for migrating species in the spring and fall, and as wintering habitat for numerous species that breed elsewhere. The DWH oil spill exposed dozens of species of birds to oil in a variety of northern Gulf of Mexico habitats, including open water, island waterbird colonies, barrier islands, beaches, bays, and marshes. Birds were exposed to oil in several ways, including physical contact with oil in the environment; ingestion of external oil during preening; and ingestion of oil while foraging and consuming contaminated prey, water, or sediment.

The Trustees conducted controlled laboratory evaluations of toxicological, metabolic, and physical responses to DWH oil exposure. These laboratory studies demonstrated that ingestion and external exposure to DWH oil caused an array of adverse effects, including anemia, weight loss, hypothermia, heart and liver abnormalities, feather damage, reduced flight capability, and death. These studies indicated the many ways in which birds that were exposed to DWH oil were affected and highlight how exposure led to reduced health and subsequently death for some birds.

The Trustees also conducted a series of studies to quantify bird injury from the DWH oil spill. Total quantified bird injury, including both mortality and lost reproduction, was estimated to be between 56,100 and 102,400 individuals of at least 93 species. These quantified injuries represent only a portion of the total bird injury, as these do not reflect all injury thought to have occurred to marsh birds and colonial waterbirds, as well as nonlethal injuries such as impaired health.

Field studies during the spill documented numbers and distributions of thousands of bird carcasses and oil-impaired live birds collected on beaches and marsh edges. Also, thousands of externally oiled, live birds were observed. In addition, surveys were conducted in offshore, open water habitats (greater than 25 miles [40 kilometers] from shore) to determine birds at risk from oil on the water surface. Based on these data and a series of models that use the data to generate mortality estimates, the Trustees estimated that mortality ranged from 51,600 to 84,500 individual birds. Although these estimates only addressed a portion of the bird mortality, uncertainties associated with the quantification approaches indicate that mortalities for this modeled injury were likely toward the higher end of this range. The Trustees also estimated the reproductive output lost as a result of breeding adult bird mortality; this was estimated to range from 4,600 to 17,900 fledglings that would have been produced in the absence of premature deaths of adult birds as a result of the DWH oil spill, after accounting for dead fledglings that were quantified using other methods. The Trustees determined that limitations and uncertainties would likely contribute to an overall underestimate of fledglings lost due to the spill. Given the available information, the results presented here are the best estimate of fledglings lost due to the spill, recognizing that the true loss is likely higher by some unquantifiable amount.

The quantified injury described above captured only a portion of overall injury to birds. DWH oil penetrated into marsh, which is important bird habitat. Exposure and mortality of interior marsh birds was not estimated by the Trustees; however, given densities of key species, meaningful injury to marsh birds was very likely to have occurred. Similarly, island waterbird colonies were occupied by hundreds of thousands of breeding birds at the time of the spill. Although some mortality in colonies was included in quantification, the Trustees recognize that these methods were inadequate for fully describing the magnitude of injury at colonies. In addition, bird injury almost certainly occurred in the forms of poorer health, protracted exposure, and delayed effects, none of which were quantified by the Trustees.

Birds are important components of marine ecosystems across the globe. They are highly responsive to variation in prey, and also exert top-down effects on the number and distribution of prey species. They also are abundant and have high metabolic rates, and thus exhibit high food consumption relative to other taxa, which increases their influence on marine communities. Birds also serve as prey for other species, and changes in the prey base could have effects on top level predators. The Trustees, therefore, expect that the loss of birds as a result of the DWH oil spill would have meaningful effects on food webs of the northern Gulf of Mexico.

4.7.1 Introduction and Importance of the Resource

Key Points

- Over 150 species of birds occur in waters and wetlands of the northern Gulf of Mexico for at least a portion of their lives; nearly 300 species use either open water, the coast itself, or coastal upland habitats directly adjacent to the Gulf.
- Birds are highly valued and ecologically important components of the northern Gulf of Mexico ecosystem, providing recreational, aesthetic, and economic value and playing vital roles in ecosystems by serving as both predators and prey in many food webs.
- The DWH oil spill affected numerous species of birds in four general habitat types in the northern Gulf of Mexico:
    - **Nearshore habitats** (including nearshore waters, beaches, and marsh edge) support a diversity of resident and migratory birds, including shorebirds, waterfowl, wading birds, and many others.
    - **Offshore/open water habitats** are used by birds that feed on fish and zooplankton near the water surface and by birds that use *Sargassum* mats as resting spots. Offshore birds include boobies, shearwaters, storm-petrels, and several species of terns.
    - **Island waterbird colonies** are used as nesting areas by a variety of species, including brown pelicans, laughing gulls, and terns. During the breeding season, a substantial proportion of birds in the northern Gulf of Mexico occur in coastal island waterbird colonies.
    - **Interior marshes** support numerous specialized resident and migratory birds, including clapper rails and seaside sparrows.

The DWH oil spill released more than 3 million barrels of oil into ecosystems of the northern Gulf of Mexico (Section 4.2, Natural Resource Exposure). Released oil from the spill contaminated extensive areas of nearshore, offshore, coastal island, and marsh habitats that support numerous bird species. As expected in a spill of this magnitude, birds and bird habitats were significantly affected (Figure 4.7-1). This section describes the array of exposure pathways and injuries documented by the Trustees and quantifies some components of injury to birds.

Sources: U.S. Department of the Interior (top left, top right, bottom left, bottom right) and Louisiana Department of Wildlife and Fisheries (bottom middle).

Figure 4.7-1. Examples of bird habitat contamination and bird injury resulting from the DWH oil spill. Top left: Oiled marsh habitat; Top right: Oiled sandy beach habitat; Bottom left: Oiled open water habitat; Bottom middle: Dead oiled bird on sandy beach; Bottom right: Live oiled bird captured for rehabilitation.

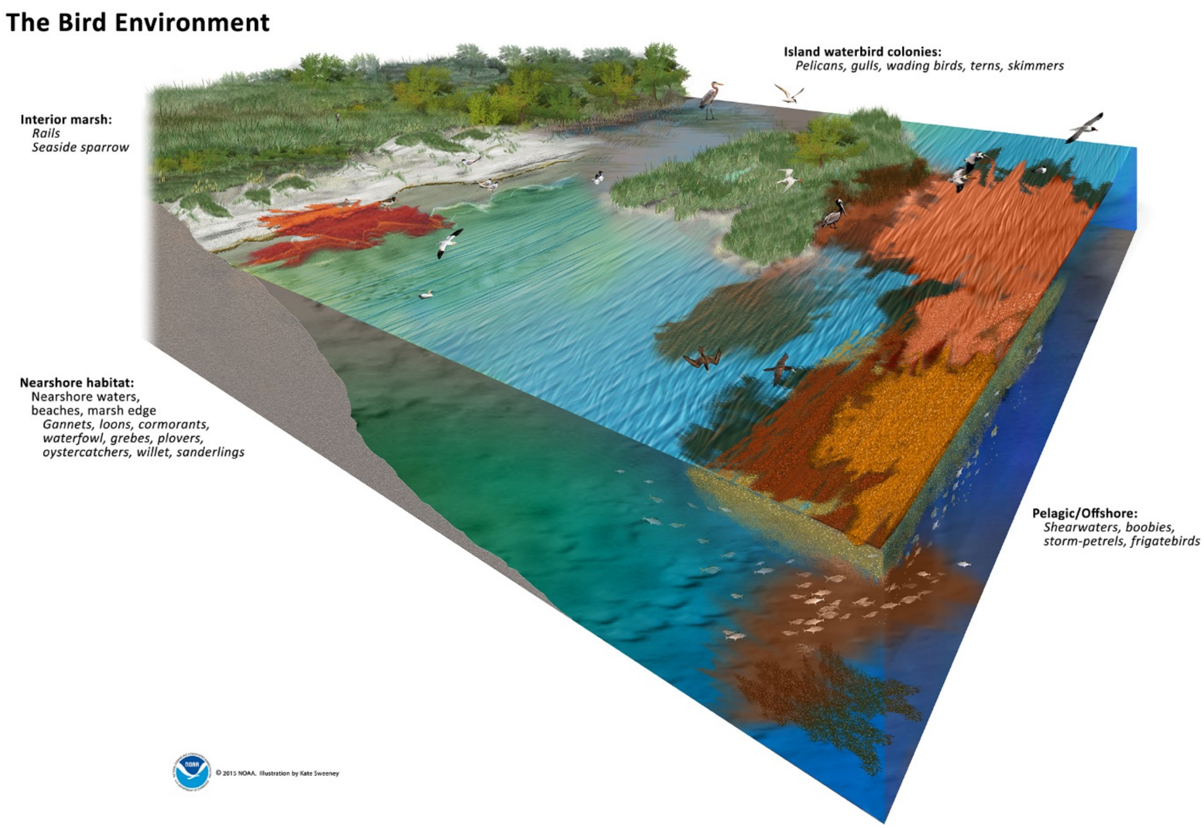
Oil spills are widely understood to injure birds. Examples include *Exxon Valdez* (Iverson & Esler 2010; Munilla et al. 2011; Piatt & Ford 1996), *Prestige* (Munilla et al. 2011), *Cosco Busan* (Cosco Busan Oil Spill Trustees 2012), *Luckenbach* (Luckenbach Trustee Council 2006), *Kure* (CDFG & FWS 2008), *New Carissa* (DOI et al. 2006), *Apex Houston* (CDFG et al. 2007; USFWS et al. 2011), and *Bean Stuyvesant* (CDFG et al. 2007). Accordingly, the Trustees conducted numerous studies to evaluate bird injuries resulting from the DWH oil spill.

Marine and coastal birds are highly susceptible to oil spill effects because of their use of the water surface, where oil tends to concentrate because of its buoyancy. Bird feathers absorb oil, which leads to ingestion through preening, loss of thermoregulation, and reductions in flight performance. Finally, birds are susceptible to ingestion of oil-contaminated prey, sediment, or water. Densities of birds are particularly high along the coastlines and marshes, where extensive oiling occurred and persisted.

Birds, including those inhabiting the northern Gulf of Mexico, have high societal value. Birds are easily recognized and valued members of coastal ecosystems, and injury to birds following oil spills invariably leads to immediate public demands for bird rehabilitation and restoration. In addition to their appeal to the general public, birds also have significant direct economic contributions. For example, both consumptive (migratory bird hunting) and non-consumptive (bird watching) activities generate billions of dollars annually in economic activity in the United States (FWS 2013). In addition to their recreational, aesthetic, and economic values, birds play vital roles in ecosystems, serving as both predators and prey in many food webs.

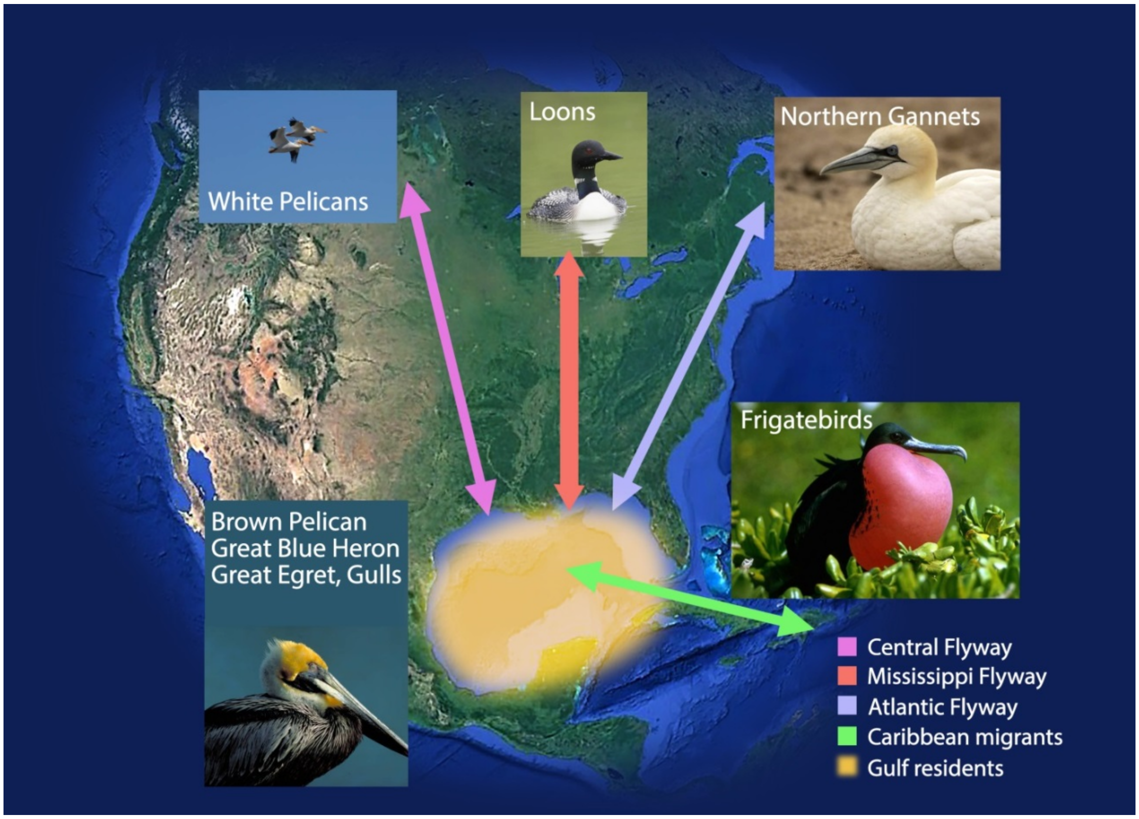
4.7.1.1 Bird Diversity and Habitats in the Northern Gulf of Mexico

The northern Gulf of Mexico consists of a variety of habitats that support a diverse and abundant assemblage of birds (Figure 4.7-2). Approximately 150 species of birds occur in waters and wetlands of the northern Gulf of Mexico for at least a portion of their lives and nearly 300 species use either open water, the coast itself, coastal marshes, or coastal upland habitats directly adjacent to the Gulf (e.g., coastal plain, cheniers, etc.). Depending on the species, birds use the northern Gulf of Mexico for their entire life cycle, as a migratory stopover, or as a wintering area. The northern Gulf of Mexico intersects with three of the four major migration flyways in North America, including the Central, Mississippi, and Atlantic Flyways (Figure 4.7-3).



Source: *Kate Sweeney for NOAA.*

Figure 4.7-2. Birds of the northern Gulf of Mexico occur in four general habitat types: nearshore, offshore, coastal islands with breeding colonies, and interior marsh, all of which were affected by the DWH oil spill. Examples of birds that occur in these habitats are given.



Source: *Kate Sweeney for NOAA.*

Figure 4.7-3. Where do birds injured by the DWH oil spill come from? Species listed are examples within each category.

There are four broad habitat types in the area of the northern Gulf of Mexico affected by the DWH oil spill (Figure 4.7-2). Each of these habitats is occupied by somewhat distinct bird assemblages (Table 4.7-1). Within the core impacted spill area, a number of national wildlife refuges, national parks, state parks, state wildlife Management Areas and Refuges, and other protected lands provide habitat for both resident and migratory bird species. Some of these public lands, such as Breton National Wildlife Refuge and Isle Dernieres Louisiana State Refuge, were created specifically for protection and conservation of birds.

Table 4.7-1. Bird habitats exposed to DWH oil in northern Gulf of Mexico.

Habitat Classification	Examples of Injured Species That Use Each Habitat
Offshore/Open Water	Shearwaters, storm-petrels, frigatebirds, terns
Nearshore	
Nearshore waters	Gannets, loons, cormorants, waterfowl, grebes
Beaches	Shorebirds, wading birds
Marsh edge	Gulls, pelicans, wading birds, shorebirds, black skimmers
Interior Marsh	Rails, seaside sparrows, waterfowl, wading birds
Island Waterbird Colonies	Pelicans, gulls, wading birds, terns, black skimmers

4.7.1.1.1 Nearshore Habitats

Nearshore habitats (waters, beaches, and marsh edges) of the northern Gulf of Mexico support a diversity of resident and migratory birds, including the federally endangered piping plover and the federally threatened red knot. Birds use multiple nearshore habitats (including shallow waters, beaches, and marsh edge) for nesting, feeding, and resting. Nearshore areas are important migration and wintering habitat for significant numbers of the continental waterfowl populations that use the Atlantic, Mississippi, and Central flyways (Figure 4.7-3). The Southeastern United States Regional Waterbird Conservation Plan identified nearshore habitats as among the most important for colonial birds, especially herons, ibises, pelicans, cormorants, skimmers, terns and gulls, and non-colonial birds such as rails (Hunter et al. 2006). It is also important for gannets, loons, shorebirds, and grebes. Oil from the DWH spill affected all nearshore habitats.

The nearshore marsh edge provides habitat for marsh-associated shorebirds, wading birds, gulls, terns, and other bird species. Marsh edge habitat also includes periodically exposed mudflats and tidal flats on the leading edge of marshes, which provide critical foraging areas.

Sandy beach habitats (primarily beaches, dunes, sand bars, and sandy inlet shorelines) provide services to numerous resident and migratory birds. They provide nesting areas for several solitary nesting shorebirds (e.g., American oystercatcher, snowy plover, and Wilson's plover), as well as colonial black skimmers, laughing gulls, and several species of terns.

4.7.1.1.2 Offshore/Open Water Habitat

Offshore birds heavily utilize open water environments. Offshore birds include boobies, shearwaters, storm-petrels, and several species of terns. Some of these species, such as Audubon's shearwater and masked booby, are frequently found in offshore areas of the northern Gulf of Mexico (Davis et al. 2000; Ribic et al. 1997), but do not nest within the northern Gulf of Mexico. Offshore birds feed in flight on fish and zooplankton as the prey swim to the surface. Free floating mats of *Sargassum* algae are also an important offshore habitat feature (Haney 1986). Offshore birds feed on fish and other organisms that these mats attract and also use *Sargassum* mats as resting spots. In offshore open water areas, birds interacted with and were injured by surface oil from the DWH spill (Section 4.2, Natural Resource Exposure).

4.7.1.1.3 Island Waterbird Colonies

Waterbirds use islands as nesting areas; when these birds occur in high densities, the nesting areas are called colonies. During the breeding season, a substantial proportion of birds in the northern Gulf of Mexico occur in colonies; these large aggregations (thousands to tens of thousands of adults, juveniles, and chicks) were susceptible to high levels of injury in cases where DWH oil was deposited in or near colonies. Many species, including brown pelicans, gulls, terns, and wading birds, nest colonially on coastal islands or in trees and shrubs over wetlands, and forage in adjacent shallow waters. Brown pelicans, often in mixed aggregations with wading birds, primarily nest on offshore islands where colonies are largely free from predation by terrestrial mammals and free of human disturbance. Wading birds are a diverse group of birds that use their physical adaptations to walk or wade in shallow water. They include the great blue heron, great egret, and snowy egret, as well as a number of other herons, egrets, and bitterns.

4.7.1.1.4 Interior Marsh Habitat

Coastal marshes, including those within the spill-affected area, support high numbers of birds throughout the year. Marshes are highly productive and serve as nursery habitats for many species of fish, shrimp, and invertebrates. This diversity and availability of prey attracts many bird species. Marsh birds include year-round residents, such as clapper rails, seaside sparrows, pied-billed grebes, common gallinules, least bitterns, marsh wrens, egrets, herons, ibis, and mottled ducks, as well as winter residents, such as long-billed curlews, soras, and many species of waterfowl (Woodrey et al. 2012). Oil that occurred on marsh edges, as well as oil that penetrated deeper into interior marsh habitats (Section 4.6, Nearshore Marine Ecosystem), contaminated habitat used by a variety of interior marsh birds.

4.7.2 Approach to the Assessment

Key Points

- The Trustees collected evidence demonstrating that birds were exposed to DWH oil in all coastal and open water habitats in which they occur.
- The Trustees conducted laboratory studies to evaluate physiological responses of birds exposed specifically to DWH oil.
- The Trustees conducted field studies to document numbers and distributions of bird carcasses and oil-impaired live birds.
- The Trustees used a number of methods to estimate bird mortality and lost reproduction as quantified injuries. The Trustees' assessment also included injuries that were not quantified but were significant.
- The Trustees used a "Shoreline Deposition Model" to quantify a portion of the nearshore bird mortality from April 20 to September 30, 2010—roughly when area-wide wildlife operations ceased. Because most dead or dying birds are never found, the model uses correction factors to account for several sources of loss of dead or impaired birds.
- The Trustees estimated bird mortalities in offshore open water habitat using an "Offshore Exposure Model," which determined the overlap between the distribution of oil and offshore birds and then estimated the degree of mortality.
- The Trustees also estimated mortality in areas that were not included in either the Shoreline Deposition or Offshore Exposure Models.
- The Trustees used a "Live Oiled Bird Model," which combined observations of rates and degrees of bird oiling with predictions of likelihood of mortality, to estimate a portion of nearshore bird mortality that occurred after September 30, 2010, for birds exposed to DWH oil through March 31, 2011.
- As the means to estimate lost productivity, the Trustees calculated the production of fledglings that would have occurred had breeding-aged birds not died. The Trustees applied species-specific annual productivity rates (average number of fledglings produced per

breeding pair) to the number of breeding-aged birds estimated through quantitative means to have died between April 2010 and April 2011.

- Although some portion of mortality that occurred at island waterbird colonies was quantified, the Trustees recognized that mortality on colonies was substantially higher than the quantified estimates.
- Mortality within marshes was not quantified. Using estimates of densities of key marsh bird species, the Trustees illustrated the substantial scope of potential exposure of marsh birds to oil.

Reflecting the magnitude of the DWH oil spill, the Trustees undertook extensive efforts to document and quantify injury to birds. Over the course of the bird injury assessment, thousands of researchers, agency staff, and volunteers conducted a broad range of activities, as illustrated in Figure 4.7-4. These activities occurred across thousands of kilometers of coastline and huge expanses of open water throughout the northern Gulf of Mexico. As a result of these efforts, more than 8,500 dead and impaired birds were collected. More than 3,000 live birds were taken to rehabilitation centers; despite tremendous effort, more than half of these were too compromised to survive. Recognizing that collected birds represent only a fraction of true mortality, significant efforts were directed toward quantifying a portion of the number of birds killed based on the data available. In addition, controlled laboratory studies were conducted to understand the array of avian health effects resulting from exposure to DWH oil.

The bird injury assessment following the DWH spill can be broken down into three inter-related categories of activities, which are described in detail in Sections 4.7.3, Exposure; 4.7.4, Injury Determination; and 4.7.5, Injury Quantification. First, information was collected confirming that birds were exposed to DWH oil (see Section 4.7.3). To evaluate the physiological, metabolic, thermoregulatory, and functional consequences of observed DWH oil exposure, a number of controlled laboratory studies were conducted using captive birds (see Section 4.7.4, Injury Determination) (Ziccardi 2015). Field measurements of physiological impairment were also evaluated. The Trustees documented health effects that were likely to result in increased rates of mortality. Using several different modeling approaches (see Sections 4.7.4, Injury Determination; 4.7.5, Injury Quantification), the Trustees also estimated a portion of the number of bird deaths as a result of the DWH oil spill (Table 4.7-2). For the bird mortalities that were quantified, the first year of lost reproduction of those birds was estimated for 2010 and 2011. In addition to quantified mortality and lost reproduction, the Trustees gathered information qualitatively indicating that additional mortality occurred in island waterbird colony and interior marsh habitats.

Table 4.7-2. Methods of assessment of bird mortality by time period and habitat (nearshore versus offshore). Includes both quantitative and qualitative means of assessing injury.

	20 April to 30 September 2010	1 October 2010 to 31 March 2011
Nearshore	• Shoreline Deposition Model • Excluded Regions • Colony Sweeps • Qualitative Assessment for Interior Marsh • Qualitative Assessment for Colonies (in addition to Shoreline Deposition Model and colony sweeps) • Qualitative Assessment of Response Impacts	• Live Oiled Bird Model
Offshore	Offshore Exposure Model	

Sources: *Louisiana Department of Wildlife and Fisheries (top left) and U.S. Department of the Interior (top right, bottom left, bottom right).*

Figure 4.7-4. Examples of field activities performed by the Trustees as part of the bird injury assessment; Top left: Searching for bird carcasses; Top right: Collecting live, oiled birds; Bottom left: Cataloging collected bird carcasses; Bottom right: Conducting observations of live, oiled birds.

4.7.2.1 Effects of Oil on Birds

Previous studies have shown that exposure to oil adversely affects birds in a variety of ways; this information informed the approach to bird injury assessment in this case.

Oil can cause feathers to lose their waterproofing and insulating ability, resulting in a bird not being able to swim or float and allowing water to penetrate and wet the skin (Helm et al. 2015). Reduced ability to swim and float increases the energy needed for swimming and diving; these increased energy requirements may not be sustainable. Oil-damaged feathers also impair a bird's ability to fly.

In addition to the physical effects of oil, birds in oiled environments also may consume oil-contaminated food, water, or sediments; ingest oil when preening; and inhale oil fumes (volatile aromatic compounds). Oil ingestion or inhalation can lead to adverse impacts, including inflammation, immune system suppression, and damage to cells (Briggs et al. 1996; Fry et al. 1986; Golet et al. 2002; Leighton et al. 1985). These in turn impact growth, alter organ function, reduce reproductive success, and likely increase risk of disease (Alonso-Alvarez et al. 2007; Briggs et al. 1996; Eppley & Rubega 1990; Esler 2000; Helm et al. 2015). Reproductive effects include adverse hormone changes, delayed egg laying, impaired egg formation, decreased eggshell thickness, and reduced hatchability. Avian embryos, especially very young ones, are very sensitive to crude oil and refined petroleum products when these substances get on egg shells. Oil can be deposited on eggs when adults build nests with oil-contaminated materials or when adults get oil on their feathers and carry it back to the nest. Embryos die not only because the oil covers the shell and suffocates the egg, but also because some of the oil penetrates through the shell and is toxic to the embryo. Numerous examples in the literature indicate high levels of bird mortality caused by oil spills, during both immediate and longer-term periods following spills; e.g., Iverson and Esler (2010); Piatt and Ford (1996).

The Trustees conducted a number of controlled laboratory studies in which exposure to DWH oil, both through ingestion and external exposure on feathers, caused a number of adverse effects on bird health and survival—see Section 4.7.3.3, Consequences of Exposure, and Ziccardi (2015). Some effects observed in the laboratory were directly associated with bird deaths, and other documented effects were severe enough that they would be expected to cause increased mortality in wild birds. Understanding specific health effects of exposure provides an important link for understanding mechanisms that lead from oil exposure to mortality and reduced reproduction endpoints (Figure 4.7-5). They also highlight the potential for significant health effects that likely affected numerous birds that did not die during the first year after the spill.

The primary measures of bird injury for this assessment are mortality and reduced reproduction. These are factors known to be consequences of oil exposure experienced by birds following oil spills (see above). Effects of mortality and reduced reproduction can be expressed as numbers of birds removed from the ecosystem or never fledged, in the case of lost reproduction. Bird injury was assessed both quantitatively and qualitatively. Quantified injuries included birds killed in offshore habitats, a portion of the birds killed in nearshore habitats, a portion of the colonial birds that were killed, and reproductive losses in 2010–2011 resulting from the quantified bird mortalities. Unquantified injuries included other island colony birds, interior marsh birds, effects on bird health (including associated reduced survival rates after 2010), and impacts of response activities.

1. Collecting Evidence of DWH Oil Exposure

- Documentation of oil in bird habitat.
- Collection of oiled birds, dead and live.
- Observations of live, visibly oiled birds.

↓

2. Measuring Health Effects of Oil Exposure

- Lab studies of DWH oil toxicity.
- Lab studies of effects of external DWH oil.
- Field measurement of physiological impairment.

↓

3. Estimating Mortality and Reproduction Consequences

- Shoreline Deposition Model estimating a portion of nearshore bird deaths (April–September 2010).
- Offshore Exposure Model estimating offshore bird deaths (April–September 2010).
- Live Oiled Bird Model estimating a portion of nearshore bird deaths (due to oiling between September 2010–March 2011).
- Lost reproduction as a result of 2010 and 2011 breeding bird mortality.
- Unquantified assessment of additional colony mortality.
- Unquantified assessment of interior marsh bird mortality.

Source: *U.S. Department of the Interior.*

Figure 4.7-5. A description of the types of activities conducted during the bird injury assessment: 1) Collecting evidence of DWH oil exposure; 2) Measuring health effects of oil exposure; 3) Estimating mortality and reproduction consequences.

Additional information addressing avian injury following the DWH oil spill has been reported independent of the Trustees' assessment activities (Belanger et al. 2010; Bergeon Burns et al. 2014; Finch et al. 2011; Franci et al. 2014; Haney et al. 2014a, 2014b; Haney et al. 2015; Henkel et al. 2012, 2014; Montevecchi et al. 2011; Paruk et al. 2014; Sackmann & Becker 2015; Seegar et al. 2015; Walter et al. 2014). The Trustees have reviewed these publications and considered their findings as part of the bird injury assessment for the DWH oil spill.

4.7.3 Exposure

Key Points

- Oil released during the DWH oil spill contaminated open water, coastal islands, beaches, bays, and marshes. These habitats are used by more than 150 species of birds.
- Birds were exposed to oil through physical contact with oil in the environment; subsequent ingestion of external oil during preening; and ingestion of oil through consumption of contaminated prey, water, or sediment.
- More than 8,500 dead and impaired birds were collected during and following the spill. Of collected birds, more than 3,000 live individuals were taken to rehabilitation centers; more than half of these were too compromised to survive.
- Over 60 percent of captured, live, impaired birds had evidence of external oiling.
- More than 3,500 uncollected birds were observed with visible external oiling.

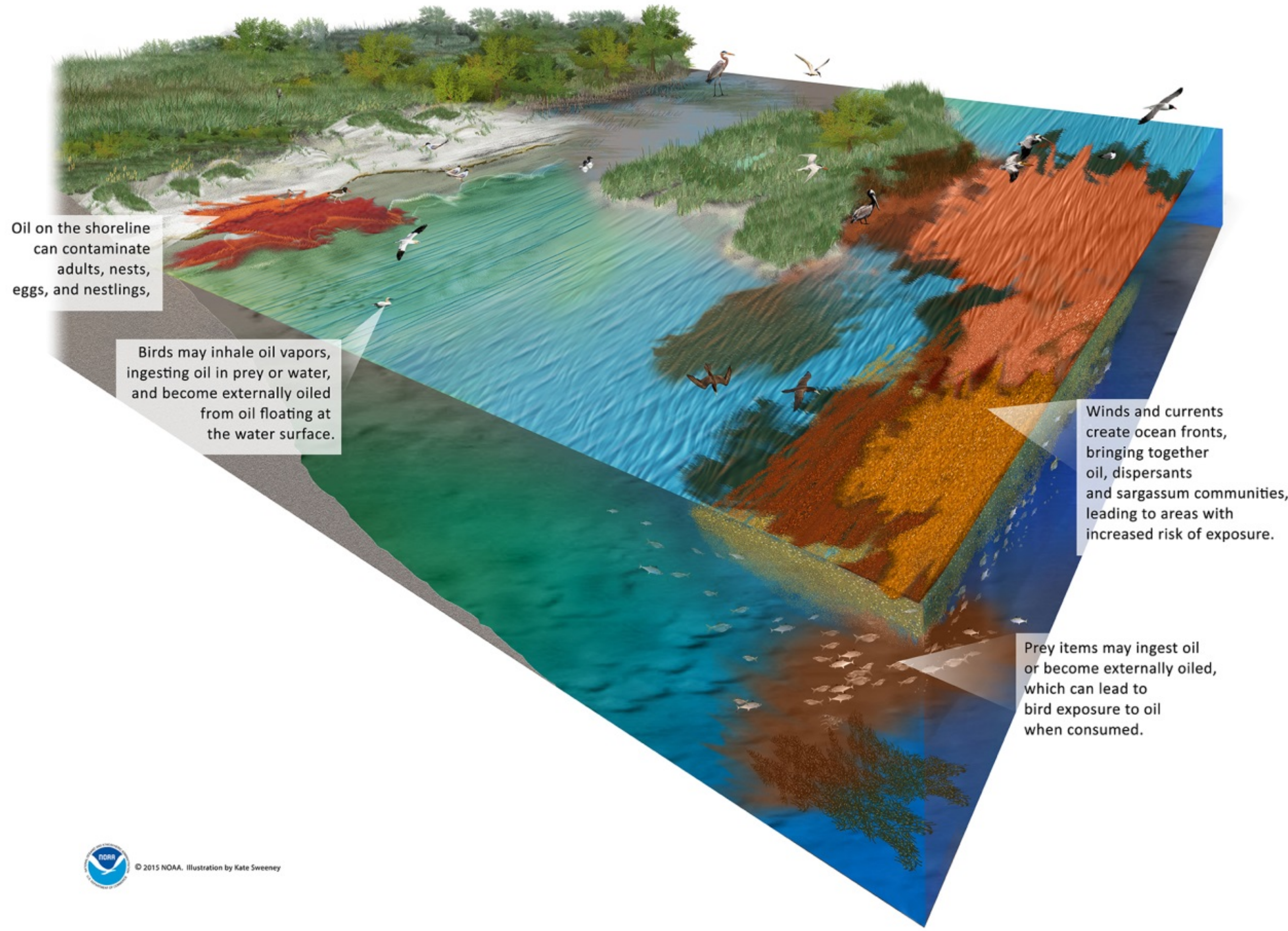
4.7.3.1 Distribution and Duration of DWH Oil in Bird Habitats

As described in Section 4.2 (Natural Resource Exposure), DWH oil contaminated the water surface (where birds rest and feed), the air (where birds fly and breathe), and various coastal habitats (where birds feed, roost, and nest). Oil was discharged into the environment over 87 consecutive days, resulting in a protracted period of habitat contamination and subsequent bird exposure. All the main bird habitat types described above in Section 4.7.1.1 (Bird Diversity and Habitats in the Northern Gulf of Mexico) were exposed to oil, leading to direct and indirect exposure of associated bird communities. DWH oil occurred cumulatively over 112,000 square kilometers of ocean surface during the course of the spill, exposing offshore birds to floating oil and oiled *Sargassum*. As the oil moved into nearshore habitats, a broad suite of birds became exposed. At least 2,113 kilometers of shoreline were estimated to have been oiled (Section 4.6, Nearshore Marine Ecosystem). Oil occurred in all nearshore habitats, including beaches, bays, marsh edges, and island waterbird colonies (Figure 4.7-6). Oil also penetrated into marshes, exposing a variety of bird species.

There were a number of pathways of oil exposure identified for birds (Figure 4.7-7). These included direct contact with oil in contaminated habitats, ingestion of oil during preening of external oil, ingestion of oil when foraging or drinking, as well as inhalation of oil vapors. Based on previous spills, these are well-known routes of exposure that are known to result in significant health and demographic consequences for exposed individuals (Section 4.7.3.3, Consequences of Exposure).

Source: Louisiana Department of Wildlife and Fisheries.

Figure 4.7-6. Contamination of island waterbird colony habitat by DWH oil. Photo shows royal tern adults and chicks and contaminated shoreline at Queen Bess Island, Louisiana.



Source: *Kate Sweeney for NOAA.*

Figure 4.7-7. Routes of exposure of birds to DWH oil. Text boxes highlight specific details about potential exposure pathways and adverse effects to birds.

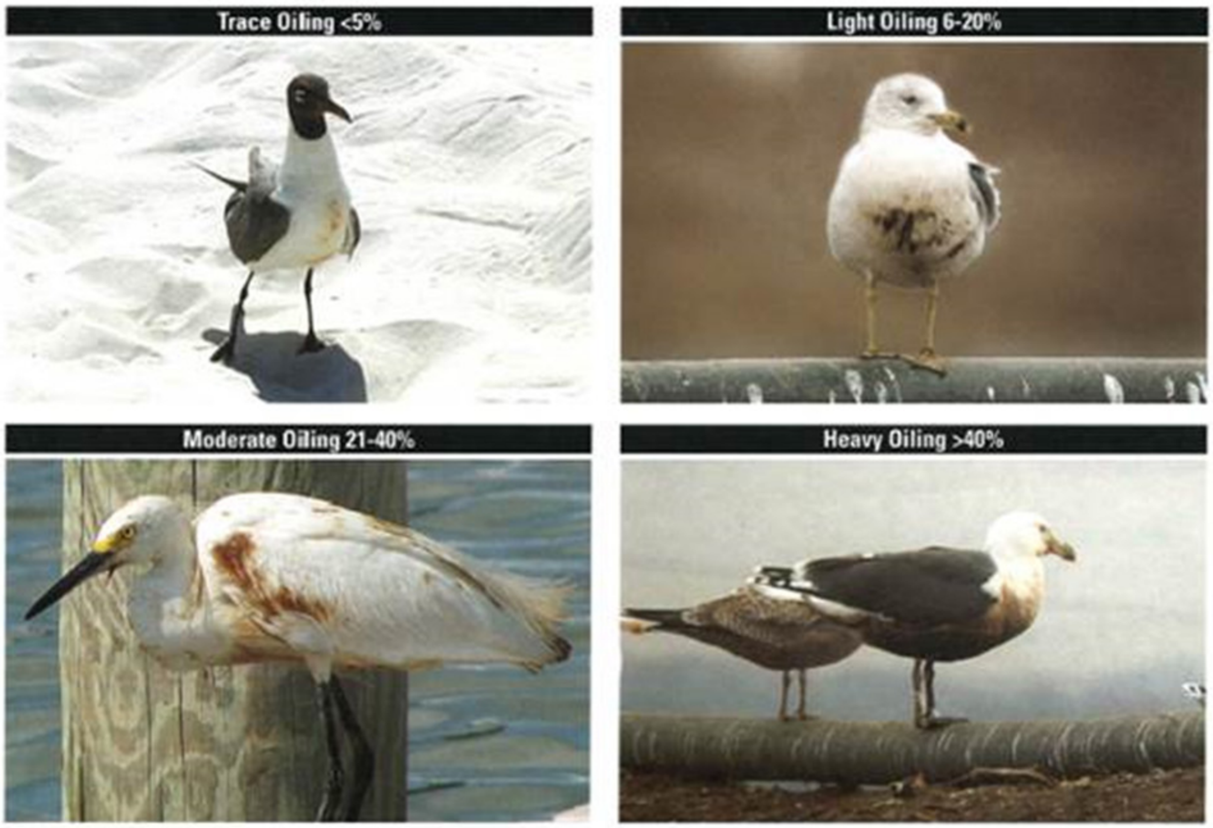
4.7.3.2 Evidence of Exposure

In addition to the spatial overlap of DWH oil with birds and their habitats described above, evidence of oil exposure included observation and collection of thousands of visibly oiled birds. Of the dead birds collected during the spill, a substantial proportion (greater than 30 percent) were visibly oiled (Figure 4.7-8). Similarly, over 60 percent of captured, live, impaired birds had evidence of external oiling. Collected or captured birds without visible external oiling were still very likely exposed to oil; this could have been the result of oil vapor inhalation, ingestion of contaminated prey, removal of oil by preening prior to collection (as observed in laboratory studies), or simply that external oil was not noted (e.g., on birds with dark plumage). Also, dead birds may have been too decomposed to determine oiling status.

Source: *U.S. Department of the Interior.*

Figure 4.7-8. Examples of live (top) and dead (bottom) oiled birds collected following the DWH oil spill.

Along with collection of oiled birds, a significant effort was undertaken to survey birds and document rates of visible external oiling (i.e., the number of birds with visible oil relative to the overall number of birds observed), as well as the degree of visible oiling. These efforts documented over 3,500 individuals with visible external oiling during the year following the spill (FWS 2015e). Observed external oiling ranged from trace to heavy (Figure 4.7-9). The proportion of birds with observable oil, as well as the intensity of oiling, declined through time after the well was capped.



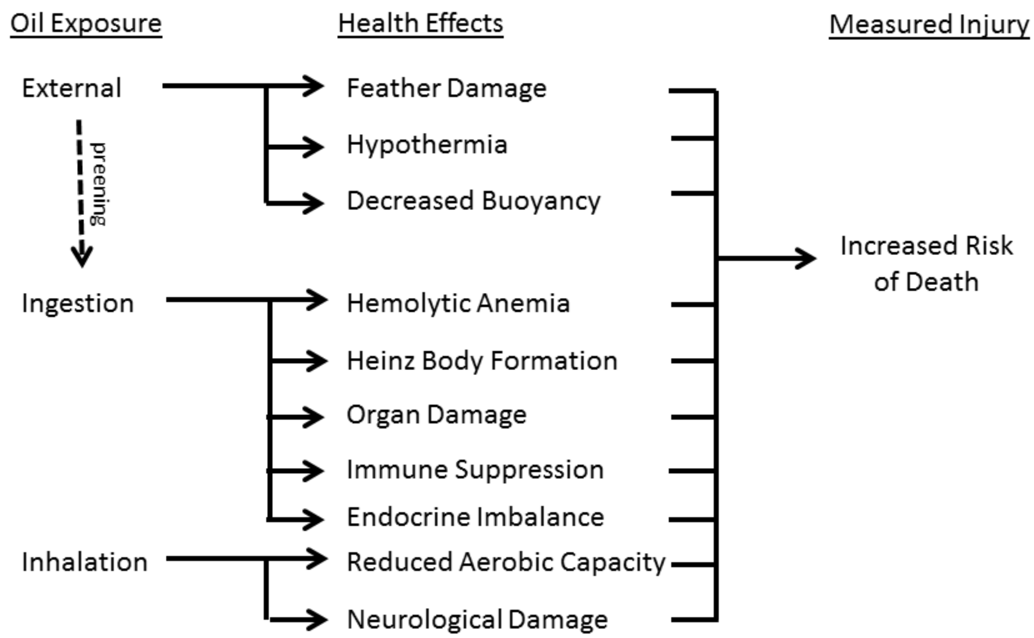
Source: U.S. Department of the Interior.

Figure 4.7-9. Categories of oiling intensity used during surveys to document rates and degree of external oiling of birds in the northern Gulf of Mexico.

4.7.3 Exposure

4.7.3.3 Consequences of Exposure

As described above, there are multiple ways birds were exposed to DWH oil. The flow chart below (Figure 4.7-10) describes examples of adverse health effects resulting from different pathways of exposure and indicates that each of these can lead to increased mortality and subsequent lost reproduction, which are the primary metrics of bird injury in this assessment. Health effects and injury are described in detail in Sections 4.7.4 (Injury Determination) and 4.7.5 (Injury Quantification).



Source: *U.S. Department of the Interior.*

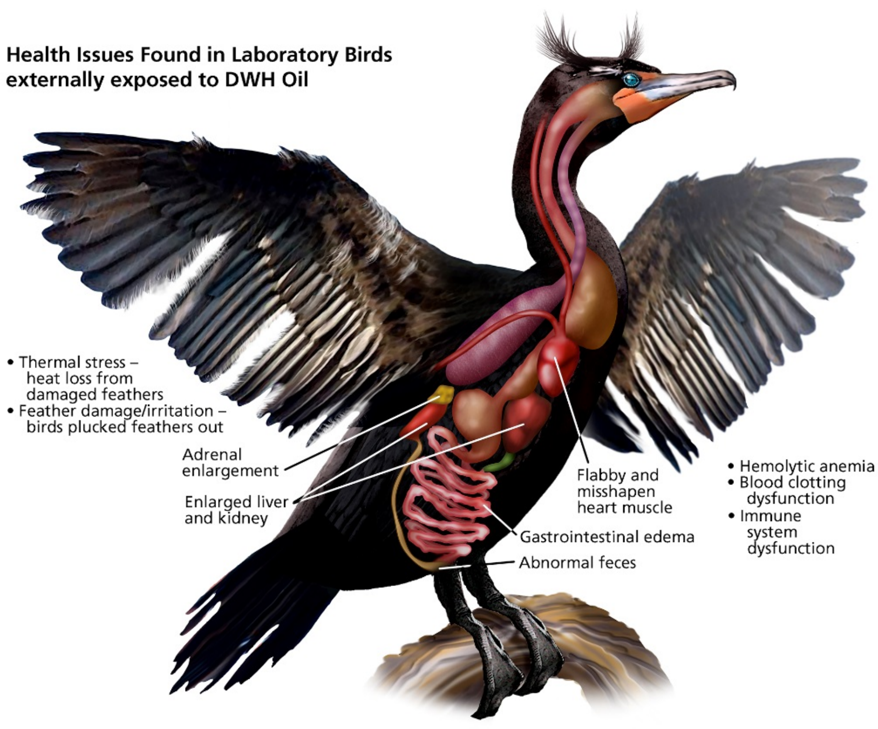
Figure 4.7-10. Conceptual pathways leading from the various types of oil exposure, through associated health effects, to the mortality endpoint used to quantify bird injury following the DWH oil spill. This figure does not capture all potential injuries, including sublethal effects, but illustrates ways in which quantified injury may have resulted from exposure.

4.7.3 Exposure

4.7.4 Injury Determination

Key Points

- The Trustees conducted a number of laboratory studies that demonstrated a suite of negative physiological effects on birds exposed to DWH oil (Figure 4.7-11).
    - Physiological effects of ingestion of DWH oil included disruption of reproductive function; anemia; changes in immune function; reduced kidney, liver, and gastrointestinal function; and heart abnormalities.
    - Physical effects of plumage oiling included structural damage to feathers, leading to impaired flight capability and behavioral alterations, thermoregulatory impairment, and loss of buoyancy.
- The impairments identified through controlled studies undoubtedly occurred in wild, oil-exposed birds and resulted in increased rates of mortality.
- Elevated mortality in wild birds was abundantly evident by the more than 8,500 dead and oil-impaired birds recovered following the DWH oil spill. These recovered birds represent only a fraction of overall mortality. The Trustees used a number of quantitative and qualitative methods to describe the magnitude of bird mortality and associated lost reproduction.



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

Figure 4.7-11. Schematic showing the array of negative effects experienced by birds following external exposure or ingestion of DWH oil.

Studies of effects of oil on bird health were conducted for two purposes: 1) to understand the types and degrees of negative effects occurring in wild birds exposed to DWH oil, and 2) to use that information to inform estimation of the fate of birds adversely impacted by exposure to DWH oil, in particular within the Live Oiled Bird Model (Section 4.7.5.1.2, Avian Mortality After September 2010) and the Offshore Exposure Model (Section 4.7.5.1.1, Avian Mortality Between April and September 2010). These effects are described in a summary report (Ziccardi 2015) and supported by technical reports cited therein (Bursian et al. 2015a; Bursian et al. 2015b; Dorr et al. 2015; Fallon et al. 2014; IEc 2015b; Maggini et al. 2015; Pritsos et al. 2015); information below is a synopsis of that document.

4.7.4.1 Physical Effects of External Oil

Thousands of birds in the northern Gulf of Mexico were externally contaminated by DWH oil (Section 4.7.3, Exposure). Conclusions of laboratory studies conducted by the Trustees were consistent with information from the literature, indicating that external oil exposure had significant effects on feather structure and function (Holmes & Cronshaw 1977; Leighton 1993). Because feathers are the primary insulation for birds, breakdown of feather structure can result in thermoregulatory challenges. If birds cannot catch enough food to meet increased energy costs, they will exhaust their energy stores and become hypo- or hyper-thermic, which results in death.

In unoiled birds, feathers form a waterproof layer that traps air and provides buoyancy (Helm et al. 2015). When feather damage occurs following external oil exposure, that buoyancy is lost, leading to significantly reduced capacity to swim or float in water, which in turn reduces birds' ability to forage and escape predators (Maggini et al. 2015; Pritsos et al. 2015; Ziccardi 2015).

Trustee laboratory studies demonstrated that damage to feathers associated with external oiling also caused significant alterations in flight ability, manifested by decreased takeoff speed, reduced takeoff angle, decreased endurance during flight, and longer flight times. These alterations in flight capabilities can directly cause a number of harmful outcomes, including an inability to evade predators, reduction in energy stores, and delayed arrival at breeding grounds (Maggini et al. 2015; Pritsos et al. 2015; Ziccardi 2015).

External oil also affects skin, mucus membranes, and other sensitive tissues, causing irritation, burning, and permanent damage or loss of function, manifested by the inability to hear or see normally and/or the presence of inflamed, ulcerated, thickened, or sloughing skin (Dorr et al. 2015; IEc 2015b; Ziccardi 2015). These multiple health consequences resulting from external oiling led to increased mortality risk.

4.7.4.2 Physiological Effects of Oil Ingestion and Inhalation

As described in Section 4.7.3 (Exposure) and summarized in Ziccardi (2015), ingestion of DWH oil occurred through preening oil from feathers, as well as through feeding or ingestion of contaminated water or sediment. Physiological effects also can result from absorption of toxic components of oil through the skin. The available literature shows that oil ingestion leads to a variety of negative effects for birds, and laboratory studies conducted by the Trustees confirmed those effects and revealed additional, previously unknown, harmful consequences of oil ingestion (Figure 4.7-10 and Figure 4.7-11). Health effects of oil exposure were likely additive, as exposure occurred through multiple pathways for some individuals, and multiple health effects were likely induced.

Trustee studies and previously published work document significant alterations to red blood cells upon ingestion of petroleum hydrocarbons, including DWH oil (Bursian et al. 2015a; Bursian et al. 2015b; Fallon et al. 2014; IEc 2015b; Leighton 1993; Ziccardi 2015). This results in reductions in oxygen-carrying capacity of the blood. In turn, this can have significant effects on bird behavior, constraining their ability to fly, swim, and forage, with subsequent increased risk of death.

White blood cells (leucocytes) also were altered by ingestion of DWH oil (Bursian et al. 2015a; Bursian et al. 2015b; Ziccardi 2015). This would be expected to reduce birds' abilities to combat bacterial, fungal, viral, or parasitic infections—increasing energetic costs and risk of death.

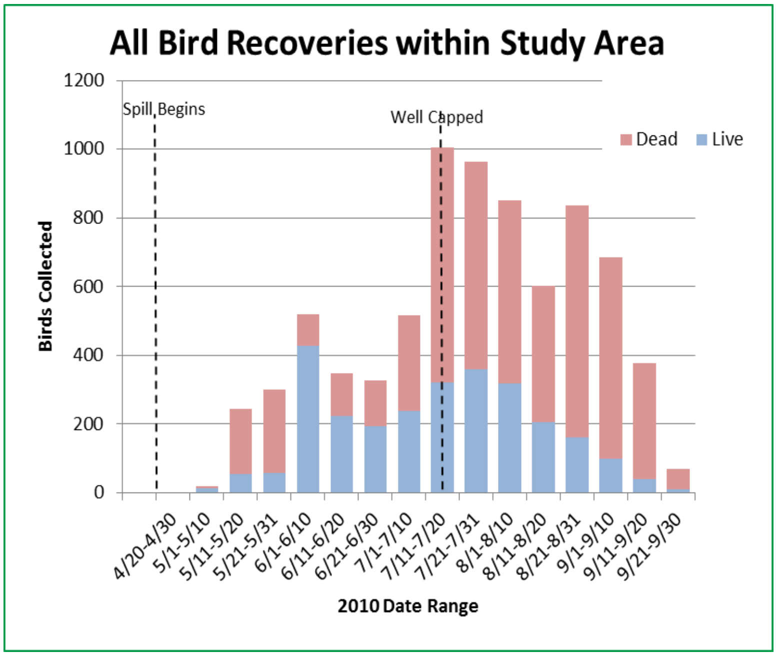
Several types of organ damage were observed in Trustee laboratory studies, including liver, kidney, and gastrointestinal systems (Bursian et al. 2015a; Bursian et al. 2015b; Dorr et al. 2015; Ziccardi 2015). In addition, Trustee studies found previously undescribed alterations in heart morphology and function following DWH oil ingestion. Overall, disruption of organ physiology and function would contribute to increased mortality rates.

Although the Trustees did not directly evaluate health effects of inhalation by birds of the volatile components of DWH oil, the existing literature indicates that PAH inhalation can cause significant alterations in neurological and respiratory function (Helm et al. 2015; IEc 2015b). Resultant behavioral modifications and constraints on birds' abilities to fly, swim, and dive would increase risk of mortality (Helm et al. 2015; IEc 2015b).

4.7.4.3 Effects of Oil on Bird Survival and Reproduction

Physical and physiological effects on birds exposed to oil, described above, are known to increase risk of death. Extensive bird mortality has been seen in many previous spills. Bird deaths due to the DWH spill also were extensive and obvious. Dead birds and oil-impaired live birds (i.e., those that were affected by oil to the point that they were behaving abnormally and, in many cases, could be easily captured) were seen in offshore habitats within 10 days of the initial release of oil. Additionally, dead and oil-impaired birds were found on shorelines prior to the arrival of oil onshore—presumably these birds were oiled offshore and flew or swam to land in an attempt to preen and recover. Dead birds were collected from the time that oil was being released through 2 months after the well was capped, when intensive search efforts ended. Thousands of dead birds were collected in the spill-affected area, in all of the habitats listed in Section 4.7.1.1 (Bird Diversity and Habitats in the Northern Gulf of Mexico). Numbers of collected dead and oil-impaired birds steadily increased until the well was capped, and then declined over the following two months (Figure 4.7-12).

It is widely recognized that a tally of collected bird carcasses constitutes only a fraction of true mortality (Ford et al. 2006; Henkel et al. 2012; Velando et al. 2005). As described in Section 4.7.5 (Injury Quantification), the Trustees used several methods to estimate the number of birds that were killed as a result of the DWH oil spill, as well as qualitative assessments of the magnitude of additional injury that was not captured quantitatively.



Source: *U.S. Department of the Interior.*

Figure 4.7-12. Timing and number of collections of dead and impaired birds during the DWH oil spill. Thousands of birds died, and these deaths were spread over a period of months.

Adult birds that died during the course of the oil spill were not available to lay eggs, incubate eggs, or attend to nestlings. Thus, the associated lost reproduction due to deaths of breeding birds represents additional injury caused by the DWH oil spill. This is separate from, and in addition to, quantitative and qualitative assessments of mortality. As described in Section 4.7.5 (Injury Quantification), the Trustees estimated the number of fledglings that would have been produced by adult birds that died during 2010 and 2011 as another component of the modeled oil spill injury.

4.7.5 Injury Quantification

Key Points

- The Trustees quantified mortalities representing a portion of bird injury using several approaches that account for deaths in particular habitats during specific time periods (Table 4.7-2). These quantified mortalities were estimated to range from 51,600 to 84,500 individual birds. Uncertainties associated with these methods indicate that quantified mortalities were likely underestimated and the true mortality is closer to the higher end of this range.

- Based on the portion of bird mortality that was quantified, the loss of fledglings due to mortality of their parents was estimated to be 1,700 to 6,300 in 2010 (based on adult mortalities during the 2010 breeding season, after accounting for fledgling mortality quantified using other methods) and 2,800 to 11,600 (based on adult mortalities after the 2010 breeding season). Total lost reproduction, in excess of fledgling mortality identified using other methods, was estimated to be 4,600 to 17,900. Uncertainties associated with these methods indicate that the true loss is likely higher by some unquantifiable amount.
- The Trustees recognize that additional, unquantified injury occurred in situations where quantification methods were not applied, particularly in marshes and in island waterbird colonies. Qualitative assessments were used to consider the scope of unquantified injury.
- Quantification that resulted in the mortality estimates above was conducted using several non-overlapping methods that were specific to certain habitats and time periods (Table 4.7-2). These are described below:
    - For nearshore birds (those that died within 25 miles [40 kilometers] from shore, including in open water, beaches, marsh edges, and a portion of island waterbird colonies), mortality was estimated using a Shoreline Deposition Model. The Shoreline Deposition Model used records of when and where the thousands of dead and oil-impaired birds were collected, and generated a mortality estimate that ranged from 38,900 to 58,400 for the period 20 April to 30 September 2010.
    - Some areas did not have data useable for inclusion in the Shoreline Deposition Model. Mortality in three areas (Lake Mechant, Vermilion Bay, and the Breton-Chandeleur Islands) was calculated using best estimates of densities of dead birds based on Shoreline Deposition Model results. This resulted in an additional 3,500 to 7,000 birds estimated to have died. Also, some dead birds were collected from colonies where the Shoreline Deposition Model was not applied; from these collections, 636 individuals of 22 species were added to mortality figures; much more mortality in these colonies went unquantified (see below).
    - Offshore bird mortality (greater than 25 miles [40 kilometers] from shore) during April to September 2010 was estimated using an Exposure Model, based on the spatial overlap between birds and oil. Between 2,300 and 3,100 birds were estimated to have died in this habitat.
    - Mortality that originated with exposure of nearshore birds to oil between September 2010 and March 2011 was quantified using a Live Oiled Bird Model, which combines estimates of the number of birds having external oiling and their fates; resulting mortality was estimated to range between 6,200 and 15,300.
- Considerations of unquantified mortalities centered on island waterbird colonies and marsh.
    - Although some mortality associated with island waterbird colonies was quantified, the Trustees recognize that additional mortality was undetected, due in part to restrictions on access to avoid disturbance. Given the high concentrations of birds within colonies,

many of which were known to be oiled, it is likely that additional, meaningful mortality occurred.

- Mortality quantification did not extend beyond the marsh edge. Oil was known to penetrate into marshes, which hold significant densities of specialist marsh species. Although mortality was not estimated, tens of thousands of birds were at risk of oil exposure within this habitat.

4.7.5.1 Quantified Injury: Mortality

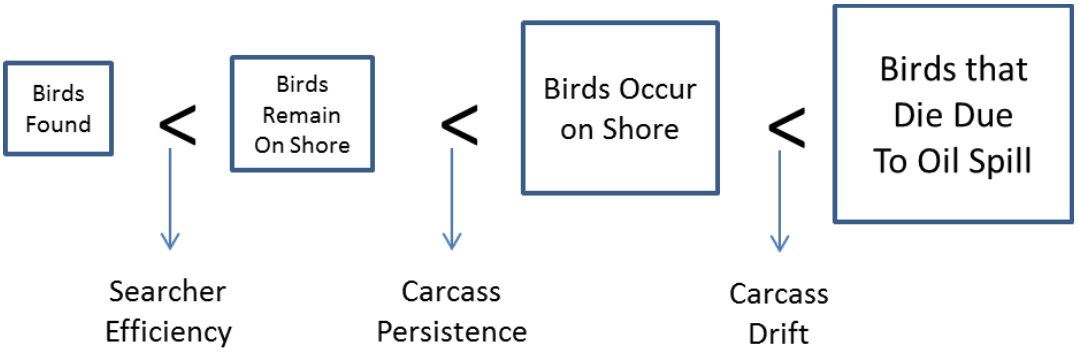
More than 8,500 individuals representing nearly 100 bird species associated with oil-affected habitats were collected dead or impaired throughout the five Gulf Coast states during wildlife rescue response and NRDA operations. More than 3,500 additional birds, across numerous species, were also observed with external oiling. In this section, estimates of mortality are presented for a portion of the bird injury, based on the observations described above. Due to a number of uncertainties within quantification methods, mortality is likely underestimated (see Section 4.7.5.5, Sources of Potential Bias and Uncertainty). In addition, some mortality occurred outside of the scope of quantification (see Section 4.7.5.4, Unquantified Injury), indicating that quantified injury constitutes only a portion of true injury.

Estimates of mortality were generated for two time periods: 1) the initial months between the beginning of the spill (April 2010) and September 2010, and 2) mortality originating from oil exposure between September 2010 and March 2011 (Table 4.7-2). During the first period, when injury was highest, mortality was estimated for a portion of the bird injury using several methods (Section 4.7.2, Approach to the Assessment) to quantify bird injury in different oil-affected habitats. During the latter period, a single method was used for a portion of the bird injuries based on observations of live nearshore birds with external oil (Section 4.7.2, Approach to the Assessment).

4.7.5.1.1 Avian Mortality Between April and September 2010

Mortality in Nearshore Areas: Shoreline Deposition Model

Because it is not practical or possible to collect all birds killed by the DWH oil spill, the Trustees estimated bird mortality in a portion of the nearshore habitats through application of a Shoreline Deposition Model for the period 20 April to 30 September 2010 (IEc 2015c). This general method has been used in previous spills, particularly along beach habitats; e.g., Ford et al. (2006). This approach uses the number of dead and impaired birds found on shorelines within the spill zone and estimates the number of birds that died, by accounting for a number of factors that influence the proportion of oil-killed birds discovered on shorelines (Figure 4.7-13).



Source: *U.S. Department of the Interior.*

Figure 4.7-13. Schematic describing several mechanisms that resulted in the number of birds collected from shorelines being less than the total number that died. The Shoreline Deposition Model uses numerical estimates of processes (carcass drift, carcass persistence, and searcher efficiency) necessary to quantitatively estimate the number of birds that died based on the number of birds found along shorelines. This method was applied to estimate a portion of nearshore bird injuries.

Factors that affected the number of birds found on shorelines relative to the number that actually died include the following: 1) some birds died at sea and sunk before they could wash up onshore or be collected in open water (corrected for by carcass drift factor); 2) some incapacitated or dead birds ended up on a shoreline but did not persist long enough to be found by searchers (corrected for by carcass persistence factor); and 3) some incapacitated or dead birds ended up onshore but were not found by searchers (corrected for by searcher efficiency factor). For the DWH oil spill, the Trustees conducted studies to quantify searcher efficiency, carcass persistence, and carcass drift so that an estimate of the total number of birds injured could be calculated for a portion of the bird injury.

Carcass Drift

When birds die during an oil spill, they do not always die on a shoreline. If a bird dies on the water, wind and water currents might push it to either a beach or marsh edge where it could be found by searchers. However, while in the water, a bird carcass also might be eaten or sink. The Trustees conducted studies to estimate carcass drift (IEc 2015c), which is an estimate of the likelihood that birds dying on water would float to a shoreline.

The Trustees placed 248 radio-tagged bird carcasses in numerous locations nearshore and offshore across the northern Gulf of Mexico (Figure 4.7-14). Placement of carcasses corresponded to the distribution of birds at risk of oil exposure. Birds were tracked until they were found on a marsh edge or beach or until the radio signals could no longer be detected within the study area. No radio-tagged bird carcasses released in offshore habitat (greater than 40 kilometers from shore) or carcasses released near the DWH well ever made it to shore. Of the 187 carcasses released in nearshore habitats and considered useable, 29 were found onshore. Therefore, the likelihood that birds dying on water (either in open sea or open water in marsh areas) drifted to shore was estimated to be 0.16, or 1 in 6.5.

Source: U.S. Department of the Interior.

Figure 4.7-14. A radio-tagged bird carcass prior to release (left) during studies to quantify carcass drift as part of a Shoreline Deposition Model, and a carcass upon discovery along shore (right).

Carcass Persistence

For a bird carcass or oil-impaired bird to be found, it must remain on a beach or along a marsh edge long enough for a searcher to find it. Bird carcasses disappear for a number of reasons, including scavenging, burial, and decomposition. Disappearance of carcasses is accounted for in the Shoreline Deposition Model by using a carcass persistence factor to estimate the likelihood that a carcass along a shoreline would disappear before searchers arrived.

The Trustees conducted studies on beaches and marsh edges to estimate carcass persistence (IEc 2015c). To determine how long the carcasses remained on the shoreline, bird carcasses were placed at known locations along sandy beach and marsh edge transects. Carcass locations were revisited on a nearly daily basis for 11 to 14 days. Bird carcasses disappeared at a faster rate at the beginning of the time period than they did later in the study for both sandy beaches and marsh edges. Bird carcasses also disappeared at a faster rate in marshes than on sandy beaches. The type of marsh was also significant. Carcasses disappeared more quickly in *Phragmites*-dominated marsh habitat compared to *Spartina*-dominated marsh habitat. These sources of variability in carcass persistence were considered when applying persistence values in the Shoreline Deposition Model.

Searcher Efficiency

Searcher efficiency is the probability that a person walking along a beach or riding in a boat near a marsh edge would see a dead or dying bird that is present within the specified search area. The Trustees conducted studies on beaches and along marsh edges to estimate the proportion of birds a searcher is likely to find (IEc 2015c). These studies were conducted by placing bird carcasses in known locations and then having trained searchers try to observe them while following procedures used during the spill to collect dead and dying birds. Not surprisingly, searcher efficiency was different for different habitat types. It was easier for searchers to find bird carcasses on a sandy beach than along a marsh edge. Also, large birds were easier to find than small birds. On sandy beaches, intact carcasses were easier to detect than partially decomposed carcasses.

Spatial and Temporal Extrapolation

Birds for the Shoreline Deposition Model were collected from 12 kilometers east of Cape San Blas, Florida, to 18 kilometers southeast of Freeport, Texas, spanning hundreds of kilometers of shoreline. However, all shorelines within this huge area could not be consistently surveyed. Areas without sufficient data coverage to estimate deposition rates needed to be accounted for by spatial extrapolation. Spatial extrapolation is when data from one geographic area are applied to another similar geographic area that does not have data. This process provides an estimate of birds that would have been collected on a shoreline segment if that segment had been searched with sufficient frequency for use in the Shoreline Deposition Model.

The Shoreline Deposition Model estimates the number of nearshore birds that died from the beginning of the spill (April 20, 2010), through the 87 days of oil spilling into the northern Gulf of Mexico (the well was capped July 15, 2010), and until area-wide wildlife operations were generally stopped (about September 30, 2010). However, not all places were searched for bird carcasses through that entire period. Just as spatial extrapolation filled in estimated deposition for places that were not searched, temporal extrapolation was used to fill in time periods that did not have adequate data coverage. Spatial and temporal extrapolations were conducted using the data that most closely corresponded to the area or time with missing data (IEc 2015c). Large areas with little or no useable data for extrapolation are treated as excluded regions.

Results of Shoreline Deposition Model

Mortality for this portion of the quantified bird injury is presented as a range of low and high estimates, reflecting estimated variability in input values (Section 4.7.5.4, Unquantified Injury). Using the Shoreline Deposition Model, the Trustees estimated that between 38,900 and 58,410 nearshore birds died between 20 April and 30 September 2010 as a result of the DWH oil spill. This represents only part of the overall modeled mortality; methods estimating additional bird mortality are presented below.

Mortality in Nearshore Regions Not Covered by the Shoreline Deposition Model

The Shoreline Deposition Model requires threshold quantities and frequencies of search data to estimate mortality. As was the case for many colonies (see below), search effort at three large areas was insufficient for modeling purposes: the Breton-Chandeleur Island Chain, Vermillion Bay, and a large portion of Lake Mechant. The observation of oil on the water and shorelines, as well as dead oiled birds collected from these locations, however, indicated that an estimation of bird mortality was required (FWS 2015b).

To approximate bird mortality within these areas, the Trustees used the number of dead birds per kilometer estimated by the Shoreline Deposition Model from similar, nearby habitats as the best approximation of dead bird density in the areas not covered by the Shoreline Deposition Model (FWS 2015b). Using this approach, the Trustees estimated that, for this modeled component of injury, between 3,500 and 7,000 bird deaths occurred in the Breton-Chandeleur Island Chain, Vermillion Bay, and Lake Mechant due to the DWH spill.

Mortality in Island Breeding Colonies

The onset of the DWH oil spill coincided with the early stages of the nesting season for numerous bird species in the northern Gulf of Mexico. Bird colonies in this area host high abundance and densities of several species and provide vital ecological function for bird populations. Several nesting colonies, including some of the largest nesting island colonies in the northern Gulf, were directly oiled and large numbers of oiled adult and sub-adult birds in and around the affected colonies were documented. Oiled adult birds also were documented in colonies that were not directly oiled.

Some colony mortality was reflected with the Shoreline Deposition Model above. However, this quantified injury is known to underestimate colony injury for reasons described below and in Section 4.7.5.4, Unquantified Injury.

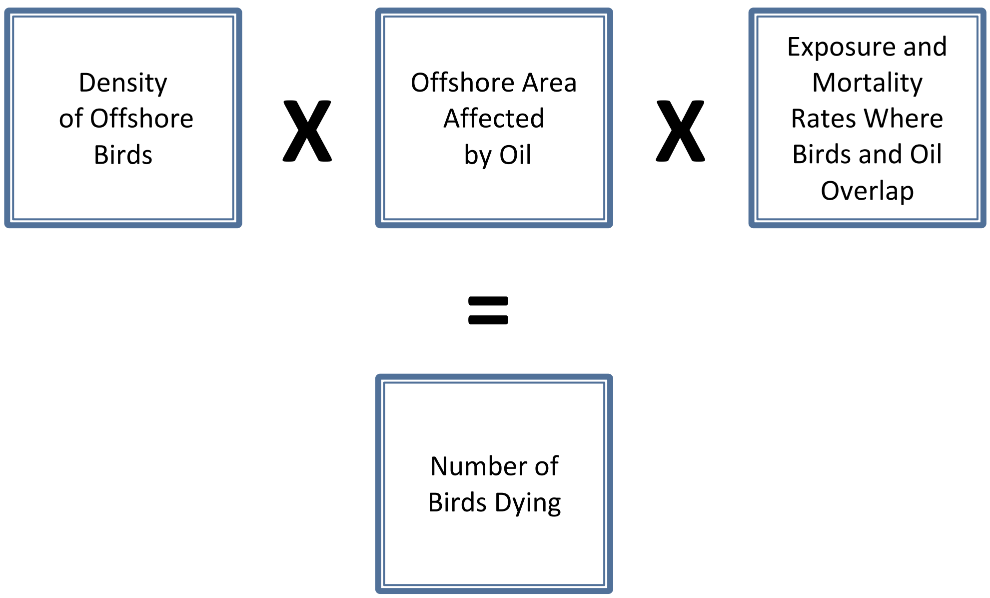
During wildlife response, colonies were handled carefully and protected to minimize disturbance and avoid additional stress being placed on the nesting birds. Consistent monitoring of bird colonies during the spill was limited by restrictions on response workers in an effort to reduce human disturbance. For this reason, search effort for some breeding colonies was not consistent with the search effort criteria required for shoreline deposition modeling. Therefore, some birds collected in colonies were not modeled in the Shoreline Deposition Model. Instead the actual dead bird count from some colonies over a limited time period was considered additional colonial bird mortality (FWS 2015a). A total of 636 dead birds of more than 20 species were collected on colonies in August and September 2010, including over 150 brown pelicans, more than 270 laughing gulls, and 125 black skimmers. The Trustees acknowledge that this is most likely a gross underestimation of mortality in these important bird habitats. Consideration of unquantified mortality within island waterbird colonies is found in Section 4.7.5.4.1.

Mortality in Offshore Open Water Areas

As described in Section 4.7.3.1 (Distribution and Duration of DWH Oil in Bird Habitats), oil occurred on surface waters of the open Gulf of Mexico for months. During that time, birds that use offshore habitats were at risk of exposure and mortality. Based on results of carcass drift studies, birds dying more than 40 kilometers from shore would not float to shore, and thus would not be represented in the Shoreline Deposition Model.

The Trustees used an Offshore Exposure Model to estimate mortality of birds in offshore areas (Figure 4.7-15) (IEc 2015a). This model first estimates the number of birds potentially affected by oil exposure by multiplying the density of birds in offshore areas (greater than 40 kilometers from shore) by the calculated offshore surface area covered by oil. Bird surveys indicated that offshore densities in water less than 200 meters deep were approximately 1.53 birds per square kilometer and in water greater than 200 meters deep were approximately 0.56 birds per square kilometer. The offshore area where oil occurred was based on the average daily coverage estimated by the Trustees for July 2010, or about 4,930 square kilometers. Estimates of exposure and mortality rates were applied to determine overall mortality. The Trustees assumed all birds occurring in the footprint of the oil slick would be exposed to oil over the 87-day course of these calculations. Because of the lack of data for categorizing degree of bird oiling in the offshore environment, the Trustees distributed the number of estimated oiled birds evenly across the four oiling categories established from all the bird types observed for this case. Ranges of mortality rates corresponding to each oiling category were applied to generate the range of overall offshore mortality (Ziccardi 2015).

Using this approach, the Trustees estimated that between 2,300 and 3,100 offshore birds were killed during the DWH oil spill. Most of the mortality occurred in shearwaters, terns, and gulls, but deaths also included members of numerous species of conservation concern to the U.S. Fish and Wildlife Service (USFWS).

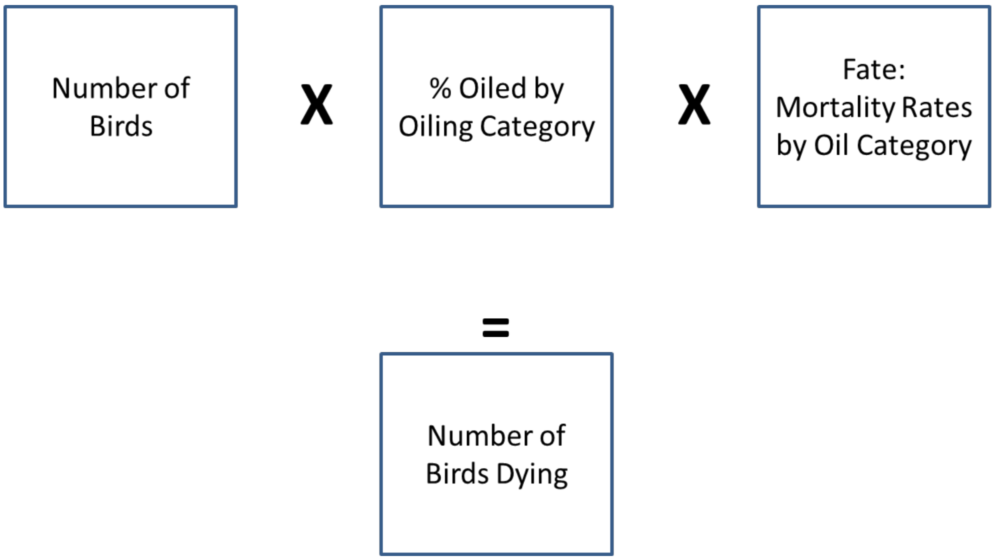


Source: *U.S. Department of the Interior.*

Figure 4.7-15. Exposure model calculation for offshore birds.

4.7.5.1.2 Avian Mortality After September 2010

Birds continued to be exposed to oil from the DWH spill past September 2010 when area-wide wildlife operations were generally stopped. To estimate a portion of the mortality that occurred as a result of oil exposure between September 2010 and March 2011, the Trustees used a Live Oiled Bird Model approach that was based on observations of live, oiled birds and estimations of mortality based on the degree of oiling (Figure 4.7-16). The difference between the Live Oiled Bird Model approach and the exposure model described above for the offshore bird mortality estimate is that these oiling rates are based on visual observations of birds, rather than an estimate of the percentage of birds that are oiled.



Source: *U.S. Department of the Interior.*

Figure 4.7-16. Calculations for a Live Oiled Bird Model used to estimate bird mortality that occurred due to oil exposure from September 2010 to March 2011.

The Live Oiled Bird Model (FWS 2015e) requires three sources of data: 1) the numbers of birds occurring in areas affected by the oil spill (abundance), 2) the incidence and degree to which birds were oiled (oiling rates), and 3) the likelihood a bird would die due to oiling (fate). Fate is further defined as the probability of a bird dying from any cause related to oil exposure, including toxic, thermoregulatory, or other effects. For this estimate, the mortality for each species is provided as a range extending from the first quartile to the third quartile of the fate estimate for each oiling rate.

The Live Oiled Bird Model calculation is a multiplication of the abundance, oiling rate, and fate information (Figure 4.7-16). This was repeated for each species and time period to estimate bird mortality originating from oil exposure during September 2010 through March 2011 (FWS 2015e). More than 2,000 individual birds were observed oiled during Live Oiled Bird Model surveys (Figure 4.7-17). Using this model, the Trustees estimated that between 6,200 and 15,300 nearshore birds were killed by the DWH oil spill during the Live Oiled Bird Model period.

Source: U.S. Department of the Interior.

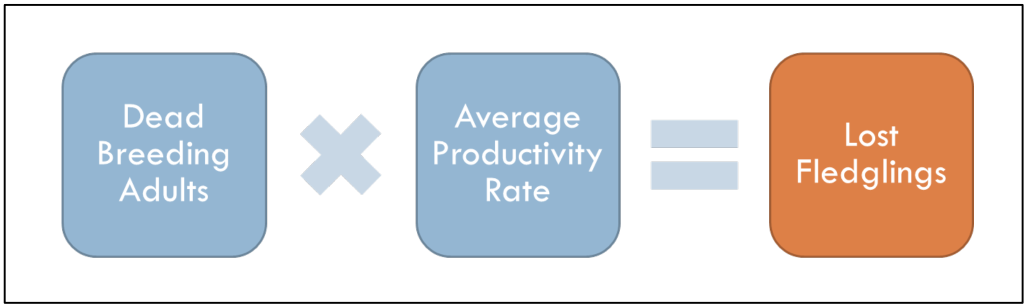
Figure 4.7-17. Examples of live birds that were visibly oiled following the DWH oil spill.

4.7.5.2 Quantified Injury: Lost Reproduction

The effects of the DWH oil spill overlapped both the 2010 and 2011 bird breeding seasons (March through August) in the northern Gulf of Mexico. The Trustees did not conduct any studies that directly quantified the impacts of the spill on reproductive success of birds nesting in the Gulf of Mexico. Logistical restrictions also prevented the Trustees from measuring reproductive success of birds that migrated through oil-contaminated areas on their way to nesting grounds outside the spill area. Nevertheless, the Trustees estimated numbers of fledglings lost in 2010 and 2011 due to the oil spill using alternative approaches that are based on the quantified mortality estimates described above (FWS 2015c, 2015d). These lost fledglings would have been the first year of progeny of breeding adults that died from the spill.

The approach to estimating the lost fledglings is shown in Figure 4.7-18. Using mortality estimates described in Section 4.7.5.1, the Trustees determined mortality for a portion of the breeding-aged birds.

Had these breeding-aged birds not died, the Trustees assumed they would have produced a number of fledglings consistent with literature on species-specific productivity (i.e., number of fledglings produced per pair). The Trustees also assumed that chicks required both parents to feed and protect them to survive to fledging. Multiplying the number of dead breeding adults by species-specific, average annual productivity values provided an estimate of number of fledglings that were not produced due to the oil spill (FWS 2015c, 2015d).



Source: *U.S. Department of the Interior.*

Figure 4.7-18. Conceptual approach to calculating lost fledglings using the average annual productivity.

The Trustees recognize that this methodology does not produce a comprehensive assessment of the loss of reproductive success in any year. There are other pathways that eggs, chicks, or fledglings could have been injured by the spill, but data were not available to quantify those injuries. Therefore, the results of these calculations are likely underestimates of lost reproduction.

4.7.5.2.1 Lost 2010 Reproduction

Estimation of lost 2010 fledglings was based on mortality ranges generated by the Shoreline Deposition Model and Excluded Regions methodologies. The Trustees estimated that between 8,500 and 12,700 dead birds were of breeding age during the 2010 breeding season, and 4,200 to 12,700 nests would have suffered a complete loss of fledglings in 2010. Using species-specific information on average annual productivity, the Trustees estimated that 4,700 to 14,200 fledglings were lost in 2010. Because a portion of these fledglings were already detected as dead birds in the Shoreline Deposition Model and excluded regions outputs, lost reproduction in 2010 is presented as "additional" lost reproduction after subtracting fledgling mortality already captured by mortality estimates in Section 5.1 (FWS 2015c, 2015d). This additional lost reproduction was estimated to be between 1,700 and 6,300 in 2010.

4.7.5.2.2 Lost 2011 Reproduction

The estimation of lost 2011 fledglings is based upon the mortality estimates generated by the Live Oiled Bird Model, the portion of the Shoreline Deposition Model output occurring after the end of the 2010 breeding season (August 8, 2010, to September 30, 2010), and dead birds from colonies. The Trustees estimated that 7,400 to 15,200 of the birds killed by the spill between August 8 to the end of the Live Oiled Bird Model period were of breeding age and that 3,000 to 12,300 nests would not have existed in the 2011 breeding season. Using species-specific information on average annual productivity, the Trustees estimate that 2,800 to 11,600 fledglings were lost in 2011.

Combining additional lost reproduction from 2010 and 2011 results in an estimate of between 4,600 and 17,900 fledglings that would have been produced in the absence of the spill, after accounting for fledglings that died in 2010 and were detected using the various mortality metrics. Limitations and uncertainties would likely contribute to an overall underestimate of fledglings lost due to the spill. Given the available information, the results presented here are the best estimate of fledglings lost due to the spill, recognizing that the true loss is likely higher by some unquantifiable amount.

4.7.5.3 Total Avian Injury Quantification

For the quantified portion of bird mortality, the Trustees estimated a spill-related injury of between 56,100 to 102,400 lost birds (Table 4.7-3). This was composed of between 51,600 and 84,500 birds that died as a direct result of the DWH oil spill (Table 4.7-3), as well as lost reproduction stemming from these mortalities that ranged between 4,600 and 17,900 fledglings. Due to a variety of factors that likely led to underestimation of mortality (Section 4.7.5.4, Unquantified Injury), the quantified portion of true injury is likely closer to the upper range of the estimates.

Ninety-three different bird species associated with oil-affected habitats showed documented injury resulting from the DWH oil spill (Table 4.7-3). There were undoubtedly other species that suffered injury that was undetected. Species showing particularly high injury included brown and white pelicans, laughing gulls, Audubon’s shearwaters, northern gannets, clapper rails, black skimmers, white ibis, double-crested cormorants, common loons, and several species of terns. The magnitude of the injury and the number of species affected makes the DWH oil spill an unprecedented human-caused injury to birds of the region.

Table 4.7-3. Estimates and ranges of bird mortality and lost productivity resulting from the DWH oil spill. Mortality estimates were generated using several methods, which are described in detail in the text. Lost productivity refers to fledglings that were not produced due to mortality of breeding-age birds during 2010 and 2011.

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
American White Pelican	Pelicans	238	358	22	43				1,836	4,052	2,096	4,453	507	2,134	2,603	6,587
Brown Pelican		7,105	10,663	642	1,277	156	12	18	2,590	6,503	10,505	18,617	2,215	8,996	12,720	27,613
Herring Gull	Gulls	161	241	15	29				36	107	212	377	10	61	222	438
Laughing Gull		19,637	29,471	1,775	3,530	273	160	264	146	503	21,991	34,041	760	2,601	22,751	36,642
Lesser Black-backed Gull		21	31	2	4				11	32	34	67			34	67
Ring-billed Gull		16	23	1	3	1			18	51	36	78	7	41	43	119
Audubon's Shearwater	Seabirds	26	39	2	5		660	887			688	931			688	931
Band-rumped Storm-petrel							16	22			16	22			16	22
Brown Booby							2	2			2	2			2	2
Cory's Shearwater		10	16	1	2		13	17			24	35			24	35
Great Shearwater		129	194	12	23		18	25			159	242			159	242
Leach's Storm-petrel		10	16	1	2						11	18			11	18

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
Magnificent Frigatebird		73	109	7	13		14	20			94	142			94	142
Manx Shearwater		5	8	0	1						5	9			5	9
Masked Booby		41	62	4	7		6	9			51	78			51	78
Northern Gannet		2,527	3,793	228	454	1	3	4	26	77	2,785	4,329	44	160	2,829	4,489
Parasitic Jaeger							4	6			4	6			4	6
Red-billed Tropicbird							7	9			7	9			7	9
Sooty Shearwater		5	8	0	1						5	9			5	9
White-tailed Tropicbird		5	8	0	1						5	9			5	9
Wilson's Storm-petrel							14	19			14	19			14	19
American Coot	Rail	16	23	1	3	1					18	27			18	27
Clapper Rail		347	521	31	62	2					380	585	324	1,054	704	1,639
Common Gallinule		16	23	1	3						17	26			17	26
Purple Gallinule		16	23	1	3						17	26			17	26
Sora		16	23	1	3						17	26			17	26
Virginia Rail		5	8	0	1						5	9			5	9
Osprey	Raptor	73	109	7	13						80	122			80	122

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
American Avocet	Shore								1	1	1	1			1	1
American Oystercatcher		73	109	7	13	1			1	2	82	125			82	125
Black-bellied Plover		5	8	0	1				10	21	15	30			15	30
Black-necked Stilt		16	23	1	3	1			5	14	23	41			23	41
Dunlin		10	16	1	2				52	138	63	156	29	155	92	311
Killdeer		21	31	2	4						23	35			23	35
Least Sandpiper		5	8	0	1						5	9			5	9
Long-Billed Dowitcher		5	8	0	1				1	1	6	10			6	10
Piping Plover[e]		21	31	2	4				3	6	26	41			26	41
Ruddy Turnstone		83	124	7	15				2	3	92	142			92	142
Sanderling		155	233	14	28				14	36	183	297	6	32	189	329
Semipalmated Plover									7	16	7	16			7	16
Semipalmated Sandpiper		21	31	2	4						23	35			23	35
Short-billed Dowitcher		16	23	1	3				3	6	20	32			20	32
Snowy Plover									0	1	0	1			0	1

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
Spotted Sandpiper		5	8	0	1						5	9			5	9
Western Sandpiper									2	5	2	5			2	5
Willet		62	93	6	11	1			8	17	77	122			77	122
Wilson's Plover		16	23	1	3				0	2	17	28			17	28
Black Skimmer	Tern	1,015	1,523	92	182	125			336	734	1,568	2564	23	99	1,591	2,663
Black Tern		47	70	4	8		745	988			796	1,066			796	1,066
Bridled Tern		10	16	1	2		46	62			57	80			57	80
Brown Noddy		5	8	0	1		15	21			20	30			20	30
Caspian Tern		119	179	11	21				62	187	192	387	29	178	221	565
Common Tern		166	249	15	30		9	12	58	127	248	418	29	128	277	546
Forster's Tern		269	404	24	48	5	3	4	16	42	317	503	14	54	331	557
Gull-billed Tern		26	39	2	5						28	44			28	44
Least Tern		642	964	58	115	7					707	1,086	40	128	747	1,214
Royal Tern		2,061	3,093	186	370	25	306	406	250	708	2,828	4,602	112	456	2,940	5,058
Sandwich Tern		513	769	46	92	9	94	124	20	44	682	1,038	11	54	693	1,092
Sooty Tern		16	23	1	3		167	221			184	247			184	247
Black-crowned Night-heron	Waders	114	171	10	20				10	19	134	210			134	210

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
Cattle Egret		264	396	24	47	1					289	444	152	459	441	903
Great Blue Heron		331	497	30	60	1			4	11	366	569	14	76	380	645
Glossy Ibis		5	8	0	1						5	9			5	9
Great Egret		109	163	10	20	1			46	137	166	321	9	51	175	372
Green Heron		93	140	8	17						101	157			101	157
Least Bittern		26	39	2	5						28	44			28	44
Little Blue Heron		26	39	2	5	3					31	47			31	47
Reddish Egret		21	31	2	4				2	4	25	39			25	39
Roseate Spoonbill		218	326	20	39	3			6	10	247	378	21	80	268	458
Snowy Egret		145	218	13	26	7			6	11	171	262			171	262
Tricolored Heron		249	373	22	45	11					282	429			282	429
White Ibis		36	54	3	7	1			508	1,421	548	1,483	73	408	621	1,891
Yellow-crowned Night-Heron		52	78	5	9						57	87			57	87
Black-Bellied Whistling-Duck	Waterfowl	10	16	1	2						11	18			11	18
Blue-winged Teal		83	124	7	15						90	139			90	139

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
Bufflehead		10	16	1	2						11	18			11	18
Canada Goose		10	16	1	2						11	18			11	18
Fulvous Whistling-Duck		5	8	0	1						5	9			5	9
Green-winged Teal		10	16	1	2						11	18			11	18
Lesser Scaup		5	8	0	1						5	9			5	9
Mallard		161	241	15	29						176	270			176	270
Mottled Duck		47	70	4	8						51	78			51	78
Red-Breasted Merganser		10	16	1	2						11	18			11	18
Ruddy Duck		10	16	1	2						11	18			11	18
Surf Scoter		16	23	1	3						17	26			17	26
Double-crested Cormorant	Cormorant	274	412	25	49				62	185	361	646	104	413	465	1,059
Neotropic Cormorant		10	16	1	2						11	18			11	18
Belted Kingfisher		5	8	0	1						5	9			5	9
Boat-tailed Grackle	Land	5	8	0	1						5	9			5	9
Red-winged Blackbird		5	8	0	1						5	9			5	9

Species[a]	Guild	Shoreline Deposition Model[b]		Excluded Regions[b]		Colony Sweeps[b]	Offshore[b]		Live Oiled Bird Model[c]		Subtotal		Additional Lost Productivity[d]		Combined Totals	
Seaside Sparrow		5	8	0	1						5	9			5	9
Common Loon	Loons and Grebes	445	668	40	80				45	64	530	812	31	98	561	910
Pied-billed Grebe		202	303	18	36						220	339			220	339
TOTALS[f]		**38,918**	**58,410**	**3,506**	**6,999**	**636**	**2,314**	**3,140**	**6,203**	**15,298**	**51,577**	**84,483**	**4,564**	**17,916**	**56,141**	**102,399**

[a] Species listed in bold are listed as federally threatened or endangered, or included as USFWS species of conservation concern (FWS 2008).

[b] Shoreline Deposition Model, Excluded Regions, Colony Sweeps, and Offshore are estimates of adult mortality from April to September 2010.

[c] Live Oiled Bird Model estimates adult mortality from October 2010 to April 2011.

[d] Lost productivity refers to fledglings that were not produced due to mortality of breeding-age birds during 2010 and 2011.

[e] The piping plover is listed as both threatened and endangered (depending on population) under the Endangered Species Act of 1973 (87 Stat. 884, as amended; 16 U.S.C. 1531 et seq.).

[f] Totals may differ somewhat from values presented in the technical reports due to a rounding error because some models (Shoreline Deposition Model, Excluded Regions, Offshore, Lost Productivity) produce a total number of birds that was distributed to the relative bird frequencies in the database; this resulted in fractions being rounded to the nearest integer for presentation.

4.7.5.4 Unquantified Injury

As a result of the immense area affected by the spill, the diversity of habitats involved, and the prolonged nature of the event, there were a number of bird injuries that were not detected or estimated using quantified portions of the Trustees' assessment approach. However, these are important to consider to more completely understand the full scope of bird injuries, as well as the habitats in which these injuries occurred.

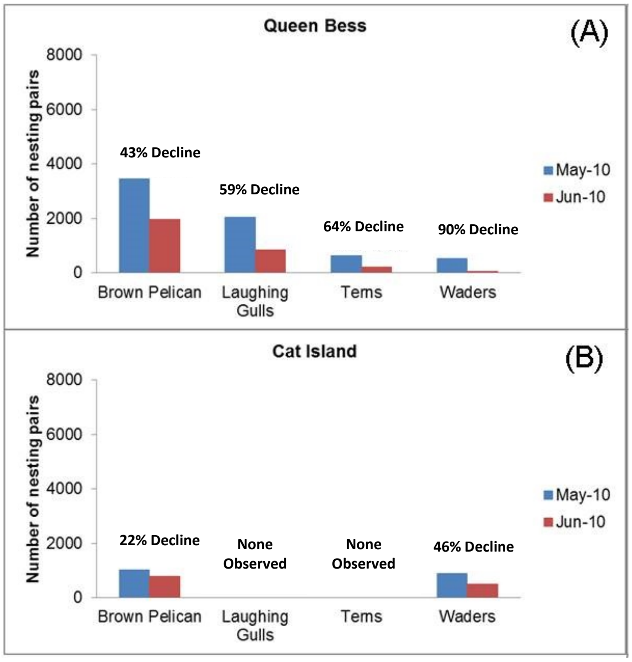
4.7.5.4.1 Island Waterbird Colonies

Some mortality in island waterbird colonies was included in Injury Quantification (see Section 4.7.5.1.1). However, the Trustees recognize that these estimates do not fully capture the total injury that occurred within colonies. There were many time periods and colonies for which neither Shoreline Deposition Model nor colony sweep methods could be used to quantify colony injury, largely due to lack of search effort with the intent of minimizing disturbance of nesting birds. Given the large aggregations of birds that were in colonies at the time of the spill and the occurrence of DWH oil contamination at some of these colonies (Figure 4.7-19), lack of completely quantified mortality likely resulted in substantial underestimation of bird injury.

To illustrate potential effects of the DWH oil spill on island waterbird colonies, the Trustees evaluated changes in bird abundance at several waterbird colonies during 2010, the year of the spill (Baker et al. 2015). This analysis indicated that reductions in representative colonial breeding bird abundance occurred coincident with oil exposure. For example, Baker et al. (2015) demonstrated that abundance of brown pelicans, laughing gulls, terns, and wading birds declined by approximately 50 percent at two colonies in Barataria Bay from May to June 2010 (Figure 4.7-20). This is a period when colony abundance would be expected to increase, under normal conditions. It is unknown whether these observations represent mortality, movement, or both. If the declines included mortality, some of that was likely captured in quantitative methods. However, these highlight the large-scale disruption to birds nesting at affected colonies during the DWH event.

Source: Louisiana Department of Wildlife and Fisheries.

Figure 4.7-19. Breeding adults and their chicks were exposed to DWH oil on island waterbird colonies, as shown in the photo above of oiled brown pelican chicks at Cat Island colony. Brown pelican chicks should be covered in white down feathers at this stage of development.



Source: Louisiana Department of Wildlife and Fisheries.

Figure 4.7-20. Change in abundance of select bird species at two oil-affected colonies during 2010, the year of the DWH oil spill.

4.7.5.4.2 Marsh

Most oil-affected habitats were considered as part of quantitative avian injury assessment. However, birds occurring in marsh habitats suffered injuries that were not quantified. The Shoreline Deposition Model captured mortality that occurred at the marsh edge. It is known that oil penetrated deeper into the marsh (Figure 4.7-21), likely causing bird injury to marsh species that went undetected. In addition, injury that occurred to marsh and nearshore birds would be undetected for individuals that moved deeper into the marsh before dying, a likely response to compromised health for many species.

Source: *Louisiana Department of Wildlife and Fisheries.*

Figure 4.7-21. DWH oil contaminated marsh edge and also penetrated into interior marsh habitats. This photo shows oil (dark brown) penetrating beyond the marsh edge (left side of photo) into the marsh interior.

Coastal marshes support an abundance of numerous bird species, such as clapper rails, seaside sparrows, mottled ducks, least bitterns, green herons, common gallinules, willets, pied-billed grebes, marsh wrens, orchard orioles, common yellowthroats, boat-tailed grackles, and red-winged blackbirds. Birds that live strictly in marsh habitats, as well as birds that extensively use coastal marsh habitat, were likely exposed to oil, died within marsh habitats, and were never collected to be quantified by the Shoreline Deposition Model. The Trustees therefore applied methods to estimate the magnitude of numbers of birds potentially exposed to oil, as a qualitative means to evaluate potential injury (Wallace & Ritter 2015).

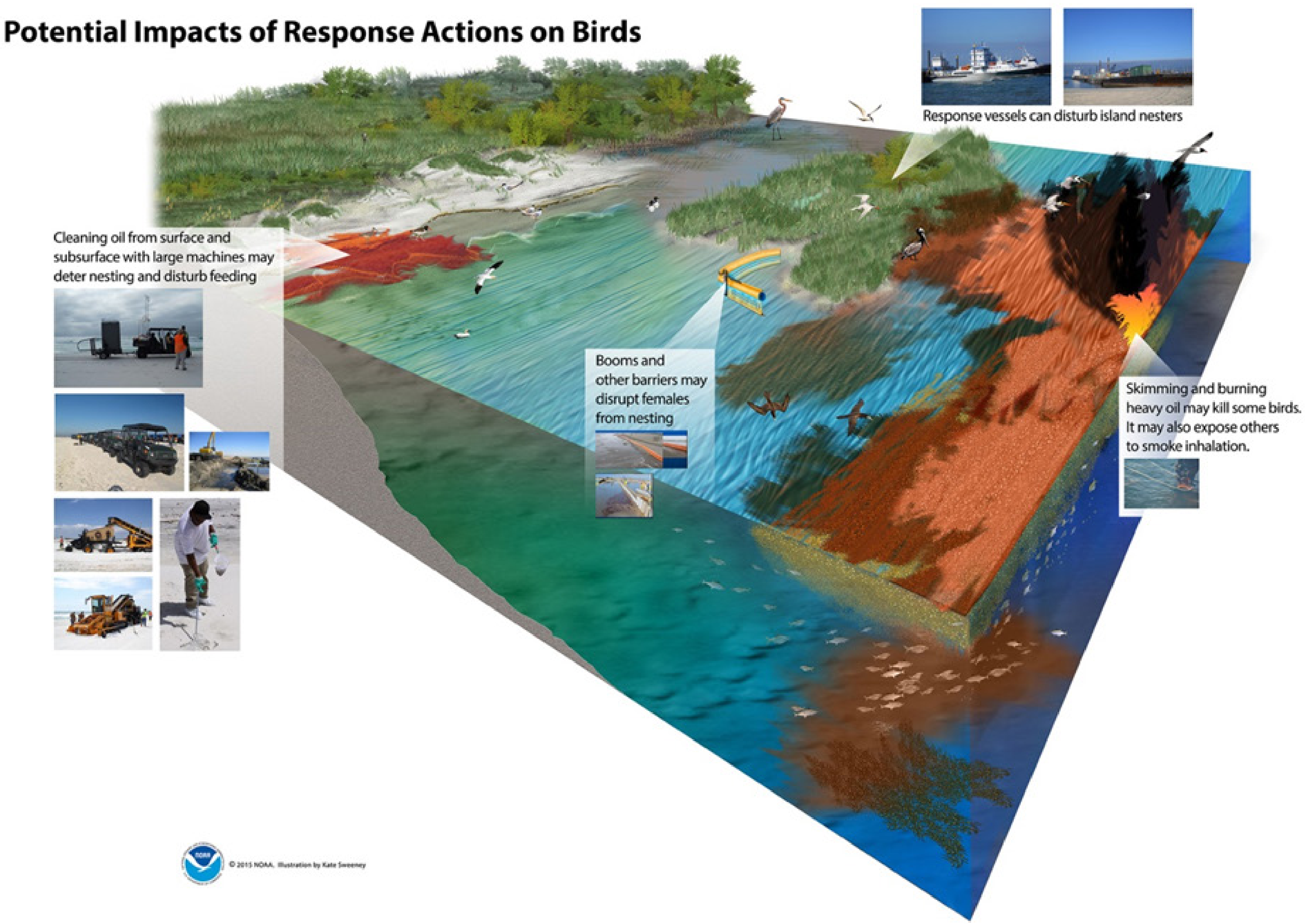
The Trustees collected data on densities of seaside sparrows, clapper rails, least bitterns, and red-winged blackbirds in different marsh habitats (e.g., *Spartina*-dominated and *Phragmites*-dominated); only density values derived from distance sampling models were used (Conroy 2013). Bird densities ranged between 0.05 individuals per hectare (least bitterns in *Spartina* marsh) to 3.4 individuals per hectare (seaside sparrows in *Spartina* marsh). Habitat-specific densities for the four example species were multiplied by the oiled shoreline lengths for each habitat type (Section 4.6, Nearshore Marine Ecosystem), using an assumption that any bird within 100 meters of an oiled shoreline was potentially susceptible to oil exposure.

Based on these methods described above, tens of thousands of individuals from these four bird species were estimated to have been present and therefore potentially exposed to DWH oil in oiled marsh habitats in Louisiana (Wallace & Ritter 2015).

The actual exposure and injury are not quantified, but this exercise indicates that substantial injury to marsh birds likely occurred. Heavily oiled marsh areas had extensive oiling on the soil, oil coating the vegetation, and oil-contaminated prey; birds that were present in these habitats would have been exposed via multiple pathways. For example, birds would have come into direct contact with oiled vegetation through walking, perching, foraging, hiding from predators, etc., and would have ingested oil when preening oil from feathers, eating contaminated prey, and ingesting soil or sediment while feeding.

4.7.5.4.3 Response Activities

Actions and activities of people and equipment deployed to control and clean up the oil spill also may have directly and indirectly injured birds, although these were not quantified as part of the avian assessment. Direct response injuries include disturbance of birds while nesting or foraging, crushing of nests or young, and intentional hazing (using propane cannons and other methods) to deter birds from heavily oiled areas. Examples of indirect effects include reduction of food sources, nest abandonment, and loss of habitat. The array of potential response effects is illustrated in Figure 4.7-22. Response activities also may have interfered with collection of carcasses that would have been used in injury assessment. For instance, machines that skimmed oil-contaminated material from sandy beaches or open water may have incidentally collected bird carcasses, and in situ burning of oil may have destroyed bird carcasses.



Source: *Kate Sweeney for NOAA. Photos from U.S. Department of the Interior.*

Figure 4.7-22. Potential effects of response actions on birds. Text boxes highlight specific details about various response actions and potential adverse effects to birds.

Several response activities took place with the intent of preventing oil from getting to shore and cleaning oiled shoreline. These activities likely affected colonial nesting birds. For example, in many colonies, boom moved into the colonies, likely impacting nests, nesting habitat, and chicks (Figure 4.7-23; (Baker et al. 2015).

Source: *Louisiana Department of Wildlife and Fisheries.*

Figure 4.7-23. Containment and sorbent boom washed ashore in a brown pelican colony. Boom is wrapped around pelican nests and fledglings on nests in the colony, which likely resulted in injuries to nesting birds, their nests, and their fledglings.

Response activities not only had physical impacts to the colonies, but also likely enhanced oil exposure to colonies and colonial birds. For example, oiled booms were found retaining oil on the water against colonies for several days at a time (Figure 4.7-24). The daily persistence of oil, in combination with tidal fluctuations, likely enhanced oil exposure on shorelines, vegetation, and sediment at colonies. Additionally, lost nesting habitat destroyed by response activities, such as boom removal, could affect nesting success in future years (Baker et al. 2015).

Source: *Louisiana Department of Wildlife and Fisheries.*

Figure 4.7-24. Oil trapped by boom against Cat (or Mangrove) Island colony in Barataria Bay.

4.7.5.4.4 Injury Outside the Domain of Quantified Injury

As demonstrated in the *Exxon Valdez* oil spill, chronic effects to birds and degradation of habitat quality may persist for extended periods (Iverson & Esler 2010). This assessment considered only effects within the first year post-spill, so any subsequent effects were not estimated, likely leading to an underestimation of overall bird injury. Similarly, longer-term health effects to birds exposed to DWH oil during the Trustees' period of study may not manifest as shortened lifespan until years later; that injury would not be captured by this assessment. Also, migratory birds with compromised health or damage to feathers as a direct result of the DWH oil spill could suffer depressed reproduction or increased risk of death (Ziccardi 2015) in times outside our period of study and/or in places outside of the northern Gulf of Mexico. These injuries would not be detected using our assessment metrics. Finally, the DWH oil spill had significant effects on the entire northern Gulf of Mexico ecosystem (Chapter 3, Ecosystem Setting and Affected Environment). The indirect effects that radiate from a perturbation of that magnitude likely had significant, negative effects on birds. For example, disruptions to food webs, loss of nesting structure, and persistent contamination are all likely mechanisms of injury that would not be detected in this assessment.

4.7.5.5 Sources of Potential Bias and Uncertainty

Within the quantitative methods used by the Trustees to estimate mortality, there are a number of potential sources of bias that likely led to underestimates of injury, including the following:

- Searcher efficiency trials conducted as part of the Shoreline Deposition Model were conducted under ideal conditions, with respect to tides, carcass condition, observer motivation, and other factors. With an oil spill of this geographic and temporal magnitude, the conditions during actual carcass collection were likely less than ideal in many instances, resulting in poorer searcher efficiency and mortality underestimation.
- Carcass persistence along marsh edges may have been overestimated, as the carcasses used in the Trustees' carcass persistence study were fresher and more intact than the average bird found during spill response efforts.
- Based on results of field and lab studies, the Trustees hypothesize that estimates of oiling rates conducted as part of the Live Oiled Bird Model may have been low for birds with dark plumage (e.g., cormorants) due to difficulties in detecting oil on these birds.
- Similarly, some birds that were determined to be unoiled may have been oiled previously but had preened away that oil prior to observation. These birds would still be subject to the detrimental health effects the Trustees documented in our laboratory exposures following external exposure to oil.
- Throughout the multitude of bird surveys conducted during the spill, oiling observations were limited to locations on a bird's body that were visible to observers. Most birds observed were resting, nesting, or loafing—positions that would have limited observers' ability to evaluate oiling on bird breasts and bellies, the body parts where they would be most likely to come in contact with oil on the surface of the water or land.

The Trustees made a conscientious attempt to estimate avian injury as accurately as possible, but recognize that variable and uncontrolled field conditions and factors that were impossible to account for resulted in uncertainty and variability in our estimates. Ranges of mortality are presented for the quantified components of injury, rather than point estimates, to attempt to account for some of that uncertainty and variability. The Trustees consider their estimate of injury to be scientifically reasonable but conservative (i.e., injuries were underestimated). Inherent bias and uncertainties like those described above resulted in underestimation of bird mortality caused by the DWH oil spill. Thus it is most likely that true mortality for the quantified components of the injury assessment was closer to the upper estimates presented in Table 4.7-3 than the lower estimates.

4.7.6 Conclusions and Key Aspects of the Injury for Restoration Planning

4.7.6.1 Summary

The Trustees have documented a large-scale and pervasive bird injury in the northern Gulf of Mexico as a result of the DWH oil spill. Bird deaths for the quantified component of injury assessment were found to be in the tens of thousands. Injured birds represented nearly every coastal bird guild and the habitats that these birds rely upon, including islands supporting nesting colonies, beaches, marshes, open water, and *Sargassum* rafts.

Specifically, the injury assessment showed that:

- At least 93 resident and migratory species of birds across all five Gulf Coast states were exposed to DWH oil in multiple northern Gulf of Mexico habitats, including open water, barrier islands, beaches, bays, and marshes. Laboratory studies showed that exposure to DWH oil led to injuries, including feather damage, abnormal blood attributes, organ damage, and death.
- The total quantified injury was estimated to range from 56,100 to 102,400 lost birds, which included between 51,600 and 84,500 birds that died as a result of the DWH oil spill. Further, of those dead birds, breeding age adults would have produced an estimated additional 4,600 to 17,900 fledglings in 2010 and 2011. Due to a number of factors that likely led to underestimation of mortality, true injury is likely closer to the upper ranges than the lower. In addition, unquantified injury in marsh and island waterbird colonies suggests that the total injury is substantially higher.
- The magnitude of the injury and the number of species affected makes the DWH spill an unprecedented human-caused injury to birds of the region.

The Trustees considered all of these aspects of the injury in restoration planning, in addition to the ecosystem effects and recovery described below.

4.7.6.2 Ecosystem Effects

Birds are important components of marine ecosystems across the globe. They are highly responsive to variation in prey, and also exert top-down effects on the number and distribution of prey species. They also are abundant with high metabolic rates, and thus exhibit high food consumption relative to other taxa, which increases their influence on marine communities. Birds also serve as prey for other species (e.g., raptors, alligators), and changes in the prey base could have effects on top level predators. The Trustees, therefore, expect that the loss of birds as a result of the DWH oil spill would have meaningful effects on food webs of the northern Gulf of Mexico.

Birds serve additional ecological functions, including transfer of nutrients between marine and terrestrial biomes, seed dispersal, and many others. These functions were disrupted as a result of the DWH oil spill due to losses of birds and changes to their behavior.

Ecosystem ramifications resulting from bird injury following the DWH spill are not limited to the northern Gulf of Mexico. Many birds that occur in the spill-affected region migrate to areas across North, Central, and South America (Figure 4.7-3) where their impaired performance or reduction in numbers could have radiating effects on ecosystems similar to those described above. These broad effects, although difficult to quantify, may have occurred due to changes in many bird species across many migration and breeding habitats.

4.7.6.3 Recovery

The DWH oil spill injured avian resources throughout the Gulf of Mexico through a variety of mechanisms, including but not necessarily limited to exposure to oil, disturbance from response activities, and degradation of habitat. The Trustees estimate that between 56,100 and 102,400 birds were killed by the spill or not produced as a result of the spill, representing dozens of species across all five Gulf Coast states (Table 4.7-3).

Ultimately, the large number of individuals injured by the spill, the diversity of species, specific life history requirements and population structures of different species, spill-related impacts to habitats and prey relied upon by birds, and other factors unrelated to the spill that can affect bird species make the estimation of recovery time challenging. The Trustees documented mortality and reproductive loss over approximately 1 year following the spill. Injury may have persisted beyond that period for some species. Complete recovery of the tens of thousands of birds lost due to the DWH oil spill would take longer if relying on natural recovery (no action) than if restoration actions were implemented. To restore the number of birds lost during the spill will require many years of restoration activities.

4.7.6.4 Restoration Considerations

As described in Chapter 5 (Restoring Natural Resources), the Trustees have identified an integrated portfolio of restoration approaches to restore for these avian injuries. The Trustees will consider restoration actions across the Gulf of Mexico, as well as in non-Gulf areas where injured bird species migrate and/or breed, potentially including but not necessarily limited to the upper Midwest, northwest Atlantic, and Caribbean.

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4.8 Sea Turtles

What Is in This Section?

- **Executive Summary**
- **Introduction and Importance of the Resource (Section 4.8.1):** Why do we care about sea turtles and their habitats?
- **Approach to the Assessment (Section 4.8.2):** How did the Trustees assess injury to sea turtles?
- **Exposure (Section 4.8.3):** How, and to what extent, were sea turtles and their habitats exposed to *Deepwater Horizon* (DWH) oil and response activities?
- **Injury Determination (Section 4.8.4):** How did exposure to DWH oil and response activities affect sea turtles?
- **Injury Quantification (Section 4.8.5):** What was the magnitude of injury to sea turtles?
- **Conclusions and Key Aspects of the Injury for Restoration Planning (Section 4.8.6):** What are the Trustees' conclusions about injury to sea turtles, ecosystem effects, and restoration considerations?
- **References (Section 4.8.7)**

Executive Summary

Sea turtles are irreplaceable natural resources in the Gulf of Mexico because they serve unique ecological roles and are highly valuable to the public. All five sea turtle species that occur in the Gulf are listed as threatened or endangered under the U.S. Endangered Species Act (ESA). The Gulf provides critically important habitats for sea turtle reproduction, feeding, migration, and refuge, including extensive *Sargassum* habitat that small, oceanic juvenile turtles depend on for survival.

Because the DWH spill footprint overlapped in time and space with sea turtles throughout the northern Gulf of Mexico, all five sea turtle species and their habitats were exposed to DWH oil in the open ocean, across the continental shelf, and into nearshore and coastal areas, including beaches. Sea turtles were exposed to oil when in contaminated water or habitats; breathing oil droplets, oil vapors, and smoke; and ingesting oil-contaminated water and prey. Response activities and shoreline oiling also directly injured sea turtles and disrupted or deterred sea turtle nesting in the Gulf. The pervasive and prolonged nature of the DWH spill and related response activities meant that sea turtle exposures to DWH oil and resulting injuries were inescapable for many turtles.

The Trustees performed several activities to assess oil exposure and injury to sea turtles from the DWH oil spill in the various geographic areas that sea turtles occupy, including surface habitats in the open ocean, across the continental shelf, and on nesting beaches. Activities included boat-based rescues and veterinary assessments, aerial surveys, satellite tracking of live sea turtles, recovery of stranded sea turtles, and monitoring of nesting sea turtles and their nests. Approximately 1,800 sea turtles, across all life stages, were directly observed within the cumulative DWH oil footprint. Nearly 600 of these were directly assessed for degree of oil exposure, and more than 300 were rehabilitated and eventually released. Oil collected from rescued and stranded turtles, and in tissues of dead turtles, was confirmed to be DWH oil. In addition, the Trustees assessed sea turtle injuries caused by response activities, such as increased boat traffic, dredging for berm construction, increased lighting at night near nesting beaches, and oil cleanup operations on nesting beaches.

Based on these studies, the Trustees concluded that sea turtles in offshore areas, continental shelf areas, and on nesting beaches suffered adverse effects from DWH oil exposure and response activities throughout the northern Gulf of Mexico. External and internal oiling were clearly documented and demonstrated that sea turtles were exposed to DWH oil by multiple pathways. Among all exposure pathways evaluated, veterinarians and sea turtle biologists concluded that the most acute physical and physiological adverse effects resulted from direct contact with surface oil, which mired and killed turtles. All turtles must spend time at the surface to breathe, rest, bask, and feed, and these fundamental behaviors put turtles at continuous and repeated risk of exposure anywhere the ocean surface was contaminated by DWH oil. Toxicity resulting from inhalation and ingestion of oil also may have significantly contributed to injury; however, these effects are not as well understood.

Not only are sea turtles required to come to the surface to breathe, but they also must come ashore to lay their eggs. The latter requirement puts nesting sea turtles and their eggs and hatchlings at risk of exposure to oil and effects of cleanup response activities on nesting beaches. Indeed, DWH response actions undertaken to remove oil from the beaches and the ocean and to prevent hatchlings from entering the Gulf during the oil spill resulted in direct injuries to turtles, including decreased nesting and loss of sea turtle hatchlings, in all areas of the northern Gulf of Mexico.

Direct observations of the effects of oil on turtles obtained by at-sea captures, sightings, and strandings (dead or debilitated turtles that wash ashore or are found floating close to shore) were only partial samples that did not represent the full scope of the injury. The vast expanse of the search area and distance from shore significantly limited the proportion of the spill area that could be physically searched for sea turtles. For example, the area surveyed during rescue operations was less than 10 percent of the total footprint of more than 38,000 square miles (100,000 square kilometers), and search efforts did not occur during the entire spill period. Further, due to safety considerations associated with ongoing response actions, these search efforts were not centered close to the wellhead where effects were likely the greatest. Inherent challenges to studying highly mobile marine animals (i.e., they are typically located in remote areas that are difficult for researchers to access and are difficult to find and capture at sea, and certain life stages spend most of their time below the surface) further restricted the Trustees' survey efforts. For these reasons, the Trustees used expert opinion, surface oiling maps, and statistical approaches to apply the directly observed adverse effects of oil exposure to turtles in areas and at times that could not be surveyed. This produced estimates of the total number of sea turtles that were injured within the entire footprint and period of the DWH oil spill.

The Trustees estimated that between 4,900 and up to 7,600 large juvenile and adult sea turtles (Kemp's ridleys, loggerheads, and hardshelled sea turtles not identified to species), and between 55,000 and 160,000 small juvenile sea turtles (Kemp's ridleys, green turtles, loggerheads, hawksbills, and hardshelled sea turtles not identified to species) were killed by the DWH oil spill. Nearly 35,000 hatchling sea turtles (loggerheads, Kemp's ridleys, and green turtles) were also injured by response activities.

Despite uncertainties and some unquantified injuries to sea turtles, the Trustees conclude that this assessment adequately quantifies the nature and magnitude of injuries to sea turtles caused by the DWH oil spill and related activities. Restoration approaches should address different life stages and geographic areas to ensure that sea turtles will be able to fulfill their unique ecological role in the Gulf of Mexico ecosystem in the future.

4.8.1 Introduction and Importance of the Resource

Key Points

- Five species of sea turtles inhabit the Gulf of Mexico: loggerhead (*Caretta caretta*), Kemp's ridley (*Lepidochelys kempii*), green turtle (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*), and leatherback (*Dermochelys coriacea*). All sea turtle species in the Gulf of Mexico are listed under the ESA.
- Sea turtles occupy unique ecological roles as long-lived, large-bodied animals that rely on both marine and terrestrial ecosystems to support their life history.
- Sea turtles require long-term, consistent, effective protection to prevent further population declines and possible extinction.
- Given the extensive nature of the DWH oil spill, it is key to understand how different life stages are distributed, and how different species of sea turtles use habitats in these different areas, in order to assess impacts of the DWH oil spill. Consequently, the Trustees assessed injury to sea turtles by species and life stage.

4.8.1.1 Resource Description

4.8.1.1.1 What Are Sea Turtles?

Sea turtles' unique biology and life history, as well as their status as easily recognizable icons of the oceans, make them flagship species for the health of marine ecosystems and for marine conservation efforts globally (Frazier 2005). Like salmon and migratory birds, turtles have evolved extremely accurate homing and navigational systems that allow them to migrate between distinct feeding areas and breeding areas (Lohmann et al. 1997), including returning to nest on the beaches where they were born (i.e., natal beaches) (Lohmann et al. 1997). Sea turtles return to their natal beaches as breeding adults decades, rather than years, after imprinting on those areas as tiny hatchlings.

Despite their marine nature, sea turtles remain inexorably tied to sand beaches for reproduction. Female sea turtles return to dry land to dig their nests and lay their eggs, just like freshwater turtles and tortoises. Eggs remain buried below 0.3 to 1 meter of sand for the duration of incubation, and embryos obtain all the oxygen, water, and heat from the surrounding sand that they need to develop into hatchlings. After 45 to 60 days of incubation, hatchlings emerge from their nests, quickly crawl to the surf, and begin a marathon swim to reach offshore areas, find refuge, and begin their lives as truly *marine* turtles. After decades in offshore and then continental shelf areas, sea turtles reach adulthood and begin to reproduce, beginning again a life cycle that has been occurring unchanged for hundreds of millions of years (see Section 4.8.1.1.4, Sea Turtle Life Stages and Habitat Use).

4.8.1.1.2 Ecological Roles of Sea Turtles and Their Values to the Public

Sea turtles occupy unique ecological roles as long-lived, large-bodied animals that move through several habitats during their lives. For example, different sea turtle species show unique dietary specialization: hawksbills eat sponges, leatherbacks eat jellyfish, and green turtles eat mainly seagrass and algae. Loggerheads and Kemp's ridleys are carnivores, but their diets vary regionally depending on available prey species (Bjorndal 1997).

Because sea turtles rely on both marine and terrestrial habitats to support their life history, they connect ocean to land in ways that few species do. Sea turtles transport marine nutrients to terrestrial environments through the eggs that they lay in sandy beach habitats (Bouchard & Bjorndal 2000), and also serve as important prey resources for many predators and scavengers (Heithaus 2013). Until sea turtles reach large body sizes, they are vulnerable to predation by predators that depend on them for food (Bolten 2003). Therefore, reductions in the number of small sea turtles mean a loss of valuable resources for marine predators, as well as scavengers and decomposers (Heithaus 2013).

In addition to serving important ecological roles, sea turtles are also extremely valuable natural resources to humans as subjects of wildlife-viewing activities, whether through formal ecotourism or informal enjoyment of nature. In nearly every country in the world where sea turtles are present, particularly where they nest, people make efforts to observe sea turtles in the wild. This is especially true in the United States, including in Gulf states.

4.8.1.1.3 Status of and Threats to Sea Turtles in the Gulf of Mexico

At present, there are seven species of sea turtles worldwide, most of which have global distributions, with nesting beaches restricted to the tropics and subtropics, and marine ranges extending into high latitudes and cold water (typically 10–15°C/50–60°F), as in the case of the leatherback turtle (Wallace et al. 2010). Five species of sea turtles inhabit the Gulf of Mexico: Kemp's ridley (*Lepidochelys kempii*), loggerhead (*Caretta caretta*), green turtle (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*), and leatherback (*Dermochelys coriacea*) (Figure 4.8-1). Kemp's ridleys, loggerheads, green turtles, and hawksbills are in the Cheloniidae family (i.e., hard shells), and

Source: *Dawn Witherington.*

Figure 4.8-1. Five species of sea turtles inhabit the Gulf of Mexico: (clockwise from top left) Kemp's ridley (*Lepidochelys kempii*), loggerhead (*Caretta caretta*), green turtle (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*), and leatherback (*Dermochelys coriacea*). Note: species not shown to scale.

leatherbacks are in the Dermochelyidae family. Kemp's ridleys, hawksbills, and leatherbacks are listed as "Endangered" under the ESA. Green turtles are listed as "Threatened" except for the Florida and Pacific Mexico breeding populations, which are listed as "Endangered." Due to the inability to distinguish the breeding population origin of individuals away from the nesting beaches, green turtles are considered "Endangered" wherever they occur in U.S. waters of the Atlantic, Gulf of Mexico, and Caribbean. Loggerheads in the Gulf of Mexico belong to the Northwest Atlantic Ocean DPS and are listed as "Threatened" under the ESA. In addition to their ESA listing status, several international conservation treaties and agreements (e.g., Inter-American Convention for the Protection and Conservation of Sea Turtles [IAC], Convention on International Trade in Endangered Species [CITES], International Union for the Conservation of Nature [IUCN] *Red List of Threatened Species*™) reflect their status as species considered to be in danger of extinction if current threats are not reduced.

Although sea turtles have survived threats to their existence over their millions of years on Earth, are geographically widespread, and are seemingly abundant, human threats have significantly reduced many sea turtle populations in less than a century (Bjorndal & Jackson 2003). Sea turtles are particularly susceptible to anthropogenic threats because their life history traits (e.g., slow-growing, late-maturing, long-lived, do not reproduce every year) increase their vulnerability at the population level (Musick 1999). Turtles frequently become accidentally entangled, ensnared, and hooked in fishing gear, including in trawls, nets, traps/pots, and on hook and line, and many of these interactions are fatal (Lewison et al. 2013). Humans consume turtle eggs, meat, and other products for subsistence and commercial purposes (Wallace et al. 2011). Coastal development can alter or destroy sea turtle nesting habitat, thereby hindering nesting, as well as reducing embryo and hatchling survival (Wallace et al. 2011). This combination of threats from humans and the unique life history traits of sea turtles makes their populations prone to rapid declines with slow recoveries from significant negative impacts. For these reasons, sea turtles require long-term, consistent, effective protection to prevent further population declines and possible extinction.

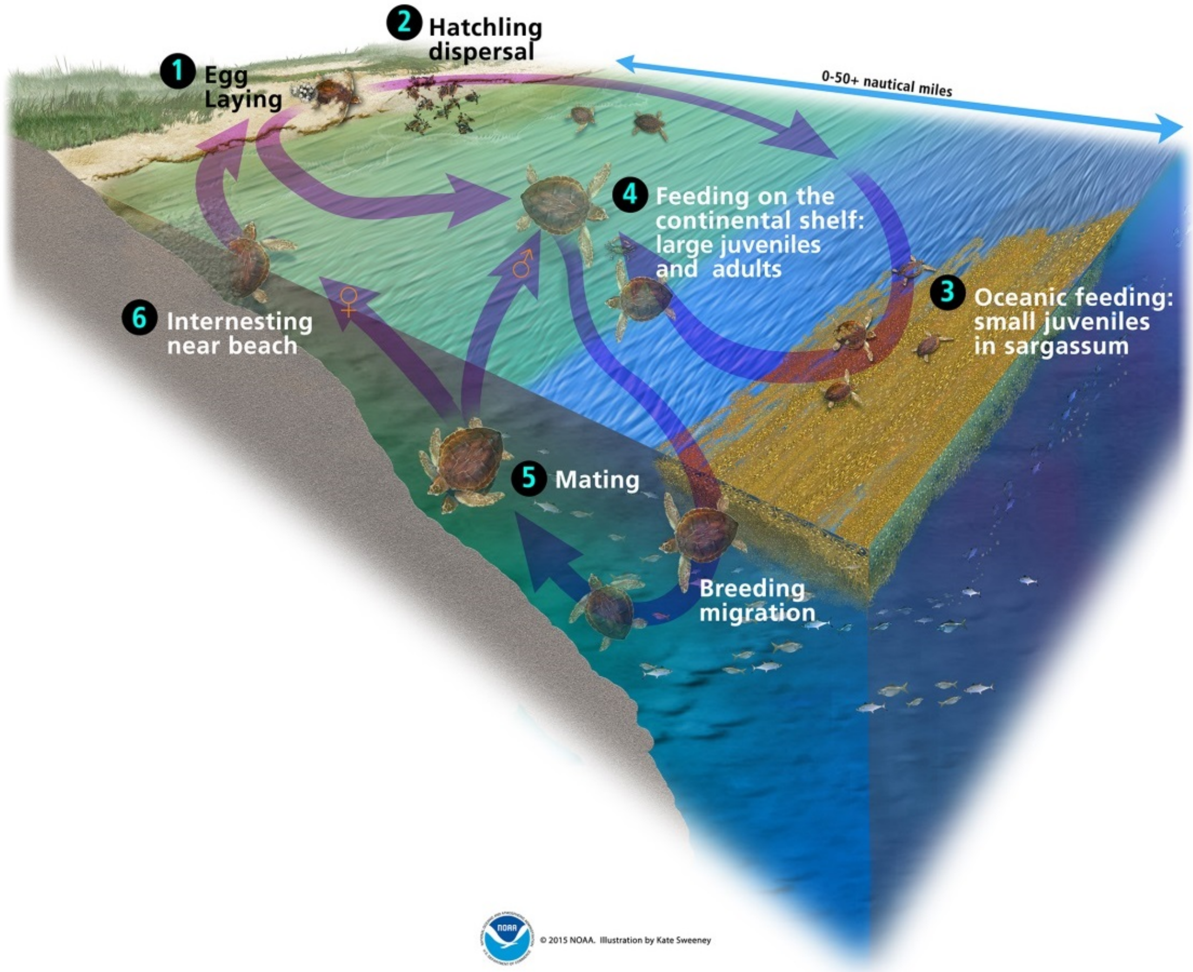
4.8.1.1.4 Sea Turtle Life Stages and Habitat Use

Within their expansive ranges, sea turtles occupy different habitats based on life stages and breeding phase. To carry out their multi-decadal life cycle, different sea turtle life stages require vast areas of different types of marine habitats. Threats can differentially affect specific sea turtle life stages depending on where they overlap in space and time (Bolten et al. 2011). Therefore, given the extensive nature of the DWH oil spill, it is critical to understand how different life stages are distributed, and how different species of sea turtles use habitats in these different areas, in order to assess impacts of the DWH oil spill. Consequently, the Trustees assessed injury to sea turtles by species and life stage. This approach is also most appropriate for developing restoration approaches to compensate for the full extent of the injury across species, life stages, and geographic areas, and for restoring sea turtles to their role in the Gulf of Mexico ecosystem.

The sea turtle life cycle (Table 4.8-1; Figure 4.8-2) begins at egg laying on nesting beaches, followed by hatchling emergence and entry into the ocean, and continues as small juvenile turtles associate with convergence zones in open-ocean areas, where they feed, grow, and evade predation for several years (Bolten 2003). Turtles in this life stage remain at or near the surface associated with floating material, specifically *Sargassum* habitats (Witherington et al. 2012). After this open-ocean (i.e., oceanic) phase, turtles recruit to continental shelf (i.e., neritic) areas, where they continue growing to larger sizes over several additional years, even decades, as in the case of loggerheads, green turtles, and hawksbills (Bolten 2003). Turtles mostly remain in continental shelf areas for the rest of their lives. Leatherback turtles are an exception and continue to frequent both the continental shelf and distant offshore waters. Apart from adult females, which come ashore approximately every 2 or 3 years to lay eggs several times in a season, sea turtles remain at sea for their entire lives, showing site-fidelity to selected foraging grounds (Hart et al. 2014; Shaver et al. 2013). A summary of the sea turtle life cycle and habitat use by different life stages is presented in Bolten (2003). Oil exposures documented in these areas inhabited by different life stages are described in Section 4.8.3 (Exposure).

Table 4.8-1. Summary of sea turtle life stages and habitats discussed in this section.

Life Stage	Habitat	Description of Turtles in This Stage
Nesting females, eggs, hatchlings	Northern Gulf of Mexico sandy beaches mainly in Florida, Alabama, Texas, and Mexico	Nesting female turtles; embryos develop while buried in sand; hatchlings emerge and enter the ocean
Small juveniles	"Oceanic": open ocean; surface habitats throughout the northern Gulf of Mexico	Spend more than 80 percent of their time at or near the sea surface; limited diving ability; tend to associate with floating *Sargassum*; drift and swim to remain in surface currents
Large juveniles and adults	"Neritic": Continental shelf; nearshore and inshore habitats	Use the entire water column, from surface to bottom; active swimmers; dive frequently and typically deeper than 20 meters; spend on average 10 percent of time at the surface; consistently use the same breeding and foraging areas; actively migrate to breed (adults)



Source: *Kate Sweeney for NOAA.*

Figure 4.8-2. Generalized sea turtle life cycle**.** 1) The life cycle starts with egg laying. 2) Hatchlings then leave nesting beaches and swim away from the coast to reach oceanic (i.e., offshore, depths typically > 200 meters) areas, 3) where they remain for several years associated with *Sargassum* and other surface habitats. 4) After growing to larger body sizes, they move onto the continental shelf and closer to shore until reaching adulthood. 5) Adults perform breeding migrations to the areas where they were born, sometimes across oceanic areas, to find mates. 6) Adult male turtles return to foraging areas after mating, while adult females remain during nesting seasons that can last 1 to 2 months for each female. Hatchlings emerge from eggs laid on sandy beaches, which initiates a new cycle.

4.8.2 Approach to the Assessment

Key Points

- The Trustees focused their assessment in areas that sea turtles must use to fulfill their physiological and life history requirements (e.g., sea surface, nesting beaches).
- To assess injuries to sea turtles caused by the DWH oil spill, the Trustees:
  1. Conducted surveys to document sea turtles in oil throughout the northern Gulf of Mexico.
  2. Characterized exposure pathways and the severity of exposure.
  3. Estimated total numbers of turtles exposed.
  4. Determined that adverse effects from exposure caused injuries.
  5. Estimated the total numbers of turtles injured by oil exposure.
  6. Estimated the total numbers of turtles injured by response activities.
  7. Summed all injuries to sea turtles caused by the DWH oil spill.

4.8.2.1 Rationale: Why the Trustees Assessed DWH Impacts on Sea Turtles

Sea turtles occupy various habitats throughout the northern Gulf of Mexico for growth and reproduction, and they presently face all of the threats described in Section 4.8.1.1.3 (Conant et al. 2009; NMFS et al. 2011). It is in this context that the DWH disaster—the largest offshore oil spill in U.S. history—occurred, during which oil moved far and wide throughout the Gulf. Oil impacted over 110,000 square kilometers of the ocean surface, over 2,100 kilometers of shoreline, more than 1,030 square kilometers of deep-sea sediments, and deep-sea water within a plume that extended more than 400 kilometers from the failed well (see Section 4.2, Natural Resource Exposure).

This extensive oiling contaminated vital foraging, migratory, and breeding habitats at the surface, in the water column, and on the ocean bottom throughout the northern Gulf of Mexico for Kemp's ridleys, loggerheads, green turtles, hawksbills, and leatherbacks, across geographic areas used by different life stages. In fact, DWH oil contaminated areas that are currently designated as "Critical Habitat" under the ESA for loggerhead sea turtles in the northern Gulf of Mexico. The pervasive and prolonged nature of the DWH spill, particularly at the air-water interface where all sea turtles must go to breathe, made exposure to oil inescapable for many sea turtles and caused significant injuries to sea turtle populations in the northern Gulf of Mexico.

Adverse physical and toxic effects to wildlife exposed to oil have been documented extensively (e.g., Helm et al. 2015; Leighton 1993; Munilla et al. 2011; Peterson et al. 2003; Piatt & Ford 1996; Shigenaka 2003). DWH oil was no exception, as discussed in Section 4.3, Toxicity. Exposure to DWH oil caused a variety of negative effects, including mortality, on a wide range of species native to the northern Gulf of Mexico. The remote location of the wellhead within deep waters distant from shore meant that organisms such as sea turtles that inhabit offshore areas were vulnerable to DWH oil exposures. This necessitated different assessment and response approaches than those used in previous spills that occurred closer to shore and in smaller areas.

Although oiling and mortality of sea turtles have been documented during previous spills in various locations around the world, detailed information, especially with regard to adverse effects of oil, is generally sparse (Shigenaka 2003). The DWH oil spill overlapped with vital habitats for sea turtles, particularly those of small juvenile turtles that are restricted to ephemeral habitats off shore, at or near the surface (i.e., *Sargassum* habitats and associated floating material; described in Section 4.8.1.1.4, Sea Turtle Life Stages and Habitat Use). The Trustees recognized that surface oil would accumulate in these convergence zones (i.e., areas where surface currents come together), which also brings together anything floating at the surface, such as *Sargassum*, sea turtles, and oil—and therefore would pose a significant risk to turtles in this life stage. As the spill progressed and oil moved into continental shelf and nearshore areas, the Trustees recognized that larger, older sea turtles in these areas, as well as sea turtle nesting beaches, would also be exposed to DWH oil.

In general, the Trustees focused assessments in areas that sea turtles must use to fulfill their physiological and life history requirements. The remainder of this section summarizes the overall approach that the Trustees followed to determine exposure and injury to sea turtles and to quantify those injuries.

4.8.2.2 Conceptual Model of the Approach to the Sea Turtle Injury Assessment

To assess injuries to sea turtles caused by the DWH oil spill, the Trustees evaluated exposures of sea turtles to DWH oil (described in Section 4.8.3), determined adverse effects caused by oil exposure (described in Section 4.8.4), and quantified the magnitude of injuries to sea turtles across life stages and geographic areas in the northern Gulf of Mexico (described in Section 4.8.5). The conceptual model of the approach to the sea turtle injury assessment is presented in Figure 4.8-3.

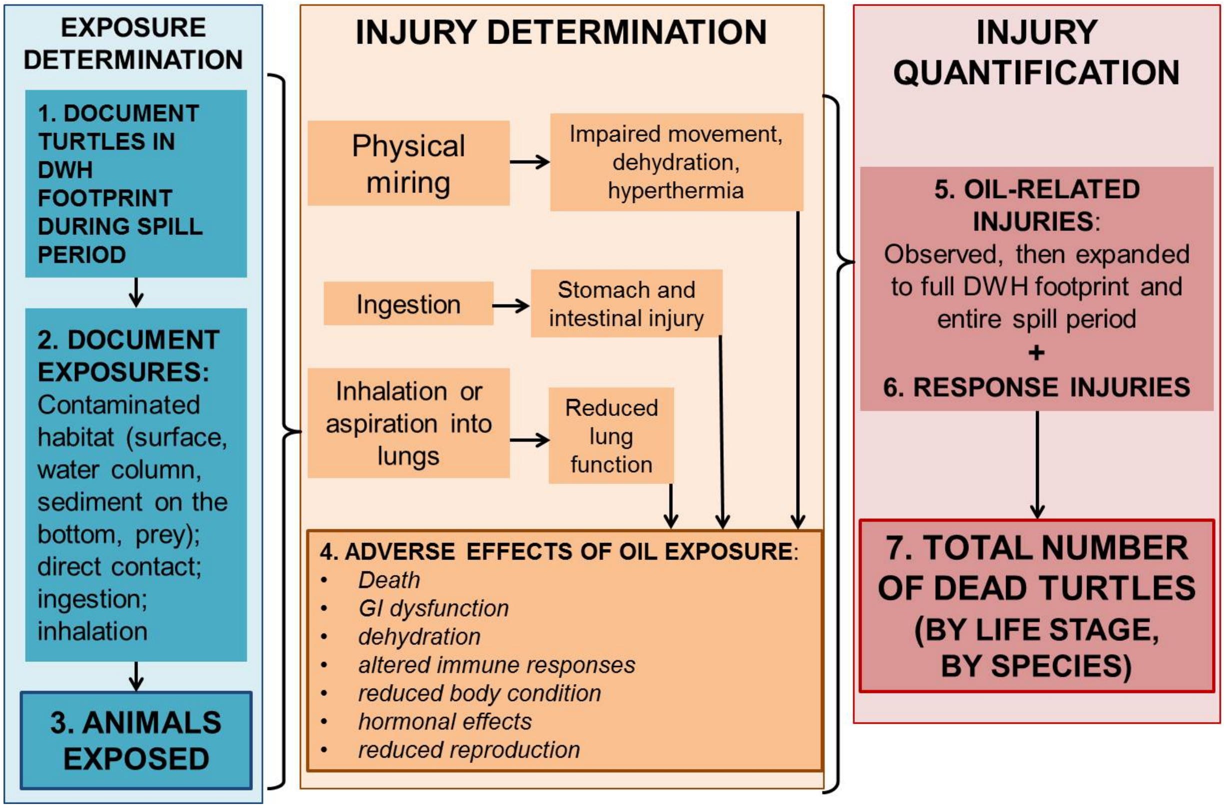


Figure 4.8-3. Conceptual model of the approach to the sea turtle injury assessment. Details for each box are provided in the sections below.

4.8.2.2.1 Exposure Determination

The Trustees conducted rescue and survey operations at sea, from the air, and on the ground to document and evaluate sea turtle exposures to DWH oil.

1. **To document turtles in the DWH footprint** during the spill period, the Trustees:

   - Performed boat-based rescue operations of juvenile turtles in offshore convergence zones (Figure 4.8-2). These efforts involved more than 1,200 transects that covered nearly 200 square kilometers of search area (McDonald et al. 2015).

   - Conducted aerial surveys to document large juvenile and adult sea turtles in relation to oil throughout continental shelf areas in the northern Gulf of Mexico. Nearly 250 transects covered approximately 6,600 square kilometers within the DWH oil spill footprint and on the continental shelf (Garrison 2015).

   - Deployed satellite transmitters on adult female turtles to track their movements and habitat use after the nesting season ended to identify overlaps between high-use areas and the DWH oil spill footprint (Hart et al. 2012; Hart et al. 2014; Shaver et al. 2013).

   - Monitored coastlines in search of stranded turtles and performed necropsies on dead, stranded turtles for signs of oil exposure (Stacy 2012, 2015; Stacy & Schroeder 2014).

- Monitored beaches for female turtles, nests, eggs, and hatchlings to document potential exposure to DWH oil. Evaluations included chemical analyses of blood from nesting female loggerhead and Kemp's ridley sea turtles to determine possible exposure to DWH oil, as well as monitoring of sea turtle nests on impacted beaches for the presence of oil on or in nests and eggs (Hooper & Schmitt 2015).

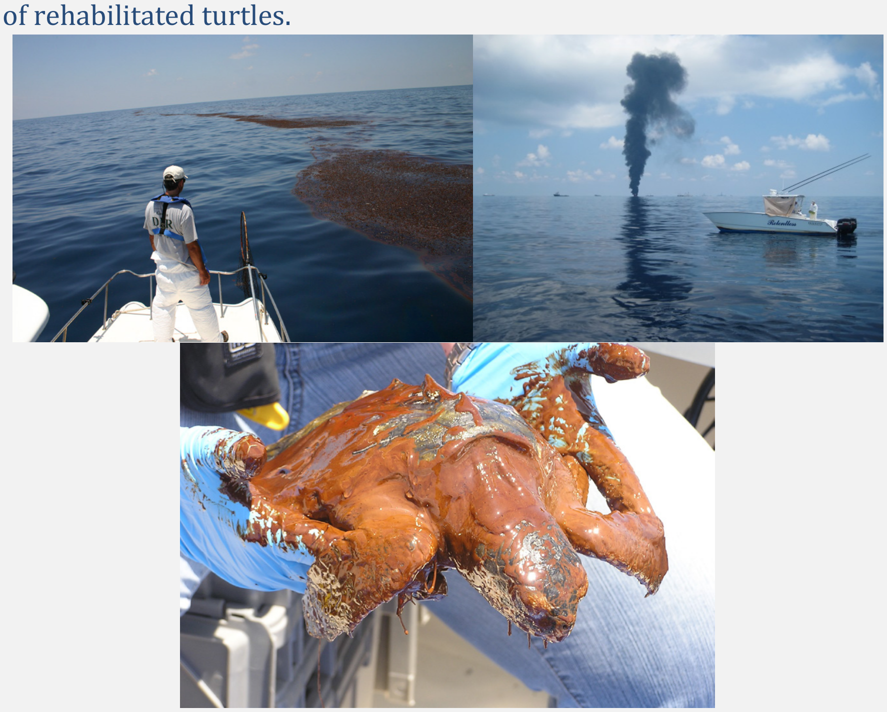
2. **To document sea turtle exposures,** the Trustees characterized exposure pathways and severity of exposure. Veterinarians assessed the conditions of turtles rescued from the Gulf during the spill and assigned turtles to oiling categories based on the extent of external and internal oiling. Toxicologists analyzed tissues from oiled turtles rescued at sea or found as beach-cast strandings, as well as nesting females and their eggs and hatchlings, for signs of oil exposure. The Trustees analyzed composition and concentration of polycyclic aromatic hydrocarbons (PAHs) in liver, lung, stomach and colon content and tissues, and bile collected from turtles captured during rescue efforts and from stranded turtles.

3. **To estimate sea turtles exposed during the DWH spill,** the Trustees used statistical techniques to estimate the magnitude of sea turtle exposures and injuries in both marine and terrestrial environments. Extrapolations were necessary because the field-based observations represent only a sample of the full extent of space and time in which sea turtles occurred and were exposed to oil.

Offshore Rescue Operations Documented Oil Exposure of Hundreds of Sea Turtles

Between May 17 and September 9, 2010, the Trustees undertook rescue operations in an effort to save sea turtles in the spill area (Stacy 2012). Observers in aircraft aided boat-based efforts by communicating locations of oil and *Sargassum*. Boat-based rescue crews searched lines of floating material with oil at low speeds in search of turtles. Crews searched nearly 250 transects totaling nearly 200 square kilometers. Searched areas typically included floating petroleum, emulsified oil, *Sargassum*, and flotsam such as marsh reeds and plastics (Figure 4.8-4). Turtles were either removed from the oil or water using a dip-net, or evaded capture by diving, often beneath surface oil and *Sargassum*. Once turtles were brought aboard, they were examined, the oil was sampled and partially cleaned from the eyes and body, and photographs were taken (Figure 4.8-4).

From May through the beginning of August 2010, turtles that were rescued were taken to rehabilitation facilities for further health assessments, treatment, and monitoring (Stacy & Innis 2012). The Trustees examined more than 300 turtles and characterized any potentially abnormal medical conditions or physiological abnormalities. More than 90 percent of the turtles that were admitted to rehabilitation centers eventually recovered and were released (Stacy 2012). However, long-term condition and survival of oiled turtles treated in rehabilitation centers are not representative of outcomes for oiled turtles in the wild that did not benefit from rescue and treatment (Stacy & Innis 2012). See Section 4.8.4 (Injury Determination) for details about clinical assessments of rehabilitated turtles.



Source: *B. Witherington (top left), M. Dodd (top right), T. Hirama (bottom).*

Figure 4.8-4. Boat-based efforts during the DWH oil spill focused on offshore areas that are inhabited by small juvenile sea turtles. Photos: (top left) Trustees searched convergence areas, which accumulate floating material, typically *Sargassum* and associated fauna, including sea turtles, as well as DWH oil; (top right) responders performed boat-based operations in offshore areas to rescue small juvenile sea turtles that inhabited convergence areas affected by the oil; (bottom) a heavily oiled, small juvenile Kemp's ridley turtle rescued during the spill.

4.8.2.2.2 Injury Determination

The Trustees combined veterinary assessments of oiled turtles that were rescued and rehabilitated with various data sources related to physical and toxicological adverse effects of oil exposure on sea turtles to determine the extent and severity of injuries related to oil exposures caused by oil.

1. **To determine adverse effects of oil exposure** on sea turtles, the Trustees conducted studies and synthesized information on physiological, toxicological, and laboratory studies, as well as veterinary assessment of recovered live and dead turtles to quantify mortality estimates for turtles based on the degree of oil exposure.

4.8.2.2.3 Injury Quantification

Direct observations of turtle exposures in DWH oil only covered a small portion of the entire oil spill footprint and spill period. Therefore, the Trustees combined these direct observations and injury determination with statistical techniques to extrapolate from surveyed areas and times to the entire footprint and spill period to estimate the full magnitude of sea turtle injuries.

2. **To determine the magnitude of oil-related injuries,** biologists, geospatial analysts, and statisticians constructed models to quantify total sea turtle injuries within the entire spill area and for the full duration of the spill. As with exposure, these models were necessary because direct observations represented only a small portion of the large area affected by the DWH oil spill. These approaches allowed the Trustees to calculate the total numbers of turtles killed by oil exposure.

3. **To determine the number of response injuries,** the Trustees also quantified injuries due to response activities that directly injured turtles or deterred reproduction in marine and terrestrial areas.

4. **To quantify the total injuries, by life stage and by species,** the Trustees summed injuries to sea turtles caused by exposure to DWH oil and caused by DWH response activities to produce an overall estimate of turtle injuries by life stage (i.e., geographic area) and by species. These quantification metrics facilitate effective restoration of sea turtles in the northern Gulf of Mexico.

4.8.2.3 Summary of Approach to the Assessment

The remote location and pervasive nature of the DWH oil spill required the Trustees to undertake several methods to assess sea turtle exposures to oil and injuries caused by oil throughout the spill footprint and response activities. The Trustees combined direct observations obtained by various types of surveys and studies (e.g., vessel-based, aerial, ground-based, veterinary, toxicological) with statistical extrapolation techniques to assess and estimate the full magnitude of the DWH oil spill effects on sea turtles. In the next section, we describe results from Trustees' efforts to document sea turtle exposure to DWH oil.

4.8.3 Exposure

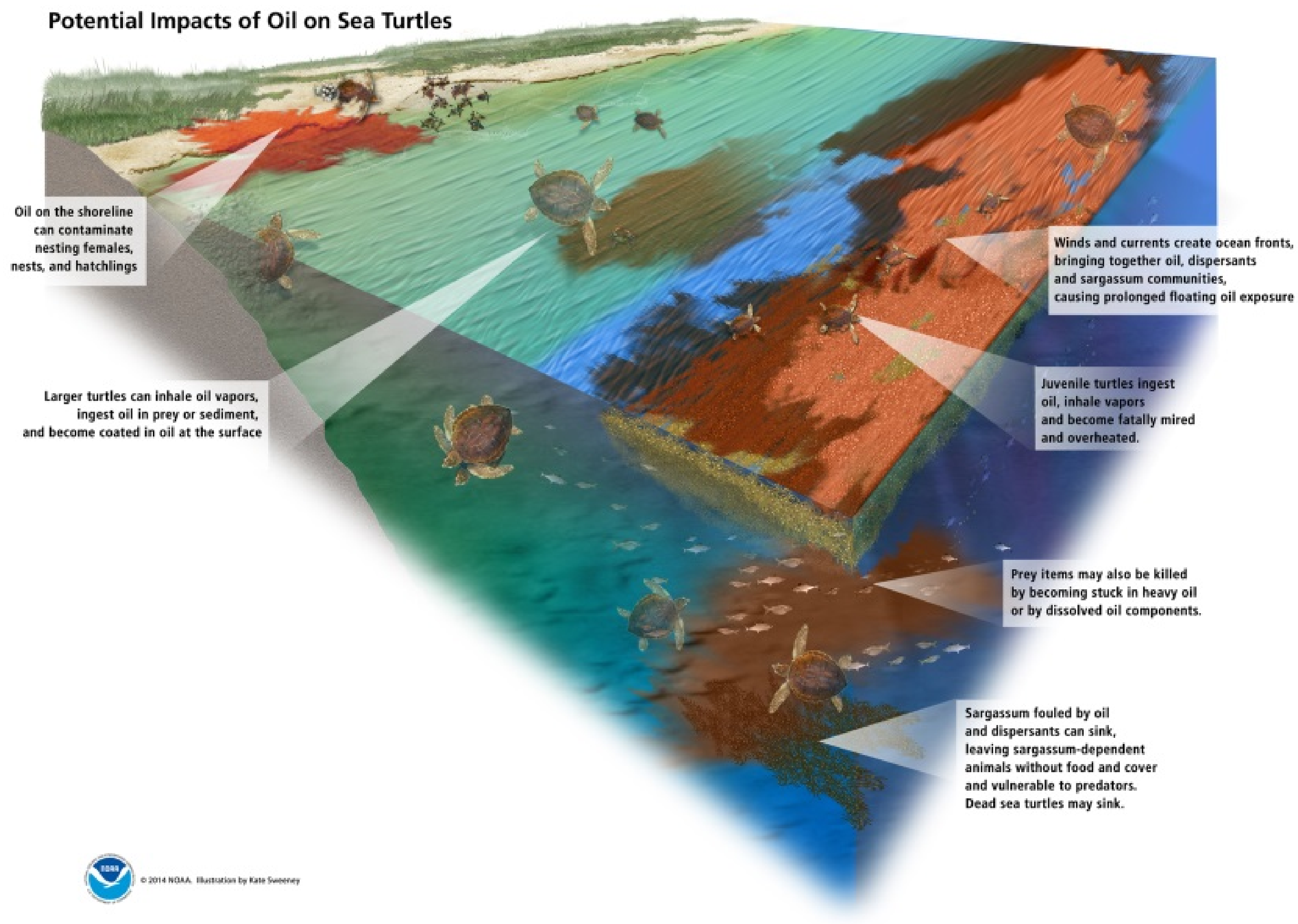
Key Points

- The DWH oil spill contaminated critical turtle habitats throughout the northern Gulf of Mexico, especially the sea surface, for extended periods of time.
- Sea turtles likely were exposed to oil through a variety of pathways:
  - Direct contact with oil when swimming at or near the surface and on nesting beaches.
  - Inhalation of oil, oil vapors, and smoke.
  - Ingestion of oil-contaminated water and prey.
  - Transfer of oil compounds from adult females to their developing embryos.
  - Oil contamination of essential turtle habitats.
- The Trustees used a multi-faceted, multi-scale approach to determine the extent of sea turtle exposure to DWH oil.
  - The Trustees observed nearly 1,800 turtles within the DWH oil footprint based on vessel- and plane-based surveys.
  - Turtle exposure to DWH oil was confirmed based on direct observation, analytical chemistry, and surface oiling data obtained by satellites.
  - The Trustees assessed the degree of external oiling of turtles, and developed categories for the degree of oiling for use in the injury determination and quantification.
  - The Trustees observed that turtles that were externally oiled generally had also ingested oil, demonstrating exposure via multiple pathways.

This section describes how sea turtles were exposed to DWH oil as it spilled into the Gulf of Mexico from the area of the wellhead and spread throughout the northern Gulf of Mexico. Utilization of both marine and terrestrial habitats by different life stages resulted in exposure in various habitats and by a variety of external and internal pathways. In marine areas, sea turtles were exposed to surface oil and to oil beneath the surface. In terrestrial areas, turtles were potentially exposed to oil on nesting beaches and potentially by maternal transfer of oil compounds to embryos. Because the DWH oil spill contaminated critical sea turtle habitats throughout the northern Gulf of Mexico, sea turtle exposure to oil was pervasive, severe, long-lasting, and—for many turtles—inescapable (Garrison 2015; McDonald et al. 2015; Stacy 2012).

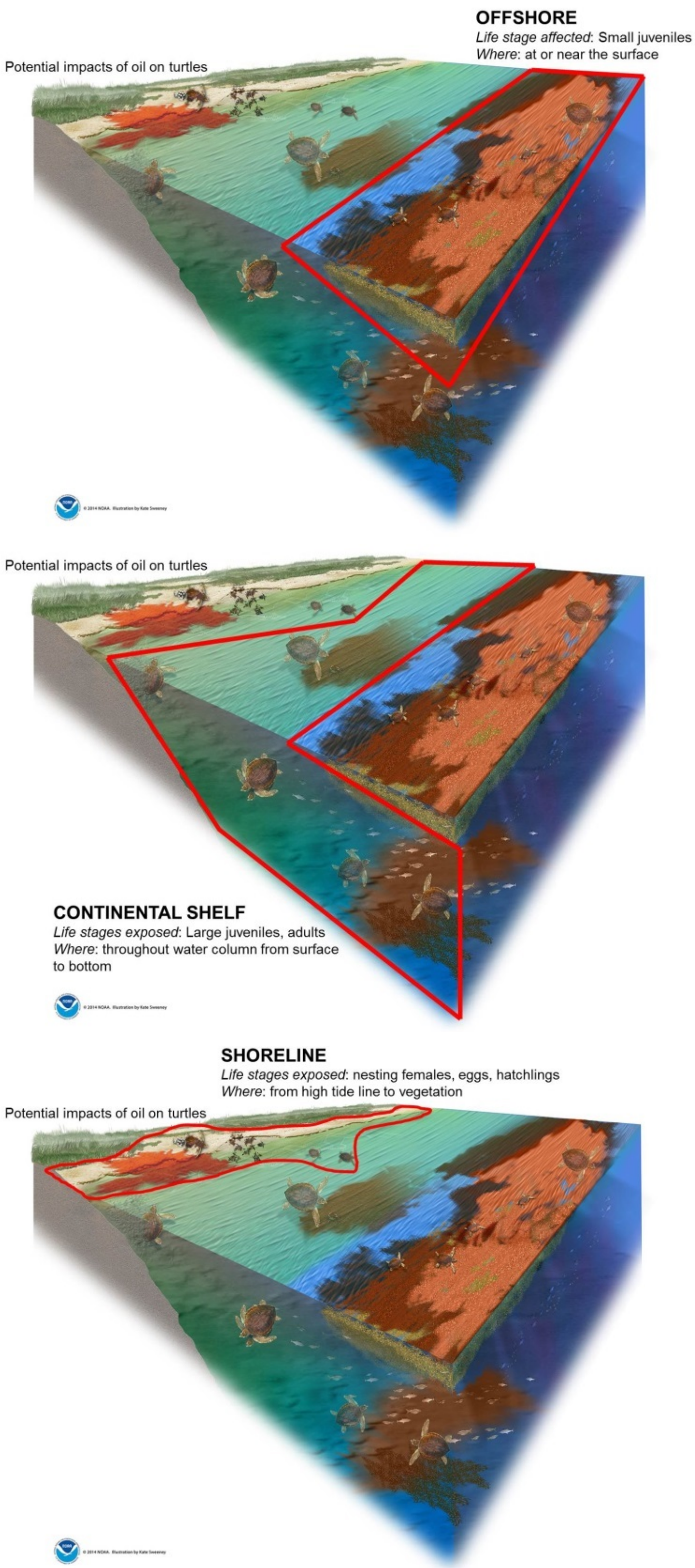
4.8.3.1 Sea Turtle Oil Exposure Followed the Oil Spill Trajectory in the Gulf of Mexico

Figure 4.8-5 shows the potential impacts of DWH oil in the northern Gulf of Mexico. As DWH oil contaminated offshore waters and later spread onto the continental shelf and coast and into inshore waters, exposure to and potential impacts of oil on sea turtles mirrored the trajectory and advance of the spill. Figure 4.8-6 illustrates the progression of oil and resulting exposures of the different life stages within the sea turtles' predominant habitat. Small juvenile sea turtles were first exposed to oil in offshore areas beyond the continental shelf in the early period of the spill and throughout the duration of the free-release period (Figure 4.8-6, top panel). As the oil moved onto the continental shelf, larger, older neritic turtles were exposed in known foraging, migratory, and breeding areas (Figure 4.8-6, middle panel). When oil came ashore on coastal shorelines, turtle nesting beaches—along with nests and hatchlings—were also potentially exposed (Figure 4.8-6, bottom panel).



Source: *Kate Sweeney for NOAA.*

Figure 4.8-5. Potential impacts of DWH oil on sea turtles in the northern Gulf of Mexico. Text boxes highlight specific details about potential exposure pathways and adverse effects to turtles in their different critical marine and terrestrial habitats.



Source: *Kate Sweeney for NOAA.*

Figure 4.8-6. Potential impacts of DWH oil on turtles in offshore (top), continental shelf (middle), and shoreline or terrestrial habitats (bottom) in the northern Gulf of Mexico.

4.8.3.2 Potential Exposure Pathways

Sea turtles likely were exposed to oil via several external and internal pathways in marine and terrestrial areas throughout their distribution in the Gulf of Mexico. Surface oil, airborne oil compounds (e.g., volatiles and semi-volatiles, aerosols), oil in the water column, and oil in nearshore sediments were all likely sources of exposure to DWH oil throughout the northern Gulf of Mexico from offshore areas to the shorelines, throughout the duration of the free-release period, and for as long as oil persisted in these areas following the capping of the wellhead. As illustrated in Figure 4.8-5 and Figure 4.8-6, sea turtles in offshore as well as continental shelf waters were exposed to oil by direct contact at or near the surface; ingestion of oil while eating or drinking contaminated food, water, or sediment; and inhalation of oil vapors while breathing at the surface. Turtles also could have been affected via contaminated habitat or reduced prey availability or quality, particularly in sub-surface habitats. Although all of these exposures are probable, some of these pathways could not be specifically incorporated into the assessment due to insufficient available information. Consequently, the exposure assessment focused on the extensive evidence of sea turtle exposures in oil at or near the surface, which were readily observable and clearly documented.

The assessment focused on the following potential exposure pathways:

- **Direct contact.** Turtles came into direct contact with oil when swimming at or near the surface, which caused their bodies to become coated with oil wherever contact occurred. Physical fouling of eyes, nares (i.e., nasal openings), and mouth resulted in ingestion of oil and exposure of sensitive mucus membranes. All life stages are at risk for direct contact exposure due to their inherent need to surface to breathe, although such exposure is especially acute for smaller juveniles that spend nearly all of their time in the top 2 meters of the water column and inhabit ocean convergence fronts formed by wind and currents, which also accumulate oil (Shigenaka 2003). For sea turtles, becoming mired in oil had drastically negative effects, including impeded locomotion and diving ability; decreased ability to feed and avoid predators; hyperthermia (overheating) in the heavy, dark oil; among others.

  On beaches, nesting turtles and their eggs and hatchlings were potentially exposed to oil through direct contact with oiled substrate on sand beaches. Additionally, oil compounds absorbed by embryos developing in contaminated sand could affect development and survival.

- **Ingestion.** Sea turtles are well-known to ingest petroleum and other anthropogenic material, possibly due to indiscriminant feeding behavior or mistaking such items for prey (Camacho et al. 2013; Witherington 2002). Ingestion of oil or oil-related compounds would have exposed turtles to adverse toxicological effects, such as function of vital physiological systems (Lutcavage et al. 1995). These effects could impair turtle health, growth, and survival.

- **Inhalation.** Turtles rely on oxygen inhaled right above the water’s surface, and thus were exposed to inhalation and/or aspiration of oil droplets, oil vapors, and smoke from burning oil when they surfaced to breathe in contaminated areas. Similar to potential effects described for marine mammals (Section 4.9), inhalation exposure would decrease respiratory and cardiovascular function, and thus hinder turtles’ abilities to dive efficiently to forage, escape predators, find mates, migrate, etc. This effect could worsen as stressed, oxygen-deprived turtles surfaced more frequently to breathe, still within surface slicks, thereby becoming exposed repeatedly. Inhalation exposure is a well-recognized concern among spill workers and other air-breathing animals (see Section 4.9, Marine Mammals).

- **Maternal transfer.** Animals that lay eggs have been shown to pass metabolized, oil-related compounds onto their offspring, which have the potential to be toxic to developing embryos (Pereira et al. 2009; West et al. 2014). Similarly, adult female turtles could have passed metabolized oil and related products into their eggs, thereby exposing developing embryos. These oil-related contaminants could have impaired the development and survival of embryos.

- **Exposure of turtle habitats.** Oil contamination of essential turtle habitat (e.g., *Sargassum* habitats), nearshore sediments, the water column, and shorelines are also highly relevant to the effects of the DWH spill on sea turtles due to the potential effect on prey items and other marine organisms and implications on the northern Gulf of Mexico food web. In addition, environmental oiling contributes to all of the above routes of exposure. For example, larger turtles feeding on the ocean floor will incidentally consume sediment with food items (Lazar et al. 2011; Preen 1996).

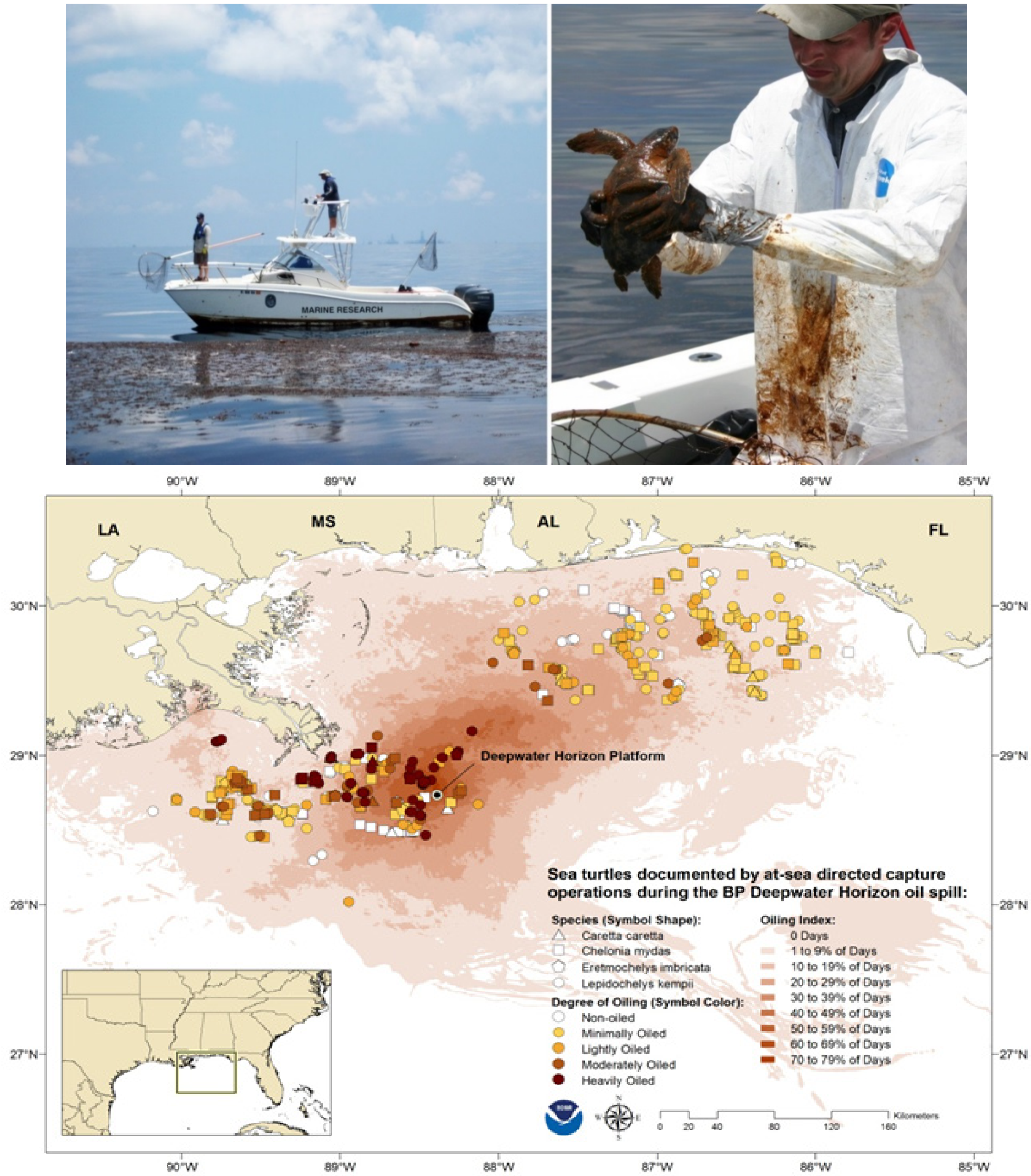
4.8.3.3 Sea Turtles Were Observed in Oil Throughout the Northern Gulf of Mexico

The Trustees synthesized results from boat-based rescue efforts, aerial surveys, and other observations with various data sources to document and quantify sea turtle exposure to DWH oil.

4.8.3.3.1 Directed Captures and Observations of Oiling in Offshore Areas

Trustee rescue teams performed active searches on more than 1,200 transects totaling over 4,200 linear kilometers and an area of nearly 200 square kilometers within potential turtle habitats to locate and capture turtles from the ocean surface (Figure 4.8-7). These directed capture efforts primarily targeted oceanic juvenile turtles within offshore convergence zones, which were considered to be under the greatest imminent threat from the spill. More than 900 turtles were sighted, 574 of which were captured and examined for oiling (Stacy 2012). Figure 4.8-7 shows boat-based rescue efforts, assessment of heavily oiled sea turtles, and locations of turtles captured and assessed during rescue operations. Of the turtles captured during rescue operations, 464 (> 80 percent) were visibly oiled (Table 4.8-2), and the quantity of oil collected from 199 oiled turtles was sufficient to identify the material as MC252 oil (Stacy 2012; Stout 2014). This high proportion of captured turtles that were oiled demonstrates the widespread inundation of offshore sea turtle habitat by oil.

From May through the beginning of August 2010, most of the turtles that were rescued were taken to rehabilitation facilities for further medical evaluation, treatment, and monitoring (Stacy & Innis 2012). Five turtles, three of which were oiled, were found dead during directed capture activities; four more were found alive, but died later. Details about clinical evaluation of rehabilitated turtles and postmortem examinations related to assessment of injury are presented in Section 4.8.4.1 (Physical Effects).



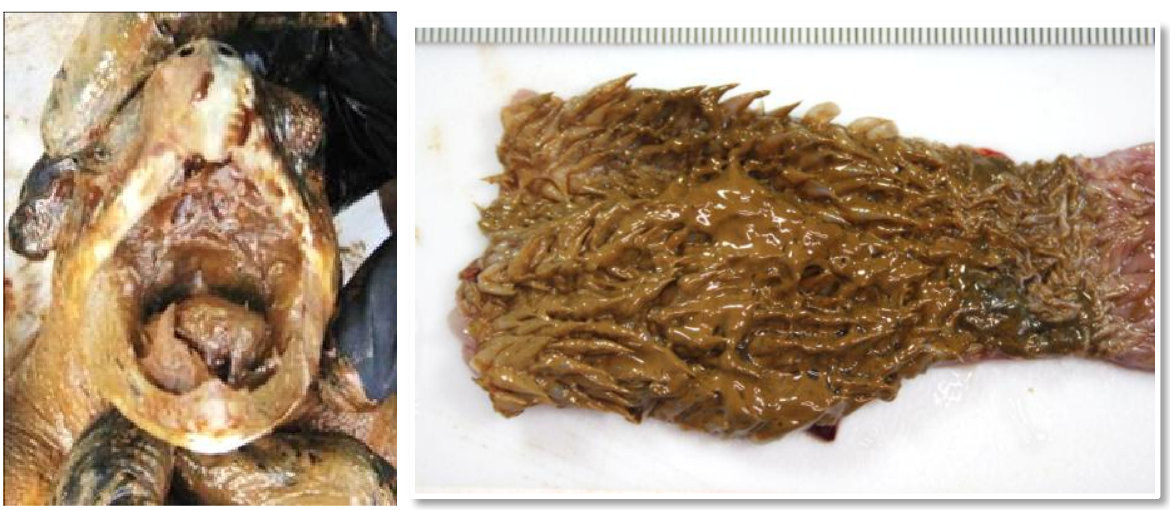
Source: *B. Witherington (top left), T. Hirama (top right).*

Figure 4.8-7. Boat-based rescue efforts documented more than 900 sea turtles in the DWH spill zone. Photos: (top left) Boat-based rescue efforts on search transects within offshore convergence areas that are known habitats for small juvenile sea turtles; (top right) the NOAA veterinarian assessing the condition of heavily oiled sea turtles rescued from oiled surface habitat; (bottom) locations of turtles captured and assessed during rescue operations, shown by species and degree of oiling, overlaid upon cumulative oil-days within the overall oiling footprint. Nearly all heavily oiled turtles were found within 90 kilometers of the wellhead and prior to August 1, 2010.

It is important to note that the turtles documented during rescue operations—especially the number of oiled, dead turtles—underestimate the actual magnitude and degree of oil exposure that affected sea turtles during the DWH oil spill. The underestimation was due to several factors that hindered the ability of field crews to document live and dead turtles during the rescue efforts. Foremost was the vast expanse of the search area and distance from shore, which limited the proportion of the spill area that could be physically searched for small turtles, which are only visible from vessels. Disappearance of carcasses due to sinking of remains, scavenging, and rapid decomposition rates in summer temperatures limited the recovery of dead turtles, as did the difficulty of seeing motionless, oiled small turtles among surface material and oil. In addition, rescue crews were restricted from working early in the spill period, during inclement weather, around the wellhead, and in more distant areas due to logistical constraints and safety concerns.

Assessment of Degree of Oiling

Once turtles were brought aboard rescue vessels, veterinarians and biologists evaluated their general physical condition, photographed the extent of oil coverage, and examined their mouths for oil (Figure 4.8-8). Eighty percent of these animals were visibly oiled to various degrees (Table 4.8-2). To define the relative degrees of visible oil exposure in sea turtles collected during directed capture operations and other activities, veterinarians reviewed field photographs and field notes to evaluate the extent to which turtles' bodies were externally oiled. Based on this evaluation of visible oiling, veterinarians categorized turtles as not visibly, minimally, lightly, moderately, or heavily oiled based on the extent of oil coverage (Stacy 2012).



Source: *B. Stacy.*

Figure 4.8-8. Assessments of externally oiled sea turtles demonstrated that turtles also ingested oil. Photos: (left) Thick crude oil found in the oral cavity and nares of a rescued Kemp's ridley sea turtle; (right) thick crude oil coats the inside of the esophagus of a deceased heavily oiled turtle found during rescue operations. Sea turtles have cornified papillae (i.e., thorny projections lining the inside of the esophagus) that prevent ingested prey from moving back toward the mouth, and perform the same function when viscous oil is ingested. Responders had already removed some oil from this esophagus prior to taking the photograph.

Table 4.8-2. Numbers of sea turtles documented by directed capture operations during the DWH oil spill response by degree of oil coverage, species, and proportion with oil observed in oral cavity (Stacy 2012). "Unknown" turtles did not have sufficient photographic information to be assigned to oiling categories.

Species	Not Visibly Oiled	Minimally Oiled	Lightly Oiled	Moderately Oiled	Heavily Oiled	Unknown	Total
Kemp's ridley	50	141	47	26	51	2	317
Green	49	112	36	17	6	0	220
Loggerhead	6	5	2	3	2	0	18
Hawksbill	5	8	2	1	2	1	19
Total	110	266	87	47	61	3	574
Oil in oral cavity	Not evaluated	49%	76%	93%	97%		

Minimally and lightly oiled turtles had relatively little visible oil on their bodies, whereas moderately and heavily oiled turtles were more extensively covered in thicker, tenacious oil. Heavily oiled turtles were essentially completely covered and heavily mired in oil. Among oiled turtles, approximately 58 percent were minimally oiled, 19 percent were lightly oiled, 10 percent were moderately oiled, and 13 percent were heavily oiled (Figure 4.8-9; Table 4.8-2). Nearly all heavily oiled turtles were found within 90 kilometers (straight-line distance) around and to the west of the DWH platform (Figure 4.8-7). The degree of external oiling decreased among captured turtles from the end of July 2010 through cessation of directed capture efforts in September 2010, coinciding with the apparent dissipation of oil from surface habitats following capping of the well and the end of free-release oil discharge into the Gulf.

Furthermore, veterinarians evaluated the amount of oil observed in oral and nasal cavities of live and dead turtles relative to the categories of visible, external oiling (Figure 4.8-8) (Mitchelmore et al. 2015; Stacy 2012). Ingested oil was found within the mouth, pharynx, and esophagus during oral examinations and necropsies. Oil was ingested via contaminated dietary items or by direct ingestion of aggregated oil that was mistaken for food. Oil adhered to the internal surfaces of turtles' mouths and throats, and required considerable effort to remove in sea turtles brought into rehabilitation centers for de-oiling (Figure 4.8-8). In addition, feces from oiled turtles brought into rehabilitation centers often produced a sheen and included globules of oil-like material.

The extent of oil ingestion was further characterized in dead oiled turtles that were necropsied. Esophagi of these turtles were found to be heavily coated with oil, and oil was also found through digestive tracts, consistent with the defecation of oil observed in the live turtles. Oil that adhered to the lining of the esophagus was a continuous source of continued exposure for days, possibly longer, following initial ingestion. The percentage of turtles in each oiling category that had oil in their oral cavities, as well as the volume of oil present, increased with the degree of external oiling (Table 4.8-2) (Mitchelmore et al. 2015; Stacy 2012). Even very lightly oiled turtles had an almost 50 percent occurrence of ingestion.

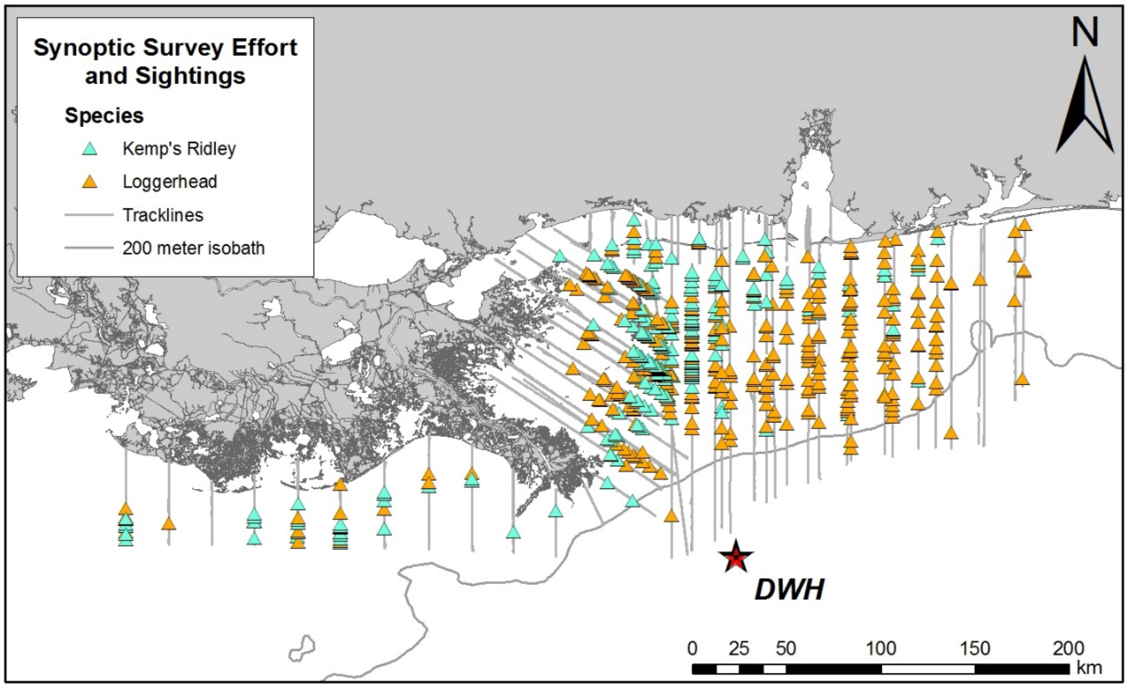
Source: B. Stacy.

Figure 4.8-9. Photographs of turtles in each oiling category defined by extent of external oiling**.** Percentages of turtles documented in each category relative to all turtles assessed are shown next to representative photograph of each oiling category.

These observations were further corroborated by chemical analyses of turtle tissues. Indicators of PAH exposure were higher in tissues (e.g., liver, lung, esophagus), colon content, stomach content, and feces collected from visibly oiled turtles compared to those collected from non-visibly oiled turtles. PAH compositions were consistent with those of weathered DWH oil (Ylitalo et al. 2014). A chemical constituent of dispersants used for detecting exposure was not found in most analyzed samples, but was observed in high concentrations in ingested oil from a turtle found off shore where most dispersant applications occurred (Ylitalo et al. 2014).

4.8.3.3.2 Observations of Turtles on Continental Shelf and on Beaches

The Trustees conducted aerial surveys in the northern Gulf of Mexico on the continental shelf (to the 200-m isobath) in 2010 and again in 2011 to locate and count larger juvenile and adult sea turtles at the surface (Figure 4.8-10) (Garrison 2015). These surveys were designed to allow for calculation of estimates of turtle abundance across the survey area throughout the study period, and subsequently for calculation of overall abundance estimates. Overall, the Trustees surveyed more than 18,000 linear kilometers along nearly 250 transects and searched more than 6,600 square kilometers of total area between April 28 and September 2, 2010. In 2011, the Trustees flew approximately 56,000 kilometers and searched nearly 23,000 square kilometers between spring 2011 and winter 2012 (Garrison 2015).



Source: *Garrison (2015).*

Figure 4.8-10. The Trustees flew aerial surveys to document locations of sea turtles within the DWH oil spill footprint. Triangles indicate all sightings of Kemp's ridleys (blue; n=287 turtles) and loggerheads (orange; n=529 turtles) along all survey transect lines flown systematically from April through September 2010. The location of the wellhead is indicated by the star symbol.

Aerial survey observers could only see turtles larger than approximately 40 centimeters in length, which omitted small juvenile Kemp's ridleys approximately 3 years of age that inhabit continental shelf areas (Avens & Snover 2013). Because these turtles also would not have been in surface habitats targeted by the vessel operations described in Section 4.8.3.3.1 (Directed Captures and Observations of Oiling in Offshore Areas), an alternative approach was used to estimate their abundance and exposure (see Section 4.8.5, Injury Quantification for details).

More than 800 turtles were documented during aerial surveys in 2010. Turtles were present throughout continental shelf waters from April through September, and consistently within the DWH oil footprint. Observations of loggerheads and Kemp's ridleys declined from spring into summer, and then increased again in the fall. High numbers of observations were documented for both species in the eastern portion of the study area between the Chandeleur Islands (Louisiana) and off the coasts of Mississippi, Alabama, and the Florida Panhandle (Figure 4.8-10). An additional area with high numbers of observations for Kemp's ridleys occurred to the west of the Mississippi River Delta (Figure 4.8-10). These areas are generally similar to high-use areas identified by satellite tracking studies of adult female loggerheads and Kemp's ridleys following nesting seasons (Figure 4.8-11) (Hart et al. 2012; Hart et al. 2014; Shaver et al. 2013), and consistent with observations from surveys in 2011 (Garrison 2015). Turtles were observed and photographed in surface oil slicks during aerial surveys. Density and abundance estimates were calculated using these observational data and environmental habitat models (see Section 4.8.5, Injury Quantification).

Biologists also satellite-tracked dozens of adult female loggerhead and Kemp's ridley turtles across several years, including 2010, to identify high-use areas (Figure 4.8-11) (Hart et al. 2012; Hart et al. 2014; Shaver et al. 2013). Loggerheads that nested in the eastern Gulf of Mexico, including the Panhandle of Florida and in Alabama, used foraging areas in the northern Gulf of Mexico throughout 2010; these were also high-use areas in the years before and after the DWH oil spill (Hart et al. 2012; Hart et al. 2014). Similarly, adult female Kemp's ridley turtles migrated to and from foraging areas throughout the continental shelf of the northern Gulf of Mexico and demonstrated foraging site fidelity across foraging seasons (Shaver et al. 2013). These results show that areas identified as important and highly used by both species significantly overlapped with areas of known oil presence.

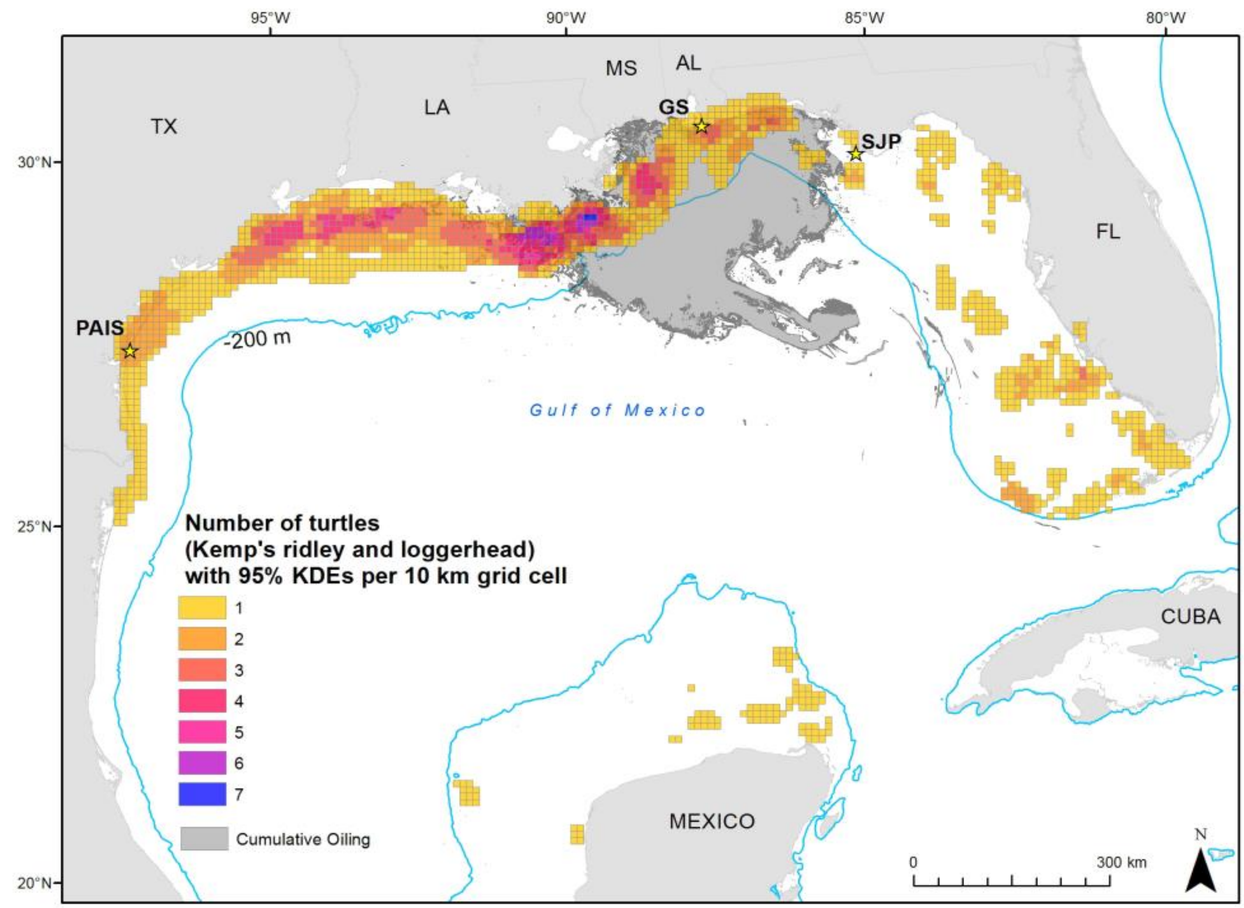


Figure 4.8-11. High-use areas of adult female Kemp's ridleys (n=20 turtles) and loggerheads (n=39 turtles) in the northern Gulf of Mexico determined using satellite telemetry overlapped with the DWH floating (or surface) oil footprint. High-use areas are defined as 95 percent kernel density estimates, which is a statistical summary of where turtle locations are concentrated in space. Colors indicate the number of turtles per grid cell, and the data include turtles tracked between 2010 and 2013. Tagging locations were nesting beaches indicated by stars: Padre Island National Seashore, Texas (PAIS); Gulf Shores, Alabama (GS); and St. Joseph Peninsula, Florida (SJP).

Stranded Turtles on Beaches

Surveillance of coastlines for stranded sea turtles was enhanced during and following the DWH spill. Searchers documented stranded turtles and veterinarians performed postmortem examinations to the extent allowed by postmortem condition (Stacy 2012; Stacy & Schroeder 2014). Among the 644 stranded turtles found in 2010, 24 (3.7 percent) were visibly oiled, and 16 oiled stranded turtles (> 66 percent of oiled turtles) were confirmed to have been oiled by MC252 oil (Stacy 2012; Stout 2014). Most of the oiled stranded turtles were small juveniles of the same size class targeted by the directed capture operations (Stacy 2012). The rest of the strandings were mainly larger juvenile turtles (predominantly Kemp's ridleys) (Stacy 2012). Analyses of biological samples collected from a subset of non-visibly oiled stranded turtles did not suggest recent exposure to petroleum or dispersant-associated compounds (Ylitalo et al. 2014).

As further discussed in Section 4.8.5 (Injury Quantification), strandings are a poor indicator of oil exposures that occurred in much of the DWH spill area as they generally are not representative of events occurring farther from shore or in areas where turtles were unlikely to be found (e.g., wetlands) (Epperly et al. 1996; Hart et al. 2006; Nero et al. 2013; Williams et al. 2011). Many turtle carcasses may sink and never resurface, decompose, be scavenged, drift away from shore, or reach a coastline, but not be detected by observers (see strandings graphic in Chapter 1, Introduction). The few oiled turtles that were documented as strandings likely represent the very rare occasions where turtles originating from deeper and/or more distant waters reach shore. Of the few that are found stranded, most were too decomposed to determine cause of death (Stacy 2012).

Potential Exposure of Nesting Turtles, Their Eggs, and Hatchlings

Oil reached Gulf shorelines in early summer 2010, including sand beaches that host nesting female sea turtles and their eggs and hatchlings. Terrestrial life stages of sea turtles were potentially exposed to oil on beaches through direct contact and maternal transfer of oil compounds to developing egg embryos. As discussed in Section 4.8.4 (Injury Determination), many eggs laid within the spill area were translocated to avoid hatchlings being killed by oil as they swam into the Gulf. Tissue samples were collected from nesting female sea turtles, eggs, and hatchlings in 2010, 2011, and 2012 for chemical and biological analysis to detect exposure and any subsequent physiological, developmental, and toxicological effects. Specifically, the Trustees evaluated PAH concentrations, blood chemistry, sex ratios, and immunological function to determine exposure and impairment of nesting female sea turtles and their eggs and hatchlings. Given the many complexities of response operations and translocation of nests during the oil spill, very little sampling was done during the actual nesting season in 2010. Studies of nesting females, eggs, and hatchlings in subsequent years primarily focused on Kemp's ridleys in Texas and were aimed at detection of ongoing exposure and effects. None of these studies yielded evidence of exposure to DWH oil; however, the limited scale of sampling, uncertainty about application of methods to sea turtles, and the variability in exposure probability among animals that forage in different areas may have prevented detection of possible oil exposure of nesting female sea turtles (Hooper & Schmitt 2015).

4.8.3.4 Summary of Exposure Determination

The Trustees' exposure determination included:

1. Direct observation of approximately 1,800 turtles within the DWH oil spill footprint.

2. Direct observation, analytical chemistry, and remote sensing that confirmed that turtles were exposed to DWH oil.

3. Assessments of exposure pathways based on veterinary examinations and clinical evaluations.

In the next section, we present the Trustees' determinations of the nature and extent of injuries to sea turtles resulting from these exposure pathways.

4.8.4 Injury Determination

Key Points

- The Trustees concluded that sea turtles throughout the northern Gulf of Mexico suffered adverse effects, including death, from DWH oil exposure and response activities.

- Miring in oil and exposure to oiled surface habitat caused significant harm to sea turtles, including decreased mobility, exhaustion, dehydration, overheating, likely decreased ability to feed and evade predators, and death.

- The Trustees determined that chronic toxic effects of oil and indirect sub-lethal effects on reproduction and health likely resulted in injury, though these effects are less well-understood.

- Response actions undertaken to remove oil from the beaches and the ocean resulted in direct injuries to turtles in all areas of the northern Gulf of Mexico. Translocation of eggs from the Gulf of Mexico to the Atlantic coast of Florida resulted in the loss of sea turtle hatchlings.

- Other response activities, including vessel strikes and dredging, also resulted in turtle deaths.

As presented in Section 4.8.3 (Exposure), sea turtles were exposed to DWH oil via several pathways. This section describes the Trustees' assessments of adverse physical and toxicological effects resulting from those exposures. Overall, the Trustees concluded that conditions resulting in heavy oiling presented a clear and imminent threat to sea turtles, and increased probability of mortality. This determination is based on the Trustees' conclusions of increased adverse effects from physical fouling in oil, toxicity of oil, and contamination of turtle prey and foraging habitat. The Trustees also determined that turtles were injured by response activities, such as cleanup operations on oiled beaches; translocation of eggs from Gulf beaches to the Atlantic coast of Florida; and by activities in marine areas, such as dredging and response-related boat traffic.

4.8.4.1 Physical Effects

4.8.4.1.1 Physical Fouling

Physical fouling in oil caused significant harm to sea turtles. As observed directly during the rescue of small juvenile turtles from surface oil and described in Section 4.8.3.3.1 (Directed Captures and Observations of Oiling in Offshore Areas), physical fouling had the most readily apparent, immediate effect of oil on sea turtles. The conditions from which heavily oiled turtles were rescued were extremely grave; turtles were unlikely to have survived without intervention (Figure 4.8-12) (Stacy 2012). Therefore, the Trustees concluded that any turtle that became heavily oiled but was not rescued would have died.

Source: *B. Witherington (top left), B. Stacy (top right, bottom).*

Figure 4.8-12. Photographs showing the debilitating effects of physical fouling of oil on sea turtles. Top left: thick, viscous oil on the surface made detection and capture of a small juvenile sea turtles difficult; top right: this heavily oiled, small juvenile sea turtle would not have survived without rescue and rehabilitation because the heavy, viscous oil impeded its movement and its ability to feed and escape predators; bottom right: heavily oiled turtles were at risk of aspirating oil as shown in this turtle found stranded in Louisiana. A clump of brown oil is present within the trachea (windpipe); bottom left: when at the surface to breathe, rest, or feed, sea turtles were exposed to lethally hot temperatures with dark oil present.

Miring in oil impeded movement and diving ability, risking physical exertion and dehydration aggravated by hyperthermia (overheating) from contact with thick, dark, hot oil under summer conditions; these effects are fatal if unabated (Figure 4.8-12). The Trustees documented these in both clinical findings in live oiled turtles and postmortem observations in dead turtles. Upon admission to rehabilitation facilities, blood abnormalities in oiled turtles included nonspecific metabolic and physiological abnormalities attributable to stress, dehydration, and exertion caused by oiling, capture, and transport (Stacy 2012). As expected, the numbers of recovered oiled, dead turtles found offshore or as strandings were low due to a number of factors (see Section 4.8.3.3, Sea Turtles Were Observed in Oil Throughout the Northern Gulf of Mexico). Those that were found and examined provided additional evidence that conditions resulting in heavy oiling were fatal. Asphyxiation by oil and ingestion of large quantities of oil were observed in these turtles (Stacy 2012). Based on these observations, the Trustees concluded that both external and internal exposure to oil were severe for small oceanic juveniles due to the dependence of these animals on surface habitats where oil accumulated (Stacy 2012; Wallace et al. 2015), and that the probability of death increased with degree of oiling (see Section 4.8.4.4, Mortality Estimates for Turtles Based on Degree of Oiling) (Mitchelmore et al. 2015). Specifically, heavily oiled, small juvenile turtles were expected to die without intervention (Stacy 2012; Wallace et al. 2015). The Trustees also concluded that similar concerns about miring in surface oil were warranted for larger turtles exposed to surface oil based on limited observations of impaired, oiled, larger turtles during the DWH spill and previous reports of oiling associated with death or stranding (Camacho et al. 2013; Shigenaka 2003).

4.8.4.2 Toxic Effects of Oil on Turtles and Their Habitats

Relatively few studies in the scientific literature have described adverse physiological toxic effects of oil on sea turtles in detail (see Shigenaka 2003 for review). Most readily apparent observations of turtles affected by oil have been attributed to physical effects resulting from miring or obstruction of the mouth or digestive system. A laboratory study that examined physiological and health effects of oil on sea turtles found oil in turtles' nares (i.e., external openings of turtles' nasal cavities), mouths, around their eyes, and in feces, indicating that turtles were ingesting oil in addition to coming into direct contact with it. This same study also associated exposure with inflammation and sloughing of skin, decreased red blood cell counts, and salt gland dysfunction (Lutcavage et al. 1995). A recent report from the Canary Islands implicated oiling as the principal cause of the stranding of small and large juveniles, and possibly adults, between 1998 and 2011 (Camacho et al. 2013). However, these animals were not oiled during discrete spills; they are described in the context of general regional oil pollution.

DWH oil caused toxic effects on many species across taxonomic groups (Section 4.3, Toxicity). To examine potential toxicity to sea turtles, toxicologists, veterinarians, and sea turtle biologists synthesized Natural Resource Damage Assessment (NRDA) and non-NRDA data on toxic effects of petroleum products on vertebrates and considered results of blood and tissue analyses performed on sea turtles recovered during the DWH oil spill (see Section 4.8.3.3, Sea Turtles Were Observed in Oil Throughout the Northern Gulf of Mexico) and from surrogate turtle species that were subjects of a controlled laboratory experiment to investigate toxic effects of DWH oil ingestion administered orally (Section 4.8.4.2.2, Laboratory Oil Toxicity Study of Surrogate Turtle Species).

4.8.4.2.1 Observations Related to Oil Toxicity Effects on Rescued and Stranded Turtles

As presented in Section 4.8.3.3.1 (Directed Captures and Observations of Oiling in Offshore Areas), the high frequency of oil ingestion observed in oiled turtles and detection of oil-related compounds in tissues and bile demonstrated internal exposure in sea turtles oiled during the DWH spill (Stacy 2012; Stacy & Innis 2012; Ylitalo et al. 2014). Evidence of damage to red blood cells and impaired salt gland function, as previously reported in loggerheads exposed to crude oil (Lutcavage et al. 1995), was not found (Stacy 2012). Also, histological evidence of organ injury was not observed in the small number of dead oiled sea turtles that were in suitable condition for detailed examination (Stacy 2012).

Cutaneous and oral exposure was inevitably associated with some degree of inhalation exposure as well, as demonstrated by increased PAH concentrations in lung tissues of oiled turtles compared to lung tissues of non-visibly oiled turtles (Ylitalo et al. 2014). Turtles that surfaced to breathe within oil would have inhaled and/or aspirated oil and oil compounds, as well as smoke from burning oil, similar to the inhalation exposure that marine mammals experienced (see Section 4.9, Marine Mammals). Sea turtles would have been exposed to toxic concentrations of oil at the sea surface where sea turtles surfaced to breathe, particularly in offshore areas near the wellhead where small juvenile turtles live and where volatiles were more likely to occur (see Section 4.2, Natural Resource Exposure). Because turtles must hold their breath while swimming, feeding, and diving, oil compounds inhaled into their lungs would be assimilated into their bodies as they use oxygen in lungs, blood, and muscles to fuel underwater activities. Average turtle dive durations last 5–30 minutes, and longer dives can last 30–45 minutes, as in the case of loggerheads, or over an hour for leatherbacks (Hochscheid 2014; Southwood 2013). In addition to duration, increased pressure with water depth may enhance systemic absorption during prolonged dives (Hochscheid 2014). Concentrated exposure caused by dive depth and duration, particularly for larger turtles that actively dive throughout the water column, could affect respiratory and cardiovascular function, and thus hinder turtles' ability to dive efficiently to forage, escape predators, find mates, migrate, etc. (see Section 4.9, Marine Mammals, for discussion of inhalation effects). Although tissue analyses showed elevated PAH concentrations in lungs of oiled turtles (Ylitalo et al. 2014), determination of adverse effects of inhalation or aspiration of oil products on sea turtles was not possible based on available information.

It is important to note that the dose of oil and duration of exposure in sea turtles that were naturally exposed during the DWH spill were unknown, which complicated comparison of blood values and other data with previously available studies of petroleum toxicity in various taxa.

4.8.4.2.2 Laboratory Oil Toxicity Study of Surrogate Turtle Species

In response to the number of uncertain variables associated with the nature, duration, and extent of exposures that sea turtles experienced in oiled habitats during the DWH oil spill, and the paucity of published data on petroleum toxicity to sea turtles (Shigenaka 2003), the Trustees conducted a laboratory study to evaluate the acute toxicological effects of ingested oil on common species of freshwater turtles using known time and dose exposures. The endangered status of sea turtles necessitated the use of the freshwater surrogate species: red-eared sliders (*Trachemys scripta*) and common snapping turtles (*Chelydra serpentina*) (Mitchelmore & Rowe 2015).

Dose-dependent increases in the levels of biliary PAH metabolites demonstrated uptake and metabolism of oil at levels similar to those of sea turtles oiled during the DWH spill (Mitchelmore & Rowe 2015; Ylitalo et al. 2014). The Trustees observed physiological abnormalities, including evidence of dehydration and decreased digestive function and assimilation of nutrients, and some measures of oxidative stress and DNA damage also showed changes that were consistent with PAH exposure. In contrast to some other studies of petroleum toxicity in other vertebrates, mortality and damage to red blood cells were not observed. Two red-eared sliders that had received a high dose of oil showed dysfunction of the hypothalamic-pituitary-adrenal (HPA) axis, which regulates stress response and other vital functions. When all individuals were compared by oil dose treatment, turtles that had received a consistent oil dose showed a dampened HPA response; however, this dysfunction was not statistically significant due to the high variation among individuals ((Mitchelmore & Rowe 2015). Therefore, the effects of experimental oil exposure on red-eared sliders and common snapping turtles did not clearly indicate HPA dysfunction as reported in minks (Mohr et al. 2008), marine mammals (Schwacke et al. 2014), and marine iguanas (Wikelski et al. 2002).

Overall, turtles orally exposed to DWH oil during the surrogate study did not show severe, life-threatening physiological derangements or mortality. The result was consistent with other observations that effects from physical fouling are the most readily apparent consequence of oil exposure in sea turtles. However, surrogate turtles were not dosed for time periods comparable to other studies where such effects were shown in orally dosed vertebrates (e.g., Mohr et al. 2008). Additional factors that are relevant to exposure during oil spills, such as delayed and longer-term effects of these low-dose oil exposures, as well as exposure via multiple routes, also were not part of the study design due to logistical constraints. Taken together, these experimental constraints limited the application of the surrogate study in estimating mortality of wild sea turtles affected by the DWH oil spill (Mitchelmore & Rowe 2015).

4.8.4.3 Potential Adverse Effects from Loss of Prey/Habitat

The Trustees documented evidence of potential impacts to sea turtle habitats. Exposure of *Sargassum* to oil can cause it to sink to the ocean floor, thus removing essential habitat for oceanic juvenile sea turtles and numerous other organisms (see Section 4.4, Water Column) (Powers et al. 2013). Approximately 4,000 to 7,000 square kilometers of *Sargassum* was oiled in relation to the DWH oil spill and determined to have been lost to the northern Gulf of Mexico ecosystem (see Section 4.4, Water Column). This loss was significant for small juvenile sea turtles that were already exposed to DWH oil via multiple pathways because it reduced the availability of already patchy and ephemeral refuge areas, thereby likely increasing transit time and energy costs to turtles between available habitats, as well as making turtles more vulnerable to predation (Witherington et al. 2012).

In continental shelf areas closer to shore, post-mortem evaluation of fat content under the carapaces (i.e., top half of the shell) of juvenile Kemp's ridley turtles that stranded dead revealed that body conditions and available fat stores declined after 2010 (Stacy 2015). This observation suggests that turtles have undergone a general decline in nutritional condition, potentially due to their reduced prey availability or quality, reduced ability to find food, or some unknown health effect. In a separate study, changes in chemical markers in carapacial scutes (i.e., the keratinized covering of turtles' shells) of nesting adult Kemp's ridleys suggested that turtles in 2011 and 2012 foraged in different locations than areas used by turtles in 2010 prior to the DWH spill (Hooper & Schmitt 2015). Although the cause(s) of these observations is unknown at this time, a persistent effect on turtle foraging areas and/or prey availability or quality related to the DWH oil spill cannot be ruled out. Furthermore, because sea turtles tend to use the same foraging areas across years (e.g., Shaver et al. 2013), it is plausible that turtles that foraged in or traveled through the DWH oil spill footprint were exposed to oil.

4.8.4.4 Mortality Estimates for Turtles Based on Degree of Oiling

Mortality estimates were developed to quantify total numbers of injured turtles based on relative degrees of oil exposure. Mortality estimates considered three key elements: 1) veterinary assessment of live and dead oiled turtles; 2) toxicological assessment of affects from oil exposure; and 3) the potential for continued, progressive exposure and oiling. Mortality estimates were developed based on degree of oil exposure and for different life stages of sea turtles with consideration of relative risk of exposure. The high survival rate of oiled turtles that were treated in rehabilitation facilities was not considered reflective of the outcome of oiling without medical intervention (Figure 4.8-12) (Stacy & Innis 2012). The relatively rapid recovery of many of these animals was consistent with the available evidence indicating that physical fouling was one of the more significant immediate consequences of oiling, and that removal of this threat resulted in a positive survival outcome. Therefore, heavy oiling without medical intervention—which would have been the case for the vast majority of turtles exposed to DWH oil—was considered a primary factor with regard to probability of mortality.

4.8.4.4.1 Mortality Estimates for Heavily Oiled Turtles

Veterinarians with specific sea turtle expertise, toxicologists, and sea turtle biologists concluded that physical effects of miring in heavy oil—as seen in heavily oiled turtles—would likely have been lethal (i.e., 100 percent mortality probability for small juveniles). This probability was based on several factors, as described in Section 4.8.4.1 (Physical Effects) and summarized as follows:

- Small juvenile turtles live, breathe, eat, and seek refuge from predators within the top 2 meters of the water column. Additionally, they are incapable of deep diving and are not powerful swimmers (Witherington et al. 2012). Therefore, the physical miring that the Trustees observed on heavily oiled turtles would have prevented them from escaping or shedding the thick, sticky, heavy oil (Stacy 2012). This heavily oiled condition would have caused death either acutely or chronically.

- Small juvenile turtles that were heavily oiled were lethargic and palpably warm when rescuers pulled them from the ocean during response efforts, and surface oil temperatures were in excess of 120°F (approximately 50°C), which is well above the lethal threshold for sea turtles (Drake & Spotila 2002; Jessop et al. 2000).

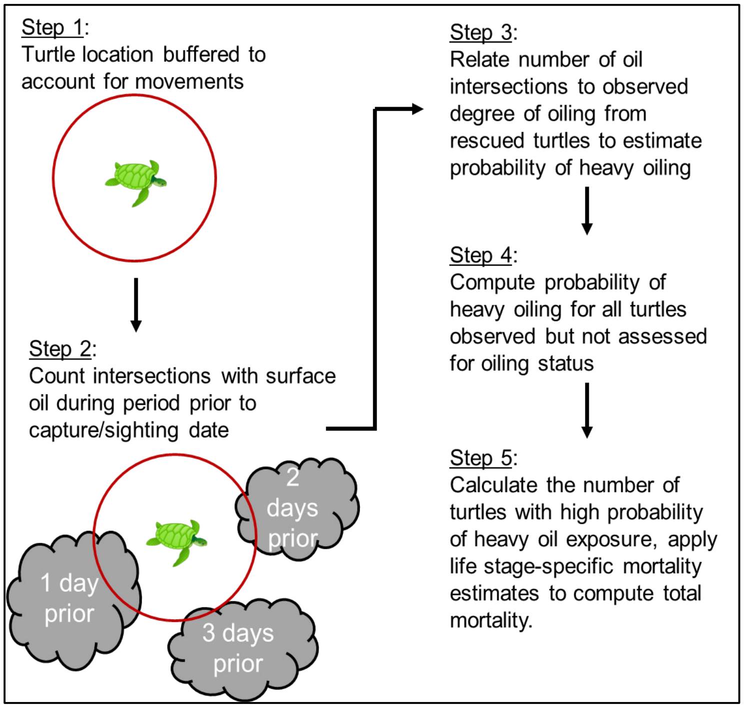
- Dead and live-stranded, debilitated, heavily oiled turtles that would have died without medical intervention provided direct evidence of the lethality of this condition (Stacy 2012; Wallace et al. 2015).

This assessment provided a basis for evaluating mortality estimates for other oiling categories as a function of the probability that they would have become heavily oiled.

Probability of Heavy Oil Exposure Based on Spatial and Temporal Proximity to Oil

Field observations and veterinary assessments of small juvenile turtles demonstrated that physical miring in surface oil and oiled surface habitats caused significant harm to sea turtles. Therefore, the Trustees concluded that surface oil conditions in locations and at times when heavily oiled turtles were observed were generally indicative of heavy oil exposure. Therefore, biologists and statisticians developed a model to estimate the probability of turtles being heavily oiled based on remotely sensed surface oil data in the area around and time preceding a turtle's capture or sighting (Wallace et al. 2015) (Figure 4.8-13).

To calculate these probabilities, each turtle location first was placed within a circle (i.e., a spatially buffered location) to account for the area within which small juvenile turtles presumably moved (Putman & Mansfield 2015)—either while actively swimming or passively drifting in surface currents—during the period prior to being found by searchers. Second, the cumulative number of intersections between a turtle's buffered location and all daily surface oil footprints within 3 weeks prior to its capture or sighting date was calculated for all turtles. Third, statistical relationships were determined between the surface oil environment in the area and time prior to turtle captures and the observed degree of oiling of those rescued turtles. Fourth, the Trustees then applied this relationship to all turtles that were observed, but not directly assessed, by using surface oil conditions associated with where and when turtles were observed to predict their estimated degrees of oil exposure (Wallace et al. 2015).



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

Figure 4.8-13. Schematic of the modeling approach to estimate probability of heavy oil exposure based on the relationship between surface oil in the area and time prior to turtle captures and their observed degree of oiling. Description of steps appears in the text.

In this way, the number of intersections with surface oil—i.e., proximity in time and space—provided a measure of persistence of surface oil, and thus a relative probability of oil exposure, for a turtle in a certain area at a certain time. This approach has the important advantage of combining ubiquitous, reliable data—i.e., satellite-derived measurements of surface oil and empirical observations of the extent of oiling on sea turtles—to statistically estimate the probability of heavy oil exposure in cases where the former data type is available, but not the latter (Wallace et al. 2015). This approach was used in injury quantification (Section 4.8.5, Injury Quantification) to estimate probability of heavy oiling for turtles and areas within the cumulative DWH footprint.

In the final step, the Trustees computed the numbers of turtles with high probabilities of being heavily oiled for small juveniles captured or sighted during rescue operations (Section 4.8.3.3.1, Directed Captures and Observations of Oiling in Offshore Areas), as well as for large juveniles and adults observed within the DWH oil spill footprint during aerial surveys (Section 4.8.3.3.2, Observations of Turtles on

Continental Shelf and on Beaches). These probabilities of heavy oil exposure were used to estimate the total numbers of turtles that were heavily oiled as a result of the DWH oil spill (see Section 4.8.5, Injury Quantification), and therefore died.

It is worth noting a few important assumptions of this approach that influenced the probability of turtles being heavily oiled. First, the surface oil data used in the model did not include any information about relative thickness of oil in time and space. Therefore, the correlation between proximity of surface oil and the degree of oiling observed on turtles was based solely on the presence of oil on the surface of the ocean in a given place and time. Second, the model counted the number, not the extent or degree, of intersections between a turtle location and surface oil. Third, because the model was fitted to the relationship between surface oil and observed degree of oiling of small juveniles, it also assumed that, when estimating probabilities of heavy oiling for larger turtles, the processes by which heavy oil exposure occurred were comparable for larger turtles and smaller turtles.

However, the Trustees concluded that these were reasonable assumptions because 1) surface oil and observed degree of oiling showed a significant positive relationship, which indicated that surface oil represented other factors—such as oil thickness—that likely influence turtle oiling (Wallace et al. 2015), and 2) all turtles—regardless of size—must spend significant time at the surface, and while at the surface in areas that also had oil consistently present, turtles would have been exposed.

4.8.4.4.2 Mortality Estimates for Turtles That Were Exposed, But Not Heavily Oiled

The majority of turtles documented were not heavily oiled but were assigned to oiling categories that were less severe, either by veterinary assessment of external oiling or by the model described above. Nonetheless, these turtles were exposed to oil externally and internally (Stacy 2012) (Section 4.8.3.3.1, Directed Captures and Observations of Oiling in Offshore Areas). To estimate the risk of mortality for this group, veterinarians examined various clinical, hematological, and blood chemistry endpoints that have predictive value in terms of survival outcome (Stacy & Innis 2012). Similar prognostic approaches are used in humans and other animals, including sea turtles (Knaus et al. 1985; Koterba & House 1996; Stacy et al. 2013). A series of prognostic scoring models using this approach yielded relatively similar results, showing that a substantial proportion of oiled turtles had clinically significant physiological abnormalities with predicted mortality rates between 6 percent and 22 percent (mean of 14 percent) without treatment based on physiological effects alone (Stacy et al. 2013). This predicted outcome did not include risk of mortality from ongoing exposure to oil and toxicity, which was likely if those turtles had not been rescued from their oiled environment (Stacy et al. 2013).

Toxicologists considered potential toxic effects of oil exposure in oiling categories that were less than heavily oiled (Mitchelmore et al. 2015). To develop ranges of mortality estimates for the turtles in lower oiling categories, toxicologists considered the estimated levels of ingested oil in wild juvenile sea turtles, together with scientific literature on the adverse effects of oil exposure to other vertebrate species (e.g., Mohr et al. 2008; Schwacke et al. 2014) and results of the laboratory toxicology study of freshwater turtles (see Section 4.8.4.2.2, Laboratory Oil Toxicity Study of Surrogate Turtle Species) (Mitchelmore & Rowe 2015). Other factors that were considered included persistent or sub-lethal effects from physical impairment (e.g., reduced fitness and implications on foraging and predator avoidance) and potential chronic toxicological effects resulting from alimentary, dermal, and inhalational exposure to oil that have been demonstrated in well-studied vertebrate taxa (Mitchelmore et al. 2015). The potential impacts of chemical dispersant applications in the DWH spill zone were considered as an additional, but unknown, toxicological risk for sea turtles. Mortality estimates were developed for each non-heavy oiling category, and then combined into a single estimate for all turtles assigned or estimated to belong to non-heavy oiling categories.

4.8.4.5 Summary of Mortality Estimates

Table 4.8-3 presents a summary of mortality estimates for sea turtles by life stage and by degree of oiling. As described above, heavily oiled, small juvenile turtles were assigned a mortality probability of 100 percent.

Table 4.8-3. Summary of mortality estimates for sea turtles by life stage and by degree of oiling (Mitchelmore et al. 2015). Estimate for less than heavily oiled small juveniles is a weighted average of mortality estimates for turtles in all non-heavily oiled categories, including those that were not visibly oiled.

	Mortality Estimates	
Sea Turtle Life Stage	**Heavily Oiled**	**Less Than Heavily Oiled**
Small juveniles	100%	30%
Large juveniles and adults	10–20%	5%

Small juvenile turtles that were exposed at levels less than heavily oiled were assigned a mortality probability of 30 percent (Table 4.8-3). This value was derived from an assessment by a panel of three toxicologists who used information including the proportion of captured turtles in different oiling categories with oil present in their oral cavities (Table 4.8-2), as well as estimations of the amount of oil in the oral cavities of those turtles (Mitchelmore et al. 2015). These observations were converted to estimated oral doses of oil for turtles in the different oiling categories. The dosing estimations were then used to assess the likelihood of significant injury, primarily to the HPA axis (Mohr et al. 2008; Schwacke et al. 2014), that could result in mortality, depending on the degree of external oiling and without veterinary intervention. Also considered within this estimate is the mortality range predicted based on the degree of physiological alteration observed in oiled turtles upon admission to rehabilitation facilities (Stacy 2012).

More specifically, the toxicology panel's consensus was that minimally, lightly, and moderately oiled turtles would have experienced mortality rates of 25 percent, 50 percent, and 85 percent, respectively (Mitchelmore et al. 2015). These percentages were derived from comparisons of the estimated oral doses of oil in wild turtles in different oiling categories to the oral doses of oil that caused severe disruption of the HPA axis in laboratory studies of other vertebrates (primarily mink) (Mohr et al. 2008), and an assumption that such disruption would likely result in mortality in small juvenile sea turtles. Using the numbers of turtles in each of these categories as reported in Table 4.8-2, and assuming that no non-visibly oiled turtles died due to oil exposure, the toxicologists calculated the overall proportion of estimated dead turtles relative to total turtles observed across all non-heavily oiled categories. Following this procedure, the toxicologists estimated that the overall proportion of dead turtles relative to total turtles observed across all non-heavily oiled categories would have been 30 percent. This estimate reflected common areas of concern among experts, similarity of estimates produced by different approaches under this assessment, and the findings from the surrogate turtle toxicity study (Mitchelmore & Rowe 2015).

Nonetheless, this mortality estimate for non-heavily oiled turtles represents the high end of reasonable values, considering that 1) the surrogate turtle toxicity study did not show severe, life-threatening physiological derangements, including HPA dysfunction, or mortality from the levels of oil exposure observed in field-collected turtles (Mitchelmore & Rowe 2015), and 2) oiled turtles that were rescued did not show consistent oil-induced adverse effects upon arrival to rehabilitation centers that could be clearly separated from confounding effects of prolonged transport handling stress (Stacy & Innis 2012).

To assign mortality estimates to larger juvenile and adult turtles in continental shelf areas, the Trustees concluded that the same processes that could lead to an oceanic turtle becoming heavily oiled would also apply to neritic turtles, and that the methods by which risk of oiling was estimated for oceanic turtles—relationships between surface oil data and direct observations of oiled animals—were also applicable to neritic turtles. Simply stated, probability of oiling was based on spatio-temporal distribution of oil relative to where turtles were either captured or sighted (Wallace et al. 2015). However, the Trustees accounted for behavioral differences between small and large turtles in terms of relative likelihood of exposure to surface oil. Although subsurface exposure was certainly a concern as well, observational evidence of turtle exposure to DWH oil was predominantly related to oil at or near the surface. Small turtles spend more than 80 percent of their time at the surface, whereas large turtles are at the surface roughly 10 percent of the time (Bolten 2003; Garrison 2015; Southwood 2013).

Therefore, mortality estimates applied to small juvenile turtles were scaled lower for larger turtles to reflect this proportional difference in probability of surface oil exposure and resulting mortality (Wallace et al. 2015). Large turtles that were exposed to surface oil conditions similar to those associated with heavy oiling of juvenile turtles were assigned to a high oil exposure category with an associated mortality probability of 10–20 percent. Larger turtles in the lower oil exposure category (i.e., turtles that experienced oil conditions similar to small turtles that had been assigned to intermediate oiling categories) had a mortality probability of 5 percent (Table 4.8-3) (Wallace et al. 2015).

These mortality estimates might underestimate the adverse effects of surface oil exposure on large juvenile and adult sea turtles. Reports of stranded, oiled, larger turtles indicate that effects can be significant and multi-systemic (Camacho et al. 2013; Shigenaka 2003). Notably, the only heavily oiled neritic juvenile encountered during DWH directed capture operations was found floating with the lateral edge of its carapace and a front flipper above the surface, a condition that did not bode well for survival without intervention (Witherington 2015 personal communication). Furthermore, it is possible that these neritic turtles would also be exposed to oil via numerous exposure routes (e.g., prey ingestion, inhalation, etc.), but also to sub-surface oil and oil/dispersant mixtures, which was an exposure scenario not considered for the oceanic turtles.

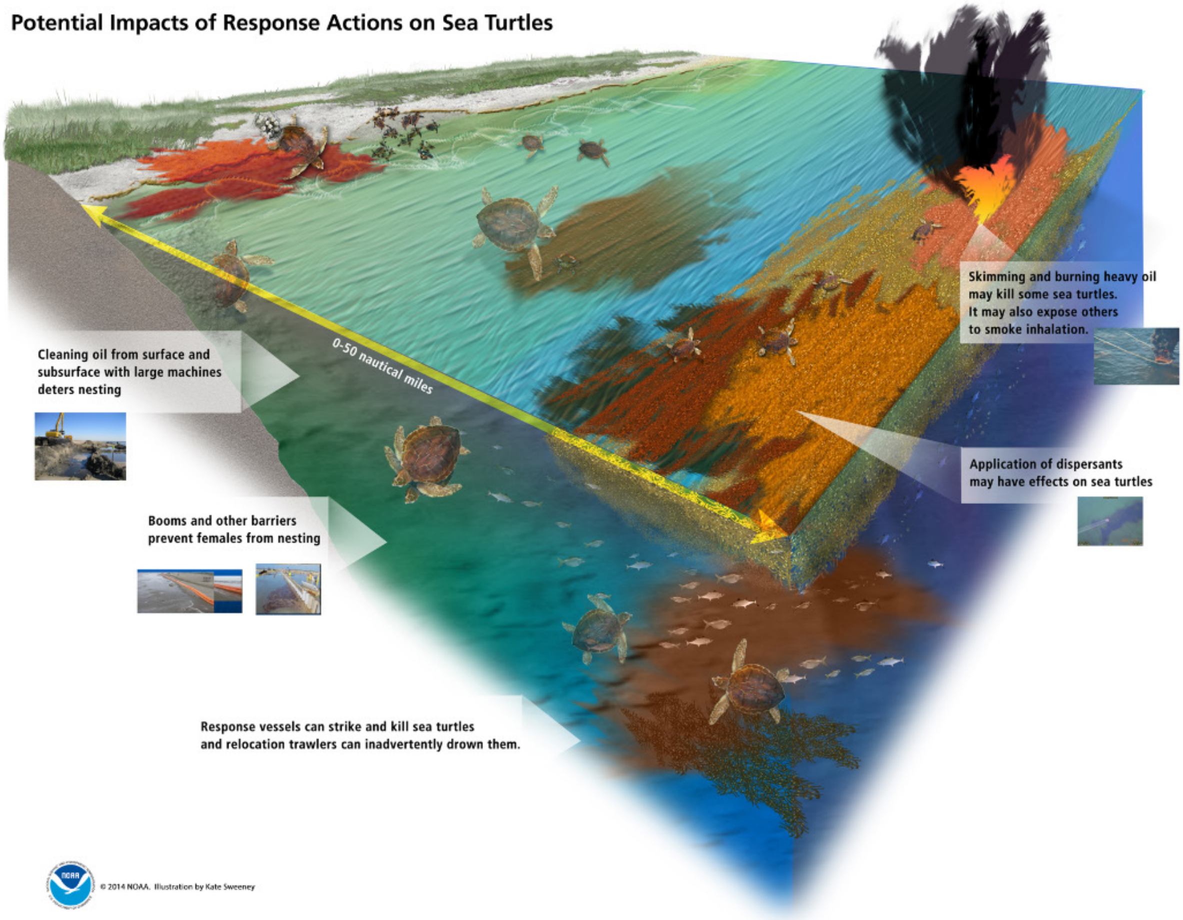
On the other hand, the Trustees did not have direct observations of how and to what extent exposure to oil at the surface affected heavily oiled large juveniles and adults (except for the single individual described above). It is possible that large juvenile and adult turtles were less susceptible to adverse effects of heavy oil exposure or to becoming heavily oiled in the first place. For example, in addition to the reduced time that these turtles spend at the surface compared to small juvenile turtles, they are larger in size and stronger swimmers, and therefore were likely more capable of moving away from contaminated areas and potentially avoiding severe oil exposures. These factors might have made large juvenile and adult turtles less susceptible to oil exposure and injury. The Trustees took all of these factors into account and concluded that the values shown in Table 4.8-3 were reasonable given the strength of evidence about adverse effects related to exposure to surface oil.

4.8.4.6 Injuries to Turtles Caused by Response Activities

The Trustees determined that turtles were injured by response activities that occurred in marine and terrestrial areas (Figure 4.8-14). In marine areas, response activities that injured turtles included relocation trawling, dredging, and response vessel traffic in nearshore areas where turtles were abundant during the spring and summer (Garrison 2015). Injuries resulting from vessel strikes were one of the most common traumatic injuries observed in stranded turtles within the response area (Stacy 2012, 2015; Stacy & Schroeder 2014). In addition, turtles were likely killed during other response activities such as oil skimming and burning operations, which were primarily conducted in more heavily oiled areas and are an additional justification for the high mortality assigned to sea turtles within these areas.

Over 320 kilometers of sand beach coastline that support sea turtle nesting were oiled by the DWH spill or affected by response activities (Michel et al. 2015). Turtles were injured in terrestrial areas by response activities, including beach cleanup operations and associated increased human presence, increased lighting at night on and near nesting beaches, and translocation of nests in the northern Gulf of Mexico to avoid direct contamination of hatchlings (Michel et al. 2015; Provancha & Mukherjee 2011). The DWH oil spill occurred at the onset of the nesting season for sea turtles in the northern Gulf. To prevent hatchlings from emerging from northern Gulf nests and entering oil-filled northern Gulf waters, and to avoid their risk of being killed by beach response activities, sea turtle nests were excavated and eggs were translocated to the Atlantic coast of Florida. The Trustees determined that hatchlings that emerged from these translocated eggs were injured because it is unknown whether those individuals will return to the Gulf of Mexico and fulfill their role in the Gulf ecosystem (Provancha & Mukherjee 2011).

Additionally, the Trustees determined that fewer loggerhead nests were observed in 2010 in the Florida Panhandle than were expected when compared to previous and subsequent years at those beaches and others outside of the impacted area (see Section 4.8.5.2, Quantification of Sea Turtle Injuries Caused by DWH Response Activities). This decline in loggerhead nesting was attributed to disturbance to nesting female turtles related to response activities on beaches (Cacela & Dixon 2013; Michel et al. 2015).

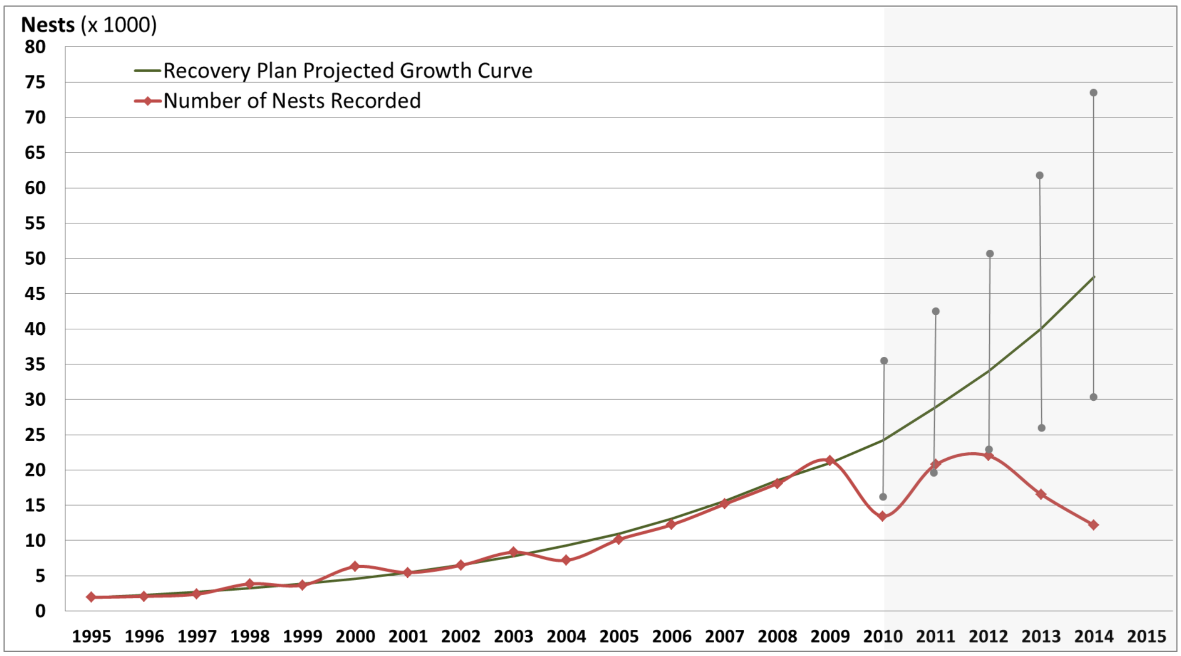


***Source**: Kate Sweeney for NOAA.*

Figure 4.8-14. Potential injuries caused by DWH response activities. Text boxes describe specific activities and potential impacts on sea turtles throughout the northern Gulf of Mexico.

4.8.4.7 Reduced Kemp's Ridley Nesting Abundance and Hatchling Production

Kemp's ridley turtles are listed as "Endangered" under the ESA and have the most restricted nesting distribution of all of the sea turtle species (Wallace et al. 2010). Nesting occurs in the Gulf of Mexico from Veracruz, Mexico, north along the Texas coast to Bolivar Peninsula, but more than 90 percent of annual nesting occurs near Rancho Nuevo in Tamaulipas, Mexico (NMFS et al. 2011). After the population declined from tens of thousands of nesting females in the 1960s to only a few hundred in the 1980s, nest numbers began increasing following intensive conservation efforts. Between 1996 and 2009, numbers of nests increased at approximately 18 percent per year (Figure 4.8-15) (Crowder & Heppell 2011; Gallaway et al. 2013; NMFS et al. 2011). However, between 2010 and 2014, annual numbers of nests were estimated to be between 8,000 and 35,000 nests lower than expected based on the population trajectory prior to 2010 (Dixon & Heppell 2015). As a result, the Trustees initiated an evaluation of the observed nesting decline and its relationship to the DWH spill.



***Source**: Chip Wood.*

Figure 4.8-15. Kemp's ridley annual nest abundance increased exponentially between 1996 and 2009, but has not reached projected abundance since 2010 (Dixon & Heppell 2015). The green line shows the estimated nesting trend from 1996 to 2009 continuing to the present, whereas the red points and line show observed nest abundance for the same time period. Vertical grey bars are 95 percent credible intervals around projected nest numbers. Between 1996 and 2009, annual Kemp's ridley nest abundance increased approximately 18 percent per year due to conservation measures that protected turtles from fisheries bycatch and from human consumption of eggs and nesting turtles. However, starting in 2010, nest abundance failed to reach projections based on the previous increasing trends, and it has remained below projected levels in recent years. Although DWH oil was unlikely to have had an impact on Kemp's ridley nesting abundance in 2010, it is likely DWH oil contributed to some unquantified extent to the observed reduction in projected nesting after 2010.

DWH oil did not arrive on the continental shelf of the northern Gulf of Mexico until late May or early June 2010. By that time, adult Kemp's ridley turtles that were going to breed in 2010 would likely have already departed the northern Gulf for their breeding and nesting areas in the western Gulf. Therefore, DWH oil was unlikely to have had a direct impact on Kemp's ridley nesting in 2010. However, DWH oil could have contributed to the reduced numbers of nests in subsequent years (2011–2014) through direct and indirect pathways. For example, adult Kemp's ridley turtles that were not breeding in 2010, as well as subadults that would have recruited to the breeding population in 2011–2014, were among the Kemp's ridley turtles present on the continental shelf in 2010 and potentially killed by DWH oil exposure (see Section 4.8.5.1.2, Exposure and Injury Quantification of Large Juvenile and Adult Sea Turtles in Continental Shelf Areas) (Garrison 2015). The loss of these animals would have affected the overall Kemp's ridley turtle nesting trajectory in subsequent years.

In addition, DWH oil was present in areas that have been identified as vital Kemp's ridley turtle migration and foraging areas (Shaver et al. 2013). Some indirect measures—chemical markers in nesting Kemp's ridley turtle tissues and changes in habitat use (Hooper & Schmitt 2015)—suggest possible behavioral responses to altered foraging area quality, perhaps related to DWH oil. Therefore, it is possible that DWH effects—in addition to other anthropogenic factors and environmental conditions—contributed to the observed reduction in Kemp's ridley turtle nesting and associated hatchling production after 2010 (Crowder & Heppell 2011; Gallaway et al. 2013). The actual nature and magnitude of a DWH effect on Kemp's ridley turtle nesting abundance requires further evaluation.

4.8.4.8 Summary of Injury Determination

As described in this section, sea turtles were injured by adverse physical effects (e.g., decreased locomotion, decreased ability to feed and evade predators, and hyperthermia) related to miring in oil and exposure to harmful conditions in oiled surface habitat. Contributing to this injury are less understood, but nonetheless concerning, toxicological effects of exposure to oil, dispersants, and oil-dispersant mixtures. These effects are based on a significant body of literature on toxicity in other taxa and involve basic biological and physiological mechanisms present in sea turtles as well. The Trustees identified several additional concerns related to persistent or chronic effects, including those associated with subsurface exposures and unobserved effects on prey or habitat. These factors were insufficiently understood to include in a quantitative assessment.

Mortality probabilities for sea turtles that were subject to different degrees of oil exposure were developed based on relative risk of becoming heavily oiled within the spill footprint and based on a comprehensive assessment of NRDA and non-NRDA data sources relevant to veterinary and toxicological assessment (Table 4.8-3). These mortality estimates were then used in injury quantification to calculate the total numbers of turtles that were killed by DWH oil exposure (see Section 4.8.5, Injury Quantification). In addition, the Trustees determined that sea turtles were injured by response activities in their marine and terrestrial habitats. In the following section, we synthesize exposure and injury determination with estimates of the numbers of turtles exposed to DWH oil to quantify sea turtle injuries caused by the DWH oil spill.

4.8.5 Injury Quantification

Key Points

- The Trustees quantified sea turtle injuries across life stages and habitats, synthesizing primary information from field-based observations and veterinary and toxicology assessments.
- The Trustees estimated that between 4,900 and 7,600 large juvenile and adult sea turtles and between 55,000 and as many as 160,000 small juvenile sea turtles were killed by exposure to DWH oil. The Trustees also estimated that nearly 35,000 hatchling sea turtles were injured by response activities associated with the DWH oil spill.
- Due to data limitations and logistical constraints, the Trustees could not quantify the following injuries:
    - Injury to leatherbacks caused by the DWH spill.
    - Injury due to response activities, including impacts from oil skimming and burning operations and collisions with response watercraft.
    - Loss of hatchlings resulting from nests missed during translocation to the Atlantic coast.
    - Reduced Kemp's ridley nesting abundance and associated hatchling production.
- Despite some uncertainties about information and assumptions used in the injury quantification, the Trustees concluded that the assessment adequately quantified the nature and magnitude of the injuries to sea turtles caused by the DWH oil spill.

Previous sections described evidence of sea turtles exposed to oil (Section 4.8.3) and the severity and probable outcomes of those exposures in marine and terrestrial areas (Section 4.8.4). This section describes how the Trustees used these observations and determinations to estimate the total numbers of turtles exposed to and injured by DWH oil.

4.8.5.1 Quantification of Turtle Abundance, Exposures, and Injuries Across the DWH Spill Footprint and Time Period

Boat-based, land-based, and aerial surveys provided valuable information about where turtles were distributed in relation to oil and the nature and degree of oil exposures. However, these observations are underestimates of the actual number of turtles that were present and potentially exposed to oil because surveys only sampled small fractions of the overall area within which turtles were present and the time period during which turtles might have been exposed (Section 4.8.3.3, Sea Turtles Were Observed in Oil Throughout the Northern Gulf of Mexico). Therefore, the Trustees applied statistical techniques to empirical data collected during assessment activities to quantify the actual numbers of exposed turtles during the DWH oil spill.

The Trustees extrapolated total sea turtle abundance in the DWH oil spill footprint based on the direct observations of turtles using the following general steps (see Figure 4.8-16 for a schematic representation).

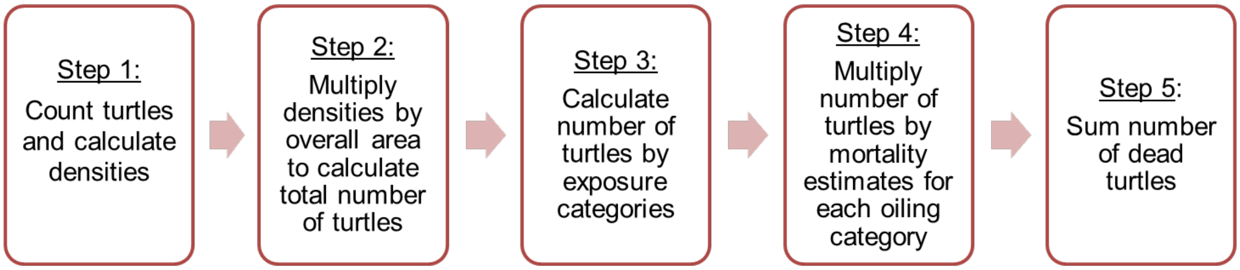


Figure 4.8-16. Schematic illustrating the Trustees' process for quantifying total injuries to sea turtles based on the sample of turtles observed within the footprint and time period of the DWH oil spill. Numbered steps correspond to steps described in the text.

1. Statisticians used the rescue and sightings data to estimate turtle densities within areas that were searched by boat-based rescue operations (McDonald et al. 2015) or by aerial surveys (Garrison 2015).

2. Turtle densities were expanded to areas within the DWH oil footprint that were not directly searched based on environmental similarities between searched and unsearched areas. This assumes that environmental conditions of searched areas are associated with observed densities, and thus observed densities could be extrapolated to environmentally similar unsearched areas (see Garrison 2015 for details).

3. The Trustees categorized turtles by observed or estimated degrees of oiling. Observed degrees of oiling were based on veterinary assessments of rescued turtles, as described in Section 4.8.3.3.1 (Directed Captures and Observations of Oiling in Offshore Areas). Estimated degrees of oiling were derived from the modeling approach described in Section 4.8.4.4.1 (Mortality Estimates for Heavily Oiled Turtles) based on an established relationship between observed turtle oiling categories and proximity to surface oil in areas where turtles were documented during the oil spill (Wallace et al. 2015).

4. The total number of turtles in each oiling category was multiplied by mortality estimates for each oiling category.

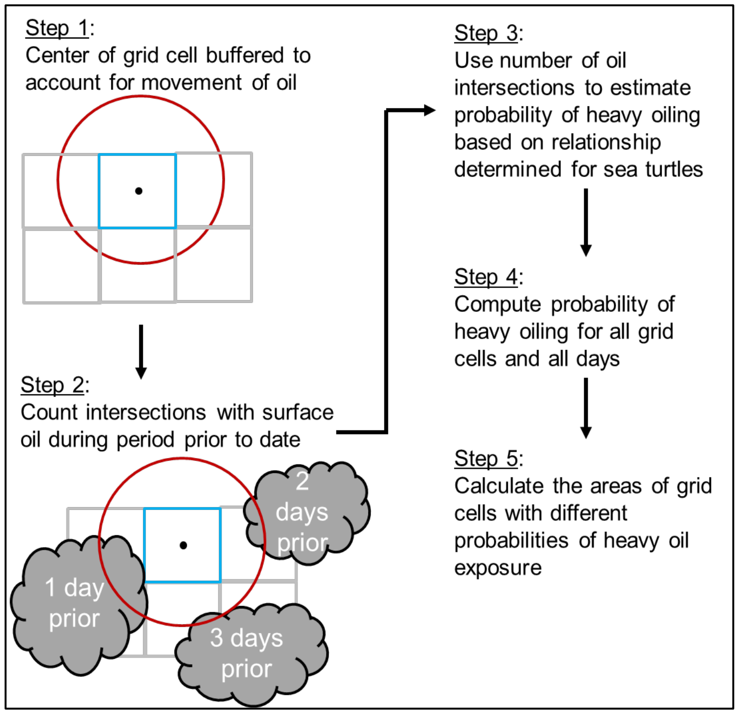
5. The total number of dead turtles was summed across oiling categories.

This general approach allowed the Trustees to calculate estimates of turtle abundance and exposures by species for the DWH spill area during the period from May through September 2010. Below, we describe in more detail how the Trustees performed these calculations using data collected during rescue and survey operations.

4.8.5.1.1 Exposure and Injury Quantification of Small Juvenile Turtles in Offshore Areas

To estimate actual abundance and exposures of turtles in open ocean areas, the Trustees first estimated densities of Kemp's ridleys, loggerheads, green turtles, and hawksbills from the boat-based turtle sightings and captures collected along search transects during rescue operations conducted between May and September 2010 (McDonald et al. 2015). These densities were calculated based on turtle capture and sightings data relative to more than 1,200 search transects that covered nearly 200 square kilometers of habitat for small juvenile sea turtles from May through September 2010. The Trustees used a well-established approach—i.e., distance sampling methodology (Buckland et al. 2001; Buckland et al. 2004)—to estimate the area that was searched during rescue operations, and then to generate turtle densities based on probability of sighting a turtle, distance from the transect line at which it was sighted, and other factors (McDonald et al. 2015).

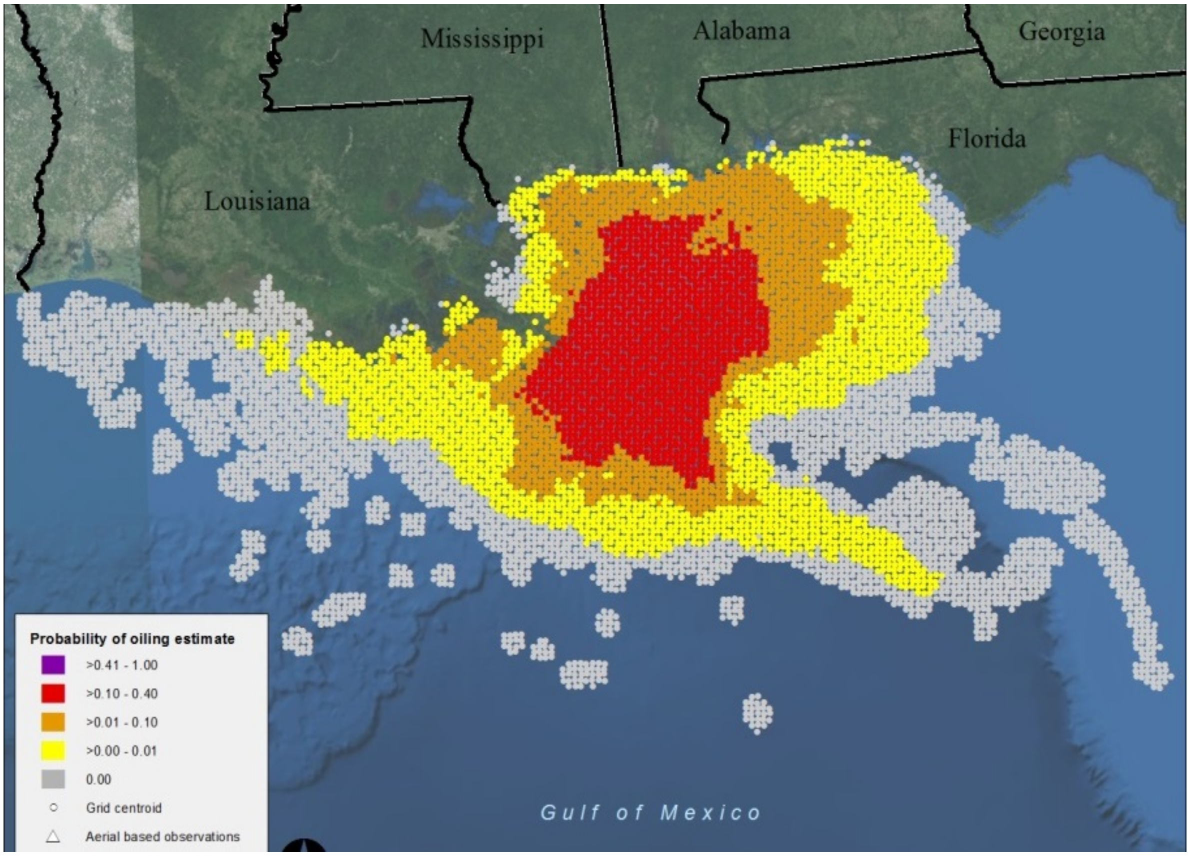
Because the cumulative area accounted for less than 1 percent of the total footprint area (approximately 100,000 square kilometers), spatial extrapolations of densities estimated in searched areas were required to estimate the total turtle abundance in the entire DWH oil spill area. To estimate total turtle abundance and the number of turtles in heavy and non-heavy oiling categories, the Trustees estimated the overall area to which the density estimates would apply—i.e., unsearched areas that would be expected to host similar densities of turtles to those observed in searched areas (Wallace et al. 2015). To do this, the Trustees adapted the modeling approach described in Section 4.8.4.4.1 (Mortality Estimates for Heavily Oiled Turtles) to estimate probability of heavy oiling across the cumulative DWH oil footprint. This was done following a similar procedure to how probability of oiling was estimated for turtles (Figure 4.8-13), but with important modifications (Figure 4.8-17) (Wallace et al. 2015).



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

Figure 4.8-17. Schematic of the modeling approach to estimate probability of heavy oil exposure across the cumulative oil footprint using the relationship between surface oil in the area and time prior to turtle captures and their observed degree of oiling described in Section 4.8.4.4.1 (Mortality Estimates for Heavily Oiled Turtles). Description of steps appears in the text.

First, instead of turtle capture/sighting locations and dates, the Trustees calculated the number of intersections between daily surface oil footprints between April 23 and August 11, 2010—i.e., the range of days for which satellite derived surface oil was present (see Section 4.2, Natural Resource Exposure)—and points within the cumulative oil footprint that intersected at all with surface oil (Figure 4.8-17 and Figure 4.8-18). Specifically, the cumulative oil footprint was divided into 5-kilometer by 5-kilometer (25 square kilometers) grid cells, and the points used in this procedure were the centers of each grid cell. The extent of the cumulative oil footprint is shown in grey in Figure 4.8-18. The quantification of intersections between surface oil and grid cells used the same spatial and temporal buffering procedure used on the turtle observation data (Wallace et al. 2015).



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

Figure 4.8-18. Mean probabilities of heavy oil exposure across the DWH oil spill zone and period. Values are mean probabilities calculated for the centers of all 5-kilometer x 5-kilometer grid cells of the integrated oil-on-water product (see Section 4.2, Natural Resource Exposure). Turtle densities calculated based on capture and sightings along search transects were multiplied by areas of cells within each probability bin shown above to estimate the number of turtles subject to heavy oil exposure.

Next, the Trustees used the numbers of intersections calculated for each grid cell for each day to calculate probabilities of heavy oil exposure for the entire DWH oil footprint for the duration of the spill period based on the relationship described in Section 4.8.4.4.1 (Mortality Estimates for Heavily Oiled Turtles) between observed degree of oiling of captured turtles and the nearby surface oil environment (Wallace et al. 2015). (See animation of *daily probabilities of heavy oiling for all turtles observed across the cumulative synthetic aperture radar [SAR] oil footprint: https://dwh.nmfs.noaa.gov/TeamCollaborationSites/DARP/Injury%20Volume/4.8%20Sea%20Turtles/Workspace/probability_maps.wmv*.) This step attributed probabilities of heavy oil exposure for every grid cell, for every day between April 23 and August 11 (Figure 4.8-18).

Once exposure areas were calculated, densities of heavily oiled turtles and non-heavily oiled turtles estimated within searched areas were multiplied by the cumulative areas that had non-zero probabilities of heavy oil exposure (i.e., all non-grey cells in Figure 4.8-18; see Figure 4.8-19 for the calculation). This straightforward approach assumed that average densities reflected both spatial and temporal variation in turtle densities, and that the DWH oil footprint encapsulated an average mix of turtle habitat and non-habitat areas (see Section 4.8.5.4, Uncertainties and Unquantified Injury, for uncertainties related to this assumption) (McDonald et al. 2015).

This calculation produced abundance estimates of heavily oiled turtles and non-heavily oiled turtles (Figure 4.8-19). The total estimated number of heavily oiled turtles that was based on the observed density of heavily oiled turtles was quantified as dead because the Trustees determined heavily oiled small juvenile turtles to have 100 percent probability of mortality (Wallace et al. 2015) (Section 4.8.4.4.1, Mortality Estimates for Heavily Oiled Turtles).

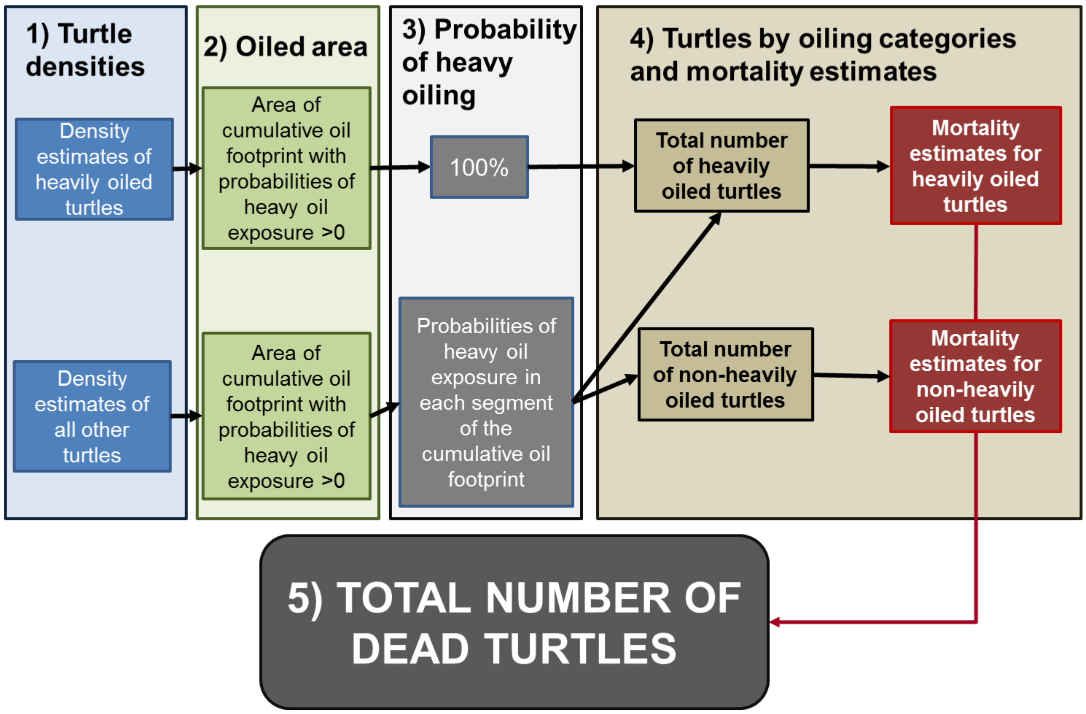
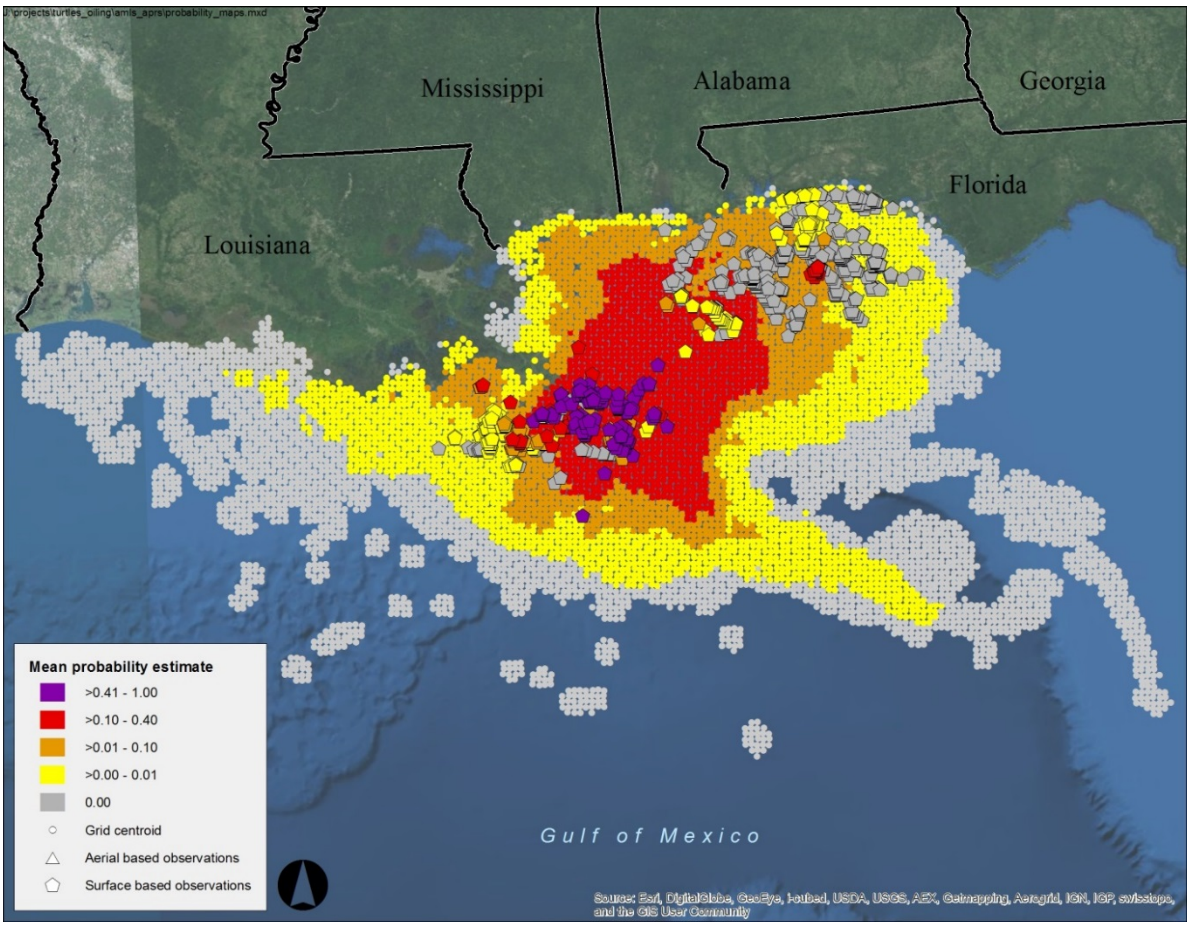


Figure 4.8-19. Schematic showing process by which the Trustees quantified injuries to small juvenile sea turtles in offshore areas using density estimates, the area within the cumulative oil exposure footprint, probabilities of heavy oil exposures, and mortality estimates. Descriptions of terms are included in the text.

The Trustees then calculated the total abundance of turtles by multiplying densities of turtles in non-heavily oiled categories by the total area defined above (Figure 4.8-18). Next, the Trustees multiplied this total abundance by the probability values themselves to estimate the number of turtles subject to heavy oil exposure. In this way, the density-area-probability of heavy oiling calculation provided estimates of numbers of turtles with a high probability of heavy oil exposure throughout the spill zone (Figure 4.8-18 and Figure 4.8-19). This number of juvenile turtles was considered heavily oiled, a condition directly linked to 100 percent mortality (Stacy 2012; Wallace et al. 2015).

The remainder of turtles from this calculation—i.e., the difference between total abundance of turtles in a given area and those estimated to be in the heavy oil category—represented turtles subject to less than heavy oil exposure. The mortality estimate for non-heavily oiled turtles (30 percent) (Table 4.8-3) (Mitchelmore et al. 2015) was applied to this number of turtles to estimate total dead turtles that were exposed to oil but not heavily oiled. Figure 4.8-19 provides a schematic summary of the calculation that the Trustees used to quantify dead turtles based on density and exposure areas.

To compare this approach with empirical observations of surface oil and degree of oiling on turtles, we assigned a probability of heavy oiling to all turtles captured or sighted during rescue operations, and the capture or sighting locations of these turtles were mapped onto the surface of average probability of heavy oil exposure estimated across the DWH oil footprint (Figure 4.8-20). As with the oiling probabilities across the DWH footprint (Figure 4.8-18), turtles with highest probabilities of heavy oil exposure were limited to areas closest to the wellhead, and probability of heavy oiling decreased with increasing distance from the wellhead (Figure 4.8-20) (Wallace et al. 2015).



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

Figure 4.8-20. Probabilities of heavy oil exposure of small juvenile sea turtles that were captured or sighted during rescue operations. Probabilities of heavy oiling associated with each turtle sighted or captured during surface-based surveys (i.e., sighted or captured from a boat on the water) overlaid on mean probabilities of heavy oil exposure across the DWH oil spill zone and period. Probabilities were estimated based on the statistical relationship between observed degrees of oiling on turtles and satellite-derived surface oil data in the area and time of turtle sighting or capture (see Section 4.8.4.4.1, Mortality Estimates for Heavily Oiled Turtles).

These results agreed with spatial patterns of degrees of oiling observed on rescued turtles; nearly all heavily oiled turtles were found within 90 kilometers of the wellhead, and turtles found farthest from the wellhead were assigned to the lowest oiling categories (Figure 4.8-20) (Stacy 2012; Wallace et al. 2015). (See animation of *daily probabilities of heavy oiling for all turtles observed across the cumulative SAR oil footprint:* *https://dwh.nmfs.noaa.gov/TeamCollaborationSites/DARP/Injury%20Volume/4.8%20Sea%20Turtles/Workspace/probability_maps.wmv*.) Given this high level of agreement with the empirical patterns of degree of oiling based on direct observations of turtles in oiled northern Gulf of Mexico habitats, we are confident in the results of the model in estimating degree of oiling for turtles—and surface habitats—that were not directly observed.

To quantify total injuries to small juvenile sea turtles, as described above and shown in Figure 4.8-19, the Trustees combined estimates of turtle density, areas within the oil footprint where turtles were expected to be, probabilities of heavy oil exposure, and mortality estimates for heavily and non-heavily oiled turtles. The Trustees estimated that approximately 402,000 small juvenile sea turtles were exposed to DWH oil across more than 100,000 square kilometers of the cumulative oil footprint (Table 4.8-4). The Trustees then applied the mortality estimates presented in Table 4.8-3 to the numbers of turtles exposed in each oiling category to estimate the numbers of turtles killed by oil exposure.

Based on these calculations, approximately 55,000 small juvenile sea turtles were likely killed by heavy exposure to DWH oil. Accounting for potential toxicological and physiological adverse effects associated with less than heavy oil exposure, up to an additional 104,000 turtles were likely killed. Thus, the Trustees estimated that as many as 160,000 small juvenile sea turtles were potentially killed by the DWH oil spill (Table 4.8-4) (Wallace et al. 2015).

Table 4.8-4. Densities, exposures, and estimated mortality of small juvenile sea turtles (Wallace et al. 2015). The Trustees calculated the number of turtles that were heavily oiled, and thus assumed dead (100 percent mortality for heavily oiled turtles), and also calculated the number of turtles exposed to a lesser degree than heavily oiled turtles. To estimate total potential mortality for the latter group, the Trustees applied a mortality rate of 30 percent, which considered potential toxic effects of oil exposure (see Section 4.8.4.4, Mortality Estimates for Turtles Based on Degree of Oiling). The Trustees considered the total number of heavily oiled dead oceanic turtles to be the low end of the range of mortality, and the addition of the number of less-than-heavily oiled dead oceanic turtles to be the high end of the range of total mortality. Exposure and injury estimates are shown to three significant digits.

Species	Density (Turtles/km^2)	Total Turtles Exposed	Heavily Oiled, Dead (low end of the range)	Non-heavily Oiled, Dead	Total Dead (high end of the range)
Kemp's ridleys	1.70	206,000	35,500	51,000	86,500
Loggerheads	0.25	29,800	2,070	8,310	10,400
Greens	1.22	148,000	15,300	39,800	55,100
Hawksbills	0.07	8,560	595	2,390	2,990
Unidentified	0.08	9,960	1,310	2,600	3,910
Total	3.32	402,000	54,8,000	104,000	159,000

Totals may not sum due to rounding.

Uncertainties Associated with Quantification of Injuries to Small Juvenile Sea Turtles

There are uncertainties associated with these estimates, as indicated by the wide range of possible injuries to small juvenile sea turtles. Sources of uncertainty in the mortality estimates, particularly those for non-heavily oiled turtles, included several factors that were not empirically observed in sea turtles (e.g., chronic toxic effects on critical physiological functions). These uncertainties could have either underestimated or overestimated mortality, as well as the resulting injury quantification. Calculations of area to which the density estimates were applied might have overestimated the total area where turtles might have been exposed, thus resulting in overestimates of small juvenile sea turtle injuries. However, other sources of uncertainty could have made the injury numbers presented in Table 4.8-4 underestimates. For example, several factors hindered searchers' ability to detect turtles, and rescue efforts covered less than 1 percent of the cumulative oil footprint. Despite uncertainties (see Section 4.8.5.4, Uncertainties and Unquantified Injury), the Trustees concluded that these injury estimates were reasonable and adequately quantify the magnitude of the injury to small juvenile sea turtles because they were based on the best available information and were quantified using sound technical approaches.

Putting Injury Estimates in Context

To put these numbers in context and evaluate their reasonableness, we estimated the potential number of small juvenile (1- and 2-year-old) Kemp's ridley turtles present in the Gulf during 2010. Although they occur throughout the northern Gulf of Mexico, 1- and 2-year-old Kemp's ridleys primarily inhabit the northern and eastern Gulf of Mexico (Putman & Mansfield 2015; Putman et al. 2013; Witherington et al. 2012). First, we started with numbers of hatchlings produced at the primary nesting beaches in 2008 and 2009 (i.e., nearly 2 million hatchlings released in those 2 years) (NMFS et al. 2011) and applied the annual survival rate for this age class (0.318) estimated by population modeling (Heppell et al. 2005). Based on this calculation, more than 430,000 small oceanic juvenile Kemp's ridleys (1- and 2-year-old) were estimated to have been in the population. Our exposure estimate of 206,000 Kemp's ridleys would mean that approximately half of these small juvenile Kemp's ridleys would have been exposed to surface oil during the DWH oil spill (Table 4.8-4). Total small juvenile Kemp's ridley mortalities were between 36,000 and 87,000 (Table 4.8-4), or approximately 10 to 20 percent of all of the 1- and 2-year-old Kemp's ridleys alive prior to the oil spill.

We emphasize that this set of calculations is qualitative in nature and not the result of robust demographic population modeling. For example, age- or stage-specific survival rates vary across years, and can significantly influence modeled population dynamics (Heppell et al. 2005). With this in mind, the results of these calculations demonstrate that our injury estimates, based on extrapolations of turtle densities from searched areas to the DWH footprint, are reasonable (i.e., well within estimates of actual population abundance), and represent a significant loss—perhaps 10 to 20 percent—of the small juvenile life stage of this endangered species. Although available data were insufficient to conduct similar calculations for loggerheads, green turtles, and hawksbills, we assume that the same approach to the injury estimate for the other sea turtle species has yielded similarly reasonable results, and the significance of the results for Kemp's ridleys apply in principle to the other species in offshore areas.

4.8.5.1.2 Exposure and Injury Quantification of Large Juvenile and Adult Sea Turtles in Continental Shelf Areas

To estimate total abundance and exposures of sea turtles in continental shelf waters, the Trustees performed similar calculations to those described above for small juveniles in offshore surface waters, with important modifications (Figure 4.8-21). First, the Trustees combined locations and dates of turtle sightings with environmental variables in the same areas and times to calculate turtle densities for biweekly periods from April to September 2010 (Garrison 2015). To correct for turtles that were submerged and thus not visible to observers performing surveys, the Trustees applied estimates of the time turtles spend at the surface by season to estimate the total number of turtles in a given area at a given time. This procedure accounts for all turtles in a given area based on the proportion that are visible. Additional corrections for detection probability, distance from tracklines, and other factors were also included in a distance sampling approach (Buckland et al. 2001; Buckland et al. 2004), as described above (Garrison 2015).

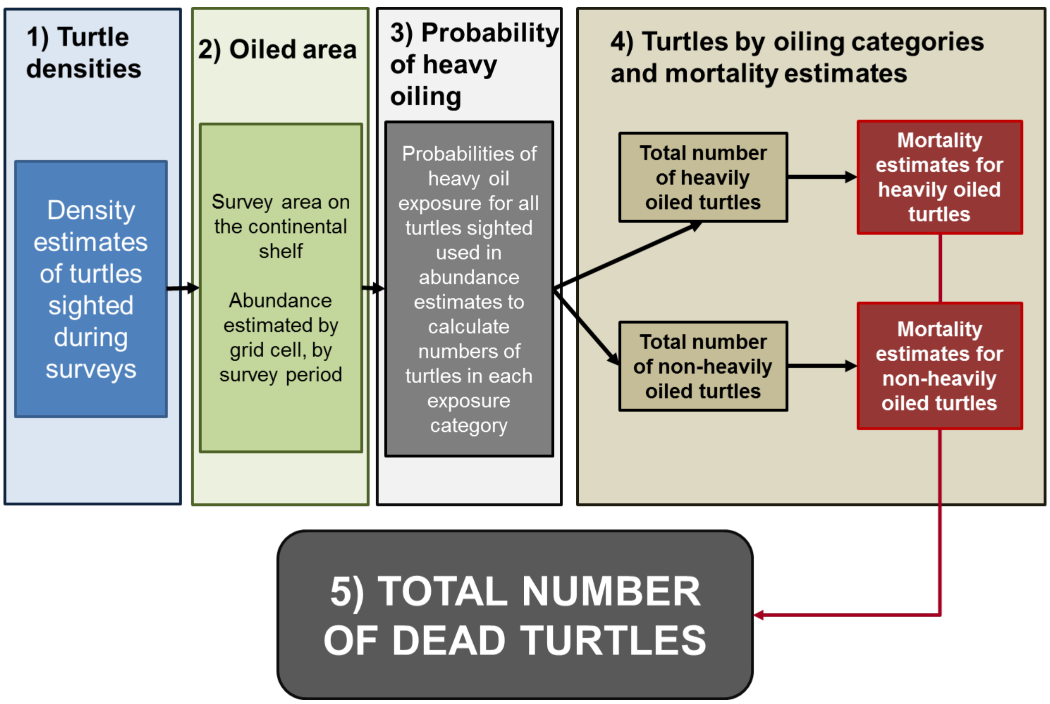
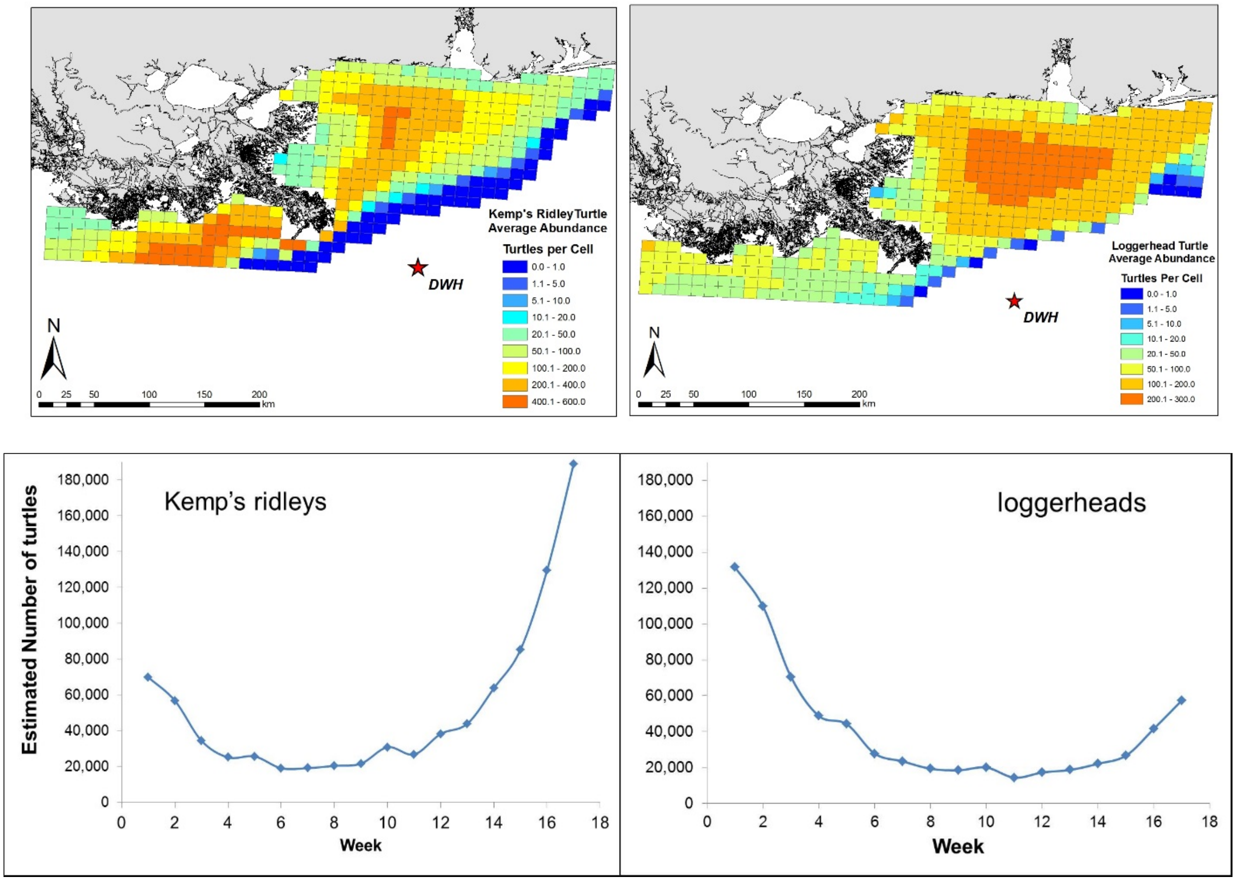


Figure 4.8-21. Schematic showing the process by which the Trustees quantified injuries to large juvenile and adult sea turtles on the continental shelf using density estimates, the area within the cumulative oil exposure footprint, probabilities of heavy oil exposures, and mortality estimates. Descriptions of terms are included in the text.

Turtle densities and estimated abundances for both species were highest in early and late summer, with lower abundance in the middle of the summer. However, areas and times of peak abundance, as well as abundance estimates themselves, differed between Kemp's ridleys and loggerheads (Figure 4.8-22) (Garrison 2015). Loggerheads were in consistently high densities in the eastern area of the survey area, whereas high densities of Kemp's ridleys were estimated both east and west of the Mississippi River Delta (Figure 4.8-22) (Garrison 2015). Both of these results are consistent with NRDA and non-NRDA satellite tracking studies that showed both migration and residency behaviors of adult female loggerheads and Kemp's ridleys in the area affected by the DWH oil spill (Figure 4.8-11) (Hart et al. 2012; Shaver et al. 2013).



__Source__: Garrison (2015).

Figure 4.8-22. Average density (top row) and abundance estimates (bottom row) for Kemp's ridleys (left) and loggerheads (right) on the northern Gulf of Mexico continental shelf during the DWH oil spill based on aerial survey observations and statistical estimates. Densities and abundance were calculated for the survey area shown during the period from April 24 to September 2, 2010.

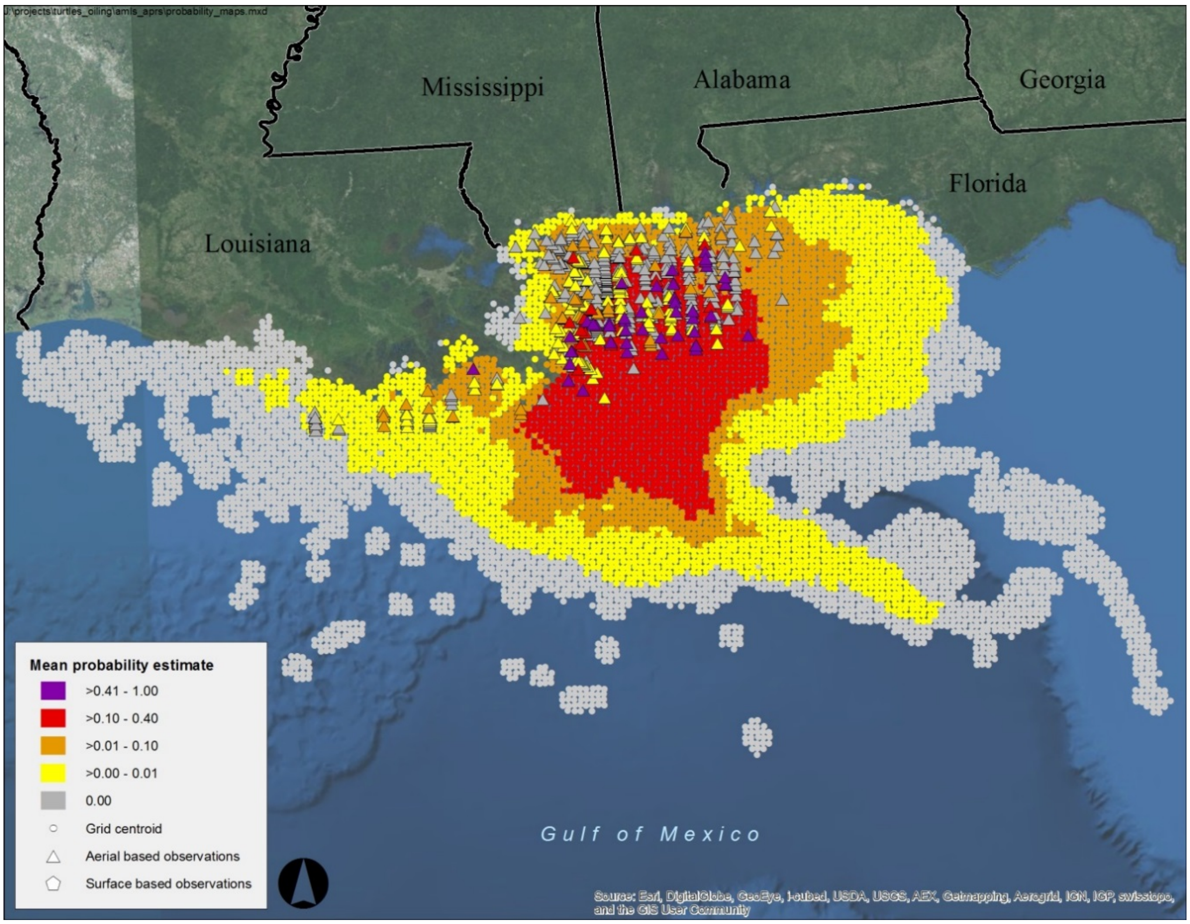
In contrast to rescue efforts to capture and evaluate oil exposure of small juvenile sea turtles, neritic turtles could not be directly assessed for oiling status or health condition. Therefore, the Trustees estimated the degree of oiling for these animals using the statistical relationship between observed degrees of oiling of rescued turtles and surface oil data derived from remote sensing (i.e., satellite data on daily size and distribution of the oil footprint) described in previous sections (Sections 4.8.4.4.1, Mortality Estimates for Heavily Oiled Turtles, and 4.8.5.1.1, Exposure and Injury Quantification of Small Juvenile Turtles in Offshore Areas) and in Figure 4.8-13 (Wallace et al. 2015). In this way, the Trustees were able first to assign oiling status to turtles that were observed remotely by plane but not assessed directly, and then to quantify oil exposures of turtles in continental shelf areas. This approach provided estimates of numbers of large juvenile and adult turtles in continental shelf waters with a high probability of heavy oil exposure throughout the spill zone. As discussed for small juvenile turtles in Section 4.8.5.1.1 (Exposure and Injury Quantification of Small Juvenile Turtles in Offshore Areas), the remainder of turtles from this calculation (i.e., the difference between total abundance of turtles in a given area and those estimated to be in the heavy oil category) represented large juvenile and adult turtles subject to less than heavy oil exposures. These calculations provided estimates of turtle abundance by exposure categories (Figure 4.8-21).

Density estimates were calculated per grid cell per survey, and included the probabilities of heavy oil exposure attributed to individual turtles that were sighted during surveys. In this way, when the Trustees multiplied densities by the survey area to calculate turtle abundances per survey period, they also were able to calculate the number of heavy and non-heavy oil exposures at these spatial and temporal scales (Figure 4.8-21) (Garrison 2015).

Probabilities of heavy oiling associated with each turtle sighted during aerial-based surveys (i.e., sighted from a plane) were overlaid on mean probabilities of heavy oil exposure across the DWH oil spill zone and period. Although aerial surveys were restricted to water depths of < 200 meters, and the survey area boundaries (Figure 4.8-10) were farther westward than areas where rescue operations occurred (Figure 4.8-7), the resulting probabilities of heavily oiled turtles detected by plane-based observers showed a similar relationship with distance from the wellhead; in general, probability of heavy oiling increased with proximity to the wellhead (Figure 4.8-23) (Wallace et al. 2015). These results were in agreement with several relevant and inter-related patterns, including: 1) directly observed frequencies of degrees of oiling on sea turtles obtained by rescue operations (Figure 4.8-7) (Stacy 2012), 2) surface oil prevalence (Figure 4.8-18), and 3) estimated probabilities of small juvenile turtles sighted or captured during rescue operations (Figure 4.8-20). (See animation of *daily probabilities of heavy oiling for all turtles observed across the cumulative SAR oil footprint: https://dwh.nmfs.noaa.gov/TeamCollaborationSites/DARP/Injury%20Volume/4.8%20Sea%20Turtles/Workspace/probability_maps.wmv*.) In other words, there was broad agreement between observed distributions of oiled turtles and surface oil in space and time, and the established relationship between the two produced reasonable estimates of degree of oiling for all sea turtles observed during NRDA response and survey efforts.

As discussed in Section 4.8.3.3.2 (Observations of Turtles on Continental Shelf and on Beaches), the Trustees identified a size class of young Kemp's ridley juveniles—approximately 25 to 40 centimeters long and approximately 3 years old (Avens & Snover 2013)—that were unobserved by vessel-based and plane-based surveys of the DWH spill area. Because the actual distribution of these animals is relatively poorly known, the Trustees estimated their abundance relative to the estimated abundance of all other continental shelf Kemp's ridleys, and assumed that they were uniformly distributed and exposed to oil in similar proportions to turtles that were sighted (Garrison 2015). This allowed the Trustees to estimate the potential number of these small, continental shelf juveniles that were exposed to and killed by the DWH oil spill (Table 4.8-5) (Wallace et al. 2015).

As described above and shown in Figure 4.8-21, the Trustees combined estimates of turtle density, areas within the oil footprint where turtles were expected to be, probabilities of heavy oil exposure, and associated mortality estimates to quantify total injuries to large juvenile and adult sea turtles in continental shelf areas. Based on these calculations, the Trustees estimated that more than 58,000 large juvenile and adult sea turtles were exposed to DWH oil in continental shelf areas (Table 4.8-5). Finally, the Trustees applied the mortality estimates for large turtles described in Section 4.8.4.4.2 (Mortality Estimates for Turtles That Were Exposed, But Not Heavily Oiled) to the numbers of turtles exposed in the heavy and non-heavy oil exposure categories. The Trustees estimated that approximately 4,900 large juvenile and adult sea turtles were likely killed by heavy exposure to DWH oil. Accounting for potential toxicological and physiological adverse effects of less than heavy oil exposure, up to an additional 2,600 sea turtles were likely killed. Thus, the Trustees estimated that a total of up to 7,600 large juvenile and adult turtles were potentially killed in continental shelf areas by the DWH oil spill (Table 4.8-5).



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

Figure 4.8-23. Probabilities of heavy oil exposure of large juvenile and adult sea turtles that were sighted during aerial surveys overlaid on mean probabilities of heavy oil exposure across the DWH oil spill zone and period. Probabilities were estimated based on the statistical relationship between observed degrees of oiling on turtles and satellite-derived surface oil data in the area and time of turtle sighting or capture (see Section 4.8.4.4.1, Mortality Estimates for Heavily Oiled Turtles).

Table 4.8-5. Total exposures and estimated mortality of large juvenile and adult sea turtles killed by the DWH oil spill in continental shelf areas (Wallace et al. 2015). We considered the total number of heavily oiled dead neritic turtles to be the low end of the range of mortality, and the addition of the number of less-than-heavily oiled dead neritic turtles to be the high end of the range of total mortality. Numbers only shown to two significant digits.

Species	Total Exposures	Heavily Oiled, Dead (low end of the range)	Non-heavily Oiled Exposures	Non-heavily Oiled, Dead	Total Dead (high end of the range)
Kemp's ridleys, ages 4+	21,000	1,700	19,000	950	2,700
Kemp's ridleys, age 3	990	380	610	30	410
Kemp's, all	*22,000*	*2,100*	*20,000*	*980*	*3,100*
Loggerheads	30,000	2,200	28,000	1,400	3,600
Unidentified	5,900	630	5,200	260	890
Total	58,000	4,900	53,000	2,600	7,600

Totals may not sum due to rounding.

Uncertainties Associated with Quantification of Injuries to Large Juvenile and Adult Sea Turtles

As with injury estimates for small juvenile sea turtles, there are uncertainties associated with these estimates of larger animals that could have either underestimated or overestimated mortality, as well as the resulting injury quantification. Mortality estimates for large juveniles and adults were developed based on strong empirical evidence about physical effects of exposure to surface oil observed in small juvenile sea turtles (Stacy 2012; Wallace et al. 2015), but empirical observations of oiled neritic sea turtles were scarce. Additionally, the Trustees made a simplifying assumption that the "missing size class" of neritic juvenile Kemp's were distributed and exposed equally to oil within the survey area, and that their mortality rates would be similar to those of larger neritic sea turtles. This distribution is unlikely as evidenced by the size classes represented among strandings, which suggests that the smaller turtles are generally more likely to be closer to shore and thus less likely to have become heavily oiled (Stacy 2012). However, there was insufficient information with which to more finely resolve size-related differences in distribution that would have been meaningful for incorporation into mortality estimates. Despite these uncertainties (see Section 4.8.5.4, Uncertainties and Unquantified Injury), the Trustees concluded that these injury estimates were reasonable and adequately quantify the magnitude of the injury to large juvenile and adult sea turtles because they were based on the best available information and were quantified using sound technical approaches.

Putting Injury Estimates in Context

To evaluate the reasonableness and significance of these injury estimates, we conducted a similar procedure to that presented for small juvenile sea turtle injury estimates above (Section 4.8.5.1.1, Exposure and Injury Quantification of Small Juvenile Turtles in Offshore Areas), again using Kemp's ridleys as an example. To estimate the total cohort of large juvenile and adult Kemp's ridleys turtles, which we estimated to correspond to turtles of ages 3 and older (Heppell et al. 2005), we started with the average number of hatchlings (approximately 1 million) produced on Kemp's ridley nesting beaches between 2007 through 2009 as a foundation for computing the total number of turtles across age classes. We then calculated the stable age distribution—i.e., the proportion of animals in each age class in a population—for the Kemp's ridley population with a growth rate of 18 percent per year based on published inputs to a stage-based population model (NMFS et al. 2011 and references therein).

By dividing the actual number of hatchlings by the estimated proportion of hatchlings in the population, we were able to estimate the total number of animals in the population. This number was then multiplied by the proportions of animals in each age class to calculate the number of animals in each age class. This procedure yielded a total estimated abundance of approximately 100,000 animals between three and 12+ years old.

Comparing the injury numbers to this estimate, approximately 22 percent (22,000 turtles) of the Kemp's ridleys in this age range were exposed to oil, and approximately 3 percent (3,100 turtles) of them died as a result (Table 4.8-5). Taking these calculations a step further to compute the potential number of adult Kemp's injured by the DWH oil spill yields an estimate of between 400 and 600 dead adult Kemp's. Using published sex ratios (i.e., the proportion of females to males) (NMFS et al. 2011) to account for only the females, estimated DWH injuries would equate to nearly 10 percent of the average annual nesting female abundance estimated between 2005 and 2010 (NMFS et al. 2011).

As above, we emphasize that this set of calculations is qualitative and not the result of robust demographic population modeling. There is variation in all of the parameters used above that is not reflected in the calculations or the results. For example, age at sexual maturity varies in natural populations, but was assumed to be fixed based on the best-fit value from population model results (Heppell et al. 2005). With this in mind, similar to the results presented above for the small juvenile injury estimates, our injury estimates based on extrapolations of turtle densities from searched areas to the DWH footprint are well within estimates of actual population abundance, but also represent a significant loss of the large juvenile and adult life stage for this endangered species. As above, we assume that both the reasonableness and the significance of the results for Kemp's ridleys apply in principle to the other species on the continental shelf.

4.8.5.2 Quantification of Sea Turtle Injuries Caused by DWH Response Activities

As mentioned in Section 4.8.4.6 (Injuries to Turtles Caused by Response Activities), the Trustees documented sea turtle injuries caused by activities associated with response to the DWH oil spill. Response injuries occurred in marine and terrestrial habitats.

4.8.5.2.1 Marine Response Injuries

The Trustees conclude that hundreds of sea turtles were likely killed by response activities in marine areas, but the actual number could not be quantified. Counting only those turtles directly observed, the Trustees documented six turtles killed by dredging operations or relocation trawling operations associated with construction of the berm in Louisiana. The Trustees also estimated that hundreds more turtles were killed by collisions with response watercraft based on those that stranded with clear evidence of watercraft collision injuries that coincided with response vessel traffic in nearshore areas throughout the northern Gulf of Mexico (Stacy 2015). In addition, many more turtles were likely killed during other response activities, such as oil skimming and burning operations; however, since these activities take place in areas of heavy oiling, these turtles are likely to have been included in the offshore injury quantification (McDonald et al. 2015). Although insufficient data were available to permit a robust quantification, the Trustees concluded that total injuries caused by response injuries in marine areas were likely to be hundreds of sea turtles, and possibly more.

4.8.5.2.2 Terrestrial Response Injuries: Hatchlings Released in the Atlantic to Avoid Oil in the Gulf of Mexico

Nests from three species—loggerheads, Kemp's ridleys, and green turtles—were excavated prior to emergence and eggs were translocated from Florida and Alabama beaches in the northern Gulf of Mexico between June 6 and August 19, 2010, to a protected hatchery on the Atlantic coast of Florida. More than 28,000 eggs from 274 nests were translocated, and nearly 15,000 hatchling turtles emerged and were released into the Atlantic Ocean (Table 4.8-6) (Provancha & Mukherjee 2011). Overall hatching success was 51.6 percent, and ranged from 0 percent to 100 percent among nests. Because these hatchlings entered the Atlantic Ocean and not the Gulf, and because sea turtle hatchlings are thought to imprint on their natal beaches to which they return as breeding adults, it is unknown whether these turtles will return to the Gulf (Provancha & Mukherjee 2011). Therefore, these hatchlings are assumed to be lost to the Gulf of Mexico breeding population as a result of the DWH oil spill.

Table 4.8-6. Summary of egg translocation and hatchling release effort to prevent Gulf of Mexico hatchlings from being exposed to DWH oil and response activities (Provancha & Mukherjee 2011).

	Clutches	Egg Count	Hatchlings Released
Kemp's ridley	5	483	125
Loggerhead	265	27,618	14,216
Green	4	580	455
Totals	**274**	**28,681**	**14,796**

4.8.5.2.3 Terrestrial Response Injuries: Disrupted Nesting

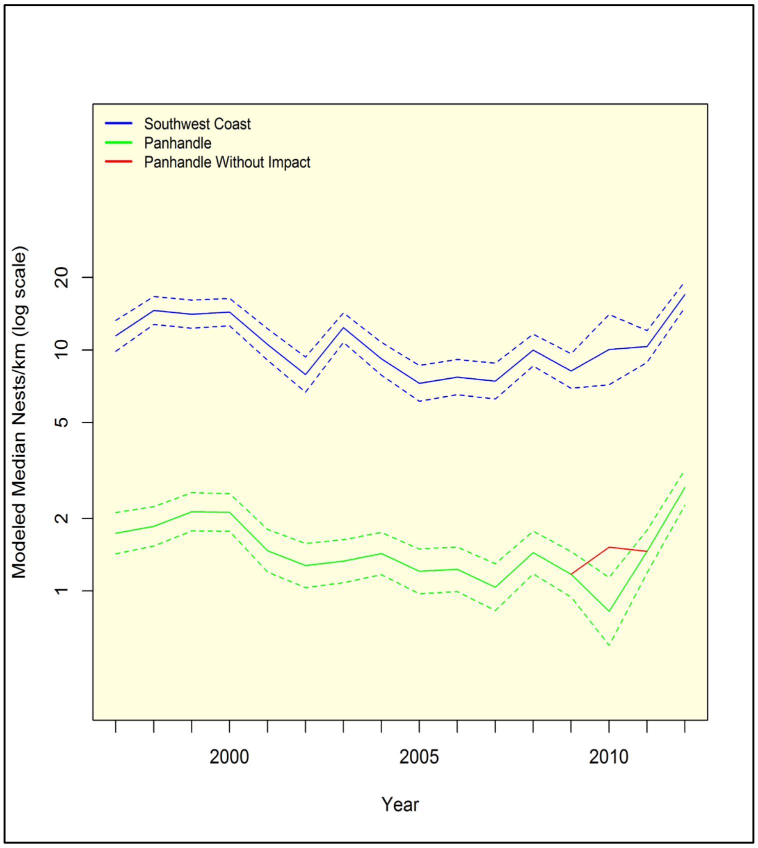
The Trustees evaluated nest losses on the Panhandle beaches of Florida due to response activities (Cacela & Dixon 2013; Frater 2015). Analyses focused on potential changes in nest densities in 2010 relative to the years before and after the spill on Florida Panhandle beaches compared to southwestern Florida beaches that were outside of the spill zone. These analyses confirmed a significant decrease of approximately 250 loggerhead nests on the Florida Panhandle nesting beaches in 2010 (Figure 4.8-24). The Trustees concluded that this decrease in nest density was related to oil cleanup operations on nesting beaches that deterred adult female loggerheads from coming ashore and laying their eggs. This estimated loss equates to approximately 18,000 unrealized hatchlings from Florida Panhandle nesting beaches in 2010 (Cacela & Dixon 2013).

In addition to the Florida Panhandle, Alabama beaches included within designated Loggerhead Critical Habitat under the ESA were also impacted by both oil and response activities during the 2010 nesting season (Michel et al. 2015). Nesting numbers for the beaches in Baldwin County, Alabama, showed a similar decline to those in the Florida Panhandle in 2010 (Frater 2015). Applying the observed proportional nesting density decrease relative to expected on the Florida beaches (Cacela & Dixon 2013) to the adjacent Alabama beaches, the Trustees estimated that approximately 30 loggerhead nests, or 2,000 loggerhead hatchlings, were lost in Alabama due to DWH response activities (Frater 2015).

Based on the response injury evaluations for Florida Panhandle and Alabama nesting beaches, the Trustees estimated that approximately 20,000 loggerhead hatchlings were lost due to DWH response activities on nesting beaches.

4.8.5.3 Total Estimate of Sea Turtles Killed by the DWH Oil Spill

Based on all efforts to quantify sea turtle injuries across life stages and habitats using observational data collected from land, air, and sea, and synthesized information from veterinary and toxicology assessments, the Trustees estimated that between 55,000 and as many as 160,000 small juvenile sea turtles, between 4,900 and as many as 7,600 large juvenile and adult sea turtles, and approximately 35,000 hatchling sea turtles were likely killed by the DWH oil spill and related response activities (Table 4.8-7). There are sources of uncertainty associated with these estimates, as were described previously and below in Section 4.8.5.4 (Uncertainties and Unquantified Injury). Most important, due to the logistical limitations of searching the vast spatial and temporal expanse of the DWH oil spill footprint, the total injury numbers presented here might underestimate the actual injury to sea turtles in the northern Gulf of Mexico. However, despite uncertainties, the Trustees concluded that the assessment adequately quantifies the nature and magnitude of injuries to sea turtles caused by the DWH oil spill. Furthermore, these uncertainties are best addressed by restoration approaches that are designed to address injuries across life stages and geographic areas.



Source: Cacela and Dixon (2013).

Figure 4.8-24. Loggerhead sea turtle nest densities in the Florida Panhandle were lower than expected due to response activities on beaches. Modeled values of annual nesting densities (nests/kilometer) in the southwest coast (blue lines) and the Panhandle (green lines) of Florida. The modeled estimate (red line) of the Panhandle nesting rate in the hypothetical absence of response activities in 2010. Dotted lines indicate the 95 percent confidence interval on the modeled median values of nest densities.

Table 4.8-7. Total estimate of sea turtles killed by the DWH oil spill, shown by life stage and by species. Two estimates of Kemp's ridley hatchling injuries are shown: 1) hatchlings lost due to response injuries, and 2) unrealized hatchling production caused by loss of breeding-age Kemp's ridleys quantified in Section 4.8.5.1.2 (Exposure and Injury Quantification of Large Juvenile and Adult Sea Turtles in Continental Shelf Areas). For more details about these calculations, see Section 4.8.5.4 (Uncertainties and Unquantified Injury). Lower ends of ranges for small juveniles and large juveniles and adults represent estimated injuries to heavily oiled turtles only; upper ends of ranges represent estimated injuries to heavily oiled turtles plus additional injuries estimated for non-heavily oiled turtles. ND=no data.

	Species					
Life Stage	Kemp's Ridley	Loggerhead	Green	Hawksbill	Unidentified Sea Turtle	Total
Hatchlings, response injuries	125	34,000	455	ND	ND	35,000
Hatchlings, unrealized reproduction	*65,000–95,000*[a]					
Small juveniles	36,000–87,000	2,100–10,000	15,000–55,000	600–3,000	1,300–3,900	55,000–160,000
Large juveniles and adults	2,100–3,100	2,200–3,600	ND	ND	630–890	4,900–7,600

Totals may not sum due to rounding.

[a] This range represents the estimated number of hatchlings that would have been produced by breeding Kemp's ridleys that were killed by the DWH oil spill. These numbers are a part of the total contribution of DWH to the unrealized Kemp's nest abundance observed between 2011 and the present, but do not include potential DWH impacts to reproductive fitness through sub-lethal impacts and impacts to habitat and foraging resources. Quantification of the full nature and magnitude of DWH effects requires further evaluation.

4.8.5.4 Uncertainties and Unquantified Injury

The total numbers of turtles killed by species and life stage shown in Table 4.8-7 include only those injuries that the Trustees could quantify, given available data. Despite the uncertainties and unquantified injuries described below, the Trustees concluded that the results of this injury quantification are reasonable and adequately reflect the nature and magnitude of the full injury to sea turtles caused by the DWH oil spill.

4.8.5.4.1 Uncertainty About Turtle Habitat Area

To quantify injuries of small juvenile sea turtles, the Trustees assumed that the entire exposure area of the DWH oil footprint represented potential turtle habitat at some point during the spill. This assumption clearly has a significant effect on the total injury estimate. If potential sea turtle habitat could be better defined in time and space (e.g., using habitat models to estimate the areas in which rescue crews would have searched based on environmental characteristics of areas that they were able to search) it is possible that the total estimated area in which turtles would have been exposed would have been smaller than what we estimated. This scenario would have resulted in lower injury quantification than presented in this assessment.

However, because the dataset of turtle observations that was the basis of the small juvenile abundance and exposure estimates was not based on a structured survey design—like the aerial survey dataset used to collect observations and estimate abundance of turtles on the continental shelf—statistically describing potential turtle habitat was not feasible. Furthermore, the assumption that average turtle densities could be applied across the cumulative oil footprint is reasonable for multiple reasons. First, given the extensive and constant movement of surface habitat with which these turtles are associated, *Sargassum* and convergence habitat could and does move throughout the northern Gulf of Mexico (see Section 4.4, Water Column). *Sargassum* grows at roughly 4 percent per day and is typically entrained in surface currents. Second, although small juvenile sea turtles are also typically entrained in these surface convergence zones, they are also capable of active, directed swimming (Putman & Mansfield 2015), which means that they can cross areas that might not be identified as putative turtle habitats.

On this basis, the Trustees concluded that the injury assessment for small juvenile turtles produced reasonable results that adequately describe the total injuries to this life stage.

4.8.5.4.2 Uncertainty Associated with Mortality Estimates of Non-Heavily Oiled Turtles

As described in detail, the Trustees concluded that heavily oiled small juvenile turtles would likely have died without medical intervention (see Section 4.8.4.4, Mortality Estimates for Turtles Based on Degree of Oiling). This assessment was based on evaluation of physical effects of miring in oil, including ingestion and internal exposure. Furthermore, this conclusion formed the basis of the modeling approach that assigned probabilities of heavy oiling to turtles based on spatial and temporal proximity to surface oil (Wallace et al. 2015). However, the Trustees had less certainty about mortality estimates for turtles that were exposed to oil but to a degree less than that observed for heavily oiled turtles. This was mainly due to the lack of conclusive adverse effects caused by oil exposure observed in 1) clinical parameters measured in oiled turtles brought to rehabilitation centers (Stacy & Innis 2012), and 2) results of the surrogate turtle toxicity study (Mitchelmore & Rowe 2015).

The toxicologist panel developed a 30 percent mortality estimate for non-heavily oiled turtles based on interpreting these studies in the context of existing literature that reported oil-induced physiological abnormalities in other vertebrates (Section 4.8.4.4.2, Mortality Estimates for Turtles That Were Exposed, But Not Heavily Oiled) (Mitchelmore et al. 2015). Because this mortality rate was applied to all exposed turtles that were not heavily oiled, it had a numerically significant effect on the total injury estimates. Given the uncertainty about toxicological effects of oil exposure on turtles (Section 4.8.4.2, Toxic Effects of Oil on Turtles and Their Habitats) (Mitchelmore & Rowe 2015; Mitchelmore et al. 2015), the 30 percent mortality rate applied to the non-heavily oiled turtles—and the resulting injury estimates—represents the high end of reasonable values. Nonetheless, it is realistic to conclude that mortality occurred due to exposure to oil at levels that did not reach the heavily oiled category.

4.8.5.4.3 Unquantified Injury to Leatherbacks

Leatherback turtles were sighted within the DWH oil spill footprint during offshore rescue efforts and aerial surveys over the continental shelf during the DWH oil spill. However, several factors prevented the Trustees from being able to quantify leatherback injuries caused by the DWH oil spill. First, because leatherbacks do not typically associate with convergence areas that were the targets of field crews searching for small juvenile turtles of other species (Bolten 2003), densities of leatherbacks could not be estimated within the searched area. Second, too few leatherbacks were seen during aerial surveys to include leatherbacks in abundance modeling conducted for other species. Third, due to logistical constraints related to leatherbacks' massive size and competing resource needs that prevented allocation of a dedicated effort, the Trustees could not capture leatherbacks to assess their degree of oiling and associated health status.

For these reasons, the Trustees did not estimate leatherback abundance and exposure in the DWH spill area. However, given that the northern Gulf of Mexico is important habitat for leatherback migration and foraging (Turtle Expert Working Group 2007), and documentation of leatherbacks in the DWH oil spill zone during the spill period, the Trustees conclude that leatherbacks were exposed to DWH oil, and some portion of those exposed leatherbacks likely died.

4.8.5.4.4 Strandings on Beaches Are a Poor Indicator of Oil-Caused Mortality of Sea Turtles

As mentioned previously (Section 4.8.3.3.2, Observations of Turtles on Continental Shelf and on Beaches), strandings represent a small fraction of total mortality and generally are not representative of mortality occurring farther from shore or in areas where animals are unlikely to be found (Epperly et al. 1996; Hart et al. 2006; Nero et al. 2013; Williams et al. 2011). To examine the probabilities that animal carcasses would reach coastlines depending on when and where they died and began to drift, the Trustees performed particle drift modeling to simulate how animal carcasses would likely move and possibly strand in the northern Gulf of Mexico (Wirasaet et al. 2015). Results demonstrated that animals that died on the outer portion of the continental shelf and remained floating at the surface would have had an extremely low probability of stranding (< 3 percent, regardless of date).

Indeed, the vast majority of turtles that the Trustees estimated to have died were located tens of kilometers (or more) from shore (Figure 4.8-20 and Figure 4.8-23), and very few stranded animals had evidence of oil exposure. Animals that died within a few kilometers of shore had a higher probability of stranding, but the strandings simulation model showed that particles appeared to strand in areas where detection was low or non-existent (e.g., Chandeleur Islands).

In summary, turtles that became impaired or died at sea beyond areas very close to shore (< 5–10 kilometers) or stranded within areas where the coastline is predominantly marsh or other relatively inaccessible habitat (e.g., much of the Louisiana coast) were extremely unlikely to have come ashore or to be found even if they did reach shore. Therefore, the Trustees concluded that turtles dying from acute effects of the spill were unlikely to be found on shore as strandings. Further, most animals that are recovered are too decomposed to determine cause of death. For these reasons, strandings were not used to quantify the magnitude of injury to sea turtles caused by DWH oil exposure.

4.8.5.4.5 Turtles That Were Sighted But Not Identified to Species

The Trustees estimated exposed and injured sea turtles that were not identified to species (Table 4.8-4, Table 4.8-6, and Table 4.8-7). These animals were sighted during rescue operations or aerial surveys, but due to various factors, including turtles evading capture attempts by rescue workers, challenging visibility conditions, and others, species identification was not possible. Nonetheless, they were included in quantification of exposure and injury because they represent confirmed locations of sea turtles within the DWH spill footprint and time period. Given the uncertainty about these injuries, restoration approaches to reduce principal threats to sea turtles across geographic areas will be most effective in restoring the full magnitude of sea turtle injuries, including turtles that were not identified to species.

4.8.5.4.6 Unquantified Injury Due to Marine and Terrestrial Response Activities

The Trustees also estimated that hundreds more turtles were killed by collisions with response watercraft. In addition, many turtles were likely killed during other response activities, such as oil skimming and burning operations (turtles in the latter group were likely accounted for in the offshore injury quantification). Although insufficient data were available to permit a robust quantification, the Trustees concluded that total injuries caused by marine response injuries were likely to be at least hundreds of sea turtles.

While there was a significant effort to translocate as many eggs as possible in 2010, not all nests were found. Oil eventually did come ashore, and extensive monitoring and cleanup activities were conducted on many of the northern Gulf of Mexico sea turtle nesting beaches. Therefore, there was an unquantified injury to nests (eggs) and/or hatchlings emerging from nests that were missed on nesting beaches where oil came ashore and where response activities took place. These nests (eggs) and/or emergent hatchlings were likely killed.

4.8.5.4.7 Unrealized Kemp's Ridley Reproduction Due to Oil Exposure of Adult Females and Recruitment Failures

As described above in Section 4.8.4.7 (Reduced Kemp's Ridley Nesting Abundance and Hatchling Production), the Kemp's ridley population has gone from high abundance to near extinction to significant recovery over the past 50 plus years, since scientific discovery of the species' primary nesting location in Tamaulipas, Mexico, in 1947 (Gallaway et al. 2013; NMFS et al. 2011). Estimated abundance peaked in the 1940s, but declined to its lowest level in the 1980s. Owing to increased conservation efforts in the 1980s and early 1990s, the nesting population increased exponentially from the mid-1980s through 2009. However, nesting was lower than expected in 2010, and has been lower than expected every nesting season since (Figure 4.8-15) (Dixon & Heppell 2015; Gallaway et al. 2013). The reduction in expected nests, based on the trajectory of nest counts prior to 2010, continues to be a critical concern and focus of analysis efforts. The DWH oil spill is one potential factor contributing to the observed decline in Kemp's ridley nesting relative to projected abundance increases based on trends from 1996 through 2009 (Crowder & Heppell 2011; Gallaway et al. 2013; NMFS et al. 2011).

The Trustees quantified mortality of Kemp's ridleys on the continental shelf during 2010 (Section 4.8.5.1.2, Exposure and Injury Quantification of Large Juvenile and Adult Sea Turtles in Continental Shelf Areas; Table 4.8-6). Between 400 and 600 adult (ages 12+ years old) and between 300 and 450 sub-adult (approximately 9–11 years old) Kemp's ridleys were killed on the continental shelf during 2010 based on the validation exercise described in Section 4.8.5.1.2 (Exposure and Injury Quantification of Large Juvenile and Adult Sea Turtles in Continental Shelf Areas). These approximately 1,000 animals included reproductive females and sub-adults that would have recruited to the breeding population between 2011 and the present. The loss of these reproductive-stage females would have contributed to some extent to the decline in total nesting abundance observed between 2011 and 2014 (Figure 4.8-15).

Using published sex ratios (i.e., the proportion of females to males) (NMFS et al. 2011) to account for only the females among these dead adult Kemp's ridleys, estimated DWH injuries would equate to roughly 10–20 percent of the average annual nesting female abundance estimated between 2005 and 2010 (NMFS et al. 2011). Using conservative values for sex ratios, clutches per female, egg to hatchling survival, and hatchlings per clutch (NMFS et al. 2011), the estimated number of unrealized Kemp's ridley nests was between 1,300 and 2,000, which translates to approximately 65,000 and 95,000 unrealized hatchlings (Table 4.8-7).

This unrealized reproduction accounts for less than 3 percent of the estimated cumulative deficit of approximately 79,000 nests from 2011 to 2014 (Figure 4.8-15) (Dixon & Heppell 2015). However, the relationship between the injury we quantified here and unrealized reproduction represents a portion, but not all, of the overall potential DWH effect. Sub-lethal effects of DWH oil on turtles, their prey, and their habitats might have delayed or reduced reproduction in subsequent years and also contributed substantially to the observed deviation from projected nest abundance. These sub-lethal effects could have slowed growth and maturation rates, increased remigration intervals, and decreased clutch frequency (number of nests per female per nesting season) of Kemp's ridley turtles, as reported in other sea turtle species when resource availability is reduced (Wallace & Saba 2009). However, the actual nature and magnitude of the DWH effect on reduced Kemp's ridley nesting abundance and associated hatchling production after 2010 requires further evaluation.

Injuries to adult turtles of other species, such as loggerheads, certainly would have resulted in unrealized nests and hatchlings to those species as well. However, we restricted this illustrative calculation of unrealized nests and hatchlings to Kemp's ridleys for several reasons. All Kemp's ridleys in the Gulf belong to the same population (NMFS et al. 2011), so total population abundance could be calculated based on numbers of hatchlings because all individuals that enter the population could reasonably be expected to inhabit the northern Gulf of Mexico throughout their lives. In contrast, other species present in the Gulf that were injured by the DWH oil spill—e.g., loggerheads—come from multiple breeding populations, and not all individuals from all breeding populations inhabit the northern Gulf of Mexico. Therefore, calculating the number of adults and sub-adults was not feasible with available data, and the number of unrealized hatchlings resulting from injuries to breeding turtles could not be verified by evaluation of nesting trends for a particular population. In summary, any injury to adult sea turtles of any species caused by DWH reduced total hatchling production, which would have a negative effect on the overall population.

4.8.5.5 Summary of Injury Quantification

In this section, the Trustees presented quantification of total injuries to sea turtles across life stages and geographic areas based on a synthesis of observations and evaluations of exposure pathways (Section 4.8.3, Exposure) and effects (Section 4.8.4, Injury Determination). Despite uncertainties and unquantified injuries, the Trustees concluded that these injury estimates adequately describe the nature and magnitude of total injuries to sea turtles caused by the DWH oil spill and associated response injuries.

4.8.6 Conclusions and Key Aspects of the Injury for Restoration Planning

4.8.6.1 Summary

Based on the Trustees' quantification of sea turtle injuries caused by the DWH oil spill, sea turtles from all life stages and all geographic areas were lost from the northern Gulf of Mexico ecosystem. In particular, up to hundreds of thousands of small juvenile turtles died from DWH oil exposure and response activities. These animals are noteworthy because they belong to a unique, ephemeral community consisting of open-ocean, surface-dwelling fauna (e.g., juvenile fish and invertebrates) and flora (e.g., *Sargassum*) that is distributed in non-uniform, non-stationary clumps, strips, and patches across the northern Gulf of Mexico. Additionally, DWH oil caused significant losses of *Sargassum* habitat itself, which further compounds the impacts on sea turtles and their ability to recover (see Section 4.4, Water Column).

Specifically, the injury assessment showed that:

- Four species of sea turtles that inhabit the Gulf were injured by the DWH oil spill (Kemp's ridley, loggerhead, green, and hawksbill). A fifth species, the leatherback, was likely exposed to DWH oil in the northern Gulf of Mexico, and some leatherbacks that were exposed likely died; however, quantification of leatherback injury was not undertaken. All of these species are listed as threatened or endangered under the ESA, are long-lived, and travel widely, occupying a variety of habitats across the Gulf and beyond.
- Sea turtles were injured by oil in the open ocean, nearshore, and shoreline environments, and resulting mortalities spanned multiple life stages. The Trustees estimated that between 4,900 and as many 7,600 large juvenile and adult sea turtles (Kemp's ridleys, loggerheads, and hardshelled sea turtles not identified to species), and between 55,000 and as many as 160,000 small juvenile sea turtles (Kemp's ridleys, green turtles, loggerheads, hawksbills, and hardshelled sea turtles not identified to species) were killed by the DWH oil spill.
- Nearly 35,000 hatchling sea turtles (loggerheads, Kemp's ridleys, and green turtles) were injured by response activities, and thousands more Kemp's ridley and loggerhead hatchlings were lost due to unrealized reproduction of adult sea turtles that were killed by the DWH oil spill.
- In addition, the injury assessment included injuries that were determined to have occurred, but were not formally quantified, such as unquantified injuries to leatherback turtles.

The Trustees considered all of these aspects of the injury in restoration planning, and also considered the ecosystem effects and recovery information described below.

4.8.6.2 Ecosystem Effects

Sea turtles have been integral and long-term members of marine ecosystems for countless generations. Significant losses of sea turtles remove them from complex inter-species relationships and ecological processes, such as transport of nutrients between marine and terrestrial habitats, likely impacting the northern Gulf of Mexico marine ecosystem (Bjorndal & Jackson 2003). In addition to serving important ecological roles, sea turtles are also extremely valuable natural resources to humans. Sea turtles' protected status under the ESA—as well as their status under international conservation treaties and agreements (e.g., IAC, CITES, IUCN *Red List*)—means that they are considered to be in danger of extinction if current threats are not reduced. In addition, their ESA-listed status means that the public values their continued existence and efforts to recover and protect them. Sea turtles are also valuable to the public as subjects of wildlife-viewing activities, whether through formal ecotourism or informal enjoyment of nature. In nearly every country in the world where sea turtles are present, particularly where they nest, people make efforts to observe sea turtles in the wild. This is especially true in the United States, including in Gulf Coast states.

4.8.6.3 Recovery

Sea turtle populations can recover when primary threats are reduced or eliminated, provided sufficient abundance remains (Dutton et al. 2005; NMFS et al. 2011) and sufficient habitat resources are available (Wallace & Saba 2009). However, the majority of historically depleted sea turtle populations worldwide tend to show slow recovery, and some show no recovery at all, despite decades of conservation management efforts (Williams et al. 2011). Slow recovery is common because sea turtles have decades-long lifespans, overlapping generations, and wide distributions over which resource availability and impacts of threats can vary tremendously. Additionally, the differences in or lack of regulatory regimes among nations complicate and can impede recovery. Therefore, given the combination of sea turtle life history traits and the magnitude of the quantified injuries, recovery of sea turtle populations affected by the DWH oil spill to baseline levels would be expected to take decades if no restoration actions were taken.

The complex and transient nature of the sea turtle population structure and the significant magnitude of the mortality resulting from the DWH oil spill will make complete recovery challenging. This is partly due to the fact that precisely assessing the structure and abundance of sea turtle populations is difficult because of the significant logistical and technical challenges associated with obtaining robust estimates of vital rates, including life stage-specific survivorship and life stage durations. For these reasons, the Trustees conclude that the recovery of sea turtles in the northern Gulf of Mexico from injuries caused by the DWH oil spill will require decades of sustained efforts to reduce the most critical threats and enhance survival of turtles at multiple life stages.

4.8.6.4 Restoration Considerations

As described in Chapter 5 (Section 5.5.10 and 5.5.2), the Trustees have identified an integrated portfolio of restoration approaches that can address all species and life stages that were injured by the spill and takes into consideration key threats to sea turtles.

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4.9 Marine Mammals

What Is in This Section?

Executive Summary

The DWH oil spill resulted in the contamination of prime marine mammal habitat in the estuarine, nearshore, and offshore waters of the northern Gulf of Mexico. In order to determine the exposure and injury to whales and dolphins due to the DWH oil spill, the Trustees synthesized data from specific NRDA field studies, stranded carcasses collected by the Southeast Marine Mammal Stranding Network, historical data on marine mammal populations, NRDA toxicity testing studies, and the published literature. Tens of thousands of marine mammals were exposed to the DWH surface slick, where they likely inhaled, aspirated, ingested, physically contacted, and absorbed oil components. The oil's physical, chemical, and toxic effects damaged tissues and organs, leading to a constellation of adverse health effects, including reproductive failure, adrenal disease, lung disease, and poor body condition. Animals that succumbed to these adverse health effects contributed to the largest and longest-lasting marine mammal unusual mortality event (UME) on record in the northern Gulf of Mexico. The dead, stranded dolphins in the UME included near-term fetuses from failed pregnancies. Similarly, in the 5 years after the oil spill, more than 75 percent of pregnant dolphins observed within the oil spill footprint failed to give birth to a viable calf.

Based on the Trustees' scientific findings, the DWH oil spill is the most likely explanation for the injuries to marine mammals observed since May 2010 in the oil spill footprint. The increases in mortality, reproductive failure, and specific adverse health effects were seen in animals within the oil spill footprint and were not observed in animals outside of the footprint. The injuries were most severe in the year immediately following the spill and have improved slightly over time (but not completely disappeared) since the well was capped. The types of injuries observed are consistent with both the known routes of exposure and the toxic effects reported in the oil toxicity literature, and were identified in both living animals and dead, stranded dolphins. For example, marine mammals that inhale or aspirate oil components are likely to experience physical and chemical damage in the lungs, which is consistent with the increased prevalence of lung disease in live animals and of bacterial pneumonia in stranded animals within the oil spill footprint (Venn-Watson et al. 2015a).

In the absence of active restoration, marine mammal stocks in the northern Gulf of Mexico will require decades to recover from the effects of the DWH oil spill. Nearly all of the stocks that overlap with the oil spill footprint have demonstrable, quantifiable injuries. The remaining stocks (for which there is no quantifiable injury) were also likely injured, but there is not enough information to make a determination at this time. The Barataria Bay and Mississippi Sound bottlenose dolphin stocks were two of the best-studied populations, with 51 percent and 62 percent, respectively, projected maximum reductions in their population sizes. Without any active restoration, these populations will take approximately 40 to 50 years to fully recover. While smaller percentages of the oceanic stocks were exposed to DWH oil, these stocks still experienced increased mortality (as high as 17 percent), reproductive failure (as high as 22 percent), and adverse health effects (as high as 18 percent).

The Trustees have determined that marine mammals in the northern Gulf of Mexico were exposed to DWH oil and were injured as a result of the spill. Without active restoration, these protected, at-risk populations will suffer from the effects of the DWH oil spill for decades to come.

4.9.1 Introduction and Importance of the Resource

Key Points

- There are 22 species of marine mammals in the northern Gulf of Mexico, including manatees in coastal seagrasses and cetaceans (dolphins and whales) in estuarine, nearshore, and offshore habitats.
- There is a wide diversity of cetacean species, including animals that differ in size and physiology, feeding habits, and life histories. Many of the species are apex predators that rely on a wide variety of resources in the marine ecosystem.
- For the purposes of tracking populations, marine mammal species in U.S. waters are delineated into stocks based on a variety of data. There are bottlenose dolphin stocks in each bay, sound, and estuary system and in coastal waters; there are dolphin stocks that live over the continental shelf; and there are dolphin and whale stocks that live in the deeper, oceanic waters.
- Marine mammal populations have been severely impacted by human activities, including commercial and recreational fisheries, pollution, industrial activities, vessel strikes, and intentional harm. To address declining populations, all marine mammals are now protected under the Marine Mammal Protection Act, which prohibits individuals from harassing, harming, or disturbing marine mammals. Some, including sperm whales and manatees, are also protected

under the Endangered Species Act.

- In order to investigate the DWH-related injuries to northern Gulf of Mexico cetaceans, scientists navigated a variety of regulatory, ethical, and logistical challenges common for research on cetaceans.

4.9.1.1 What Are Marine Mammals?

Like most mammals, marine mammals are warm-blooded, give birth to live young, nurse their young, and breathe air. In contrast, however, they spend significant periods of their lives under the water. There are 22 marine mammal species found in the northern Gulf of Mexico representing two classes of marine mammals: cetaceans (21 species), which include whales and dolphins; and sirenians (one species), which include manatees (Table 4.9-1). Manatees primarily inhabit the coastal waters of Florida, but can occasionally be found in seagrass habitats as far west as Texas. In contrast, cetaceans have adapted to a wide variety of habitats in the marine environment and can be found throughout the northern Gulf of Mexico (Rosel & Mullin 2015). They feed at all trophic levels, consuming foods ranging from invertebrates to large fish. Cetaceans are agile, efficient swimmers, and some species have been known to submerge for more than an hour. They are also highly intelligent, capable of self-recognition, have developed sophisticated communication strategies, and form complex social structures. Due to their long lives, unique physiology, and the fact that many feed at high trophic levels, researchers often consider marine mammals as sentinel species for marine ecosystem health (Bossart 2011; Moore 2008; Reddy et al. 2001; Ross 2000; Wells et al. 2004).

Table 4.9-1. Twenty-two marine mammal species are found in the northern Gulf of Mexico.

Common Name/Species	Specific Name
Atlantic spotted dolphin	*Stenella frontalis*
Blainville's beaked whale	*Mesoplodon densirostris*
Bryde’s whale	*Balaenoptera edeni*
Clymene dolphin	*Stenella clymene*
Common bottlenose dolphin	*Tursiops truncatus*
Cuvier's beaked whale	*Ziphius cavirostris*
Dwarf sperm whale	*Kogia sima*
False killer whale	*Pseudorca crassidens*
Fraser’s dolphin	*Lagenodelphis hosei*
Gervais’ beaked whale	*Mesoplodon europaeus*
Killer whale	*Orcinus orca*
Melon-headed whale	*Peponocephala electra*
Pantropical spotted dolphin	*Stenella attenuata*
Pilot whale (short-finned)	*Globicephala macrorhynchus*
Pygmy killer whale	*Feresa attenuata*
Pygmy sperm whale	*Kogia breviceps*
Risso’s dolphin	*Grampus griseus*
Rough-toothed dolphin	*Steno bredanensis*
Sperm whale	*Physeter macrocephalus*
Spinner dolphin	*Stenella longirostris*
Striped dolphin	*Stenella coeruleoalba*
West Indian manatee	*Trichechus manatus*

Whales and dolphins have suffered from their interactions with humans, including detrimental effects from unsustainable hunting, entanglements in fishing gear, chemical contaminant pollution, vessel noise pollution, incidental vessel strikes, and overfishing of prey species (Read et al. 2006; Reeves et al. 2013). Biologists and policymakers realized that without intervention to reduce such practices, many species of marine mammals would be driven to extinction. With the passage of the U.S. Marine Mammal Protection Act (MMPA) in 1972, all species of marine mammals in U.S. waters were granted protection. MMPA includes strict prohibitions against any “take” of a marine mammal, meaning it is unlawful to harass, hunt, capture, or kill, or attempt to do so. Several northern Gulf of Mexico stocks are listed as strategic under the MMPA, meaning they are particularly at risk of unsustainable population numbers. The Endangered Species Act (ESA) was passed in 1973 and further protects those species at high risk of extinction. While the MMPA and the ESA have helped to stabilize some marine mammal populations, sperm whales in the Gulf of Mexico are still at risk and are listed as endangered by the ESA (35 FR 18319, December 2, 1970) as are West Indian manatees (32 FR 4001, March 11, 1967). As demonstrated both by acts of legislation and by consensus within the field of marine biology/ecology, any threat to the well-being of marine mammals, as individuals or to populations, is a serious and detrimental activity that could have repercussions for the entire ocean ecosystem (Bossart 2011; Moore 2008; Reddy et al. 2001; Ross 2000; Wells et al. 2004).

4.9.1.2 Marine Mammal Stocks

Marine mammals live everywhere in the Gulf of Mexico, including within bays, sounds, and estuaries (BSEs); along the coast (coastal); over the continental shelf (shelf); and in deeper waters (oceanic) (Rosel & Mullin 2015). While most species reside in the oceanic habitat, the Atlantic spotted dolphin is found over the continental shelf, and the common bottlenose dolphin (referred to as “bottlenose dolphin” throughout this chapter) inhabits oceanic, shelf, coastal, and BSE waters (Figure 4.9-1)(Rosel & Mullin 2015). Although they are all the same species, bottlenose dolphins in the northern Gulf of Mexico can be separated into demographically independent populations called stocks.

A stock of bottlenose dolphins generally shows a strong attachment (i.e., site fidelity) to a geographic area, exhibiting low levels of immigration and emigration. Therefore, the population level of a given stock is generally assumed to be unaffected by population fluctuations in other stocks. For example, the Sarasota Bay bottlenose dolphin stock, which has been studied since 1970, has up to five concurrent generations of animals that have mostly spent their entire lives within the same bay (Wells & Scott 2009).

There are currently 37 stocks of bottlenose dolphins in the northern Gulf of Mexico, 13 of which (9 BSE stocks, 2 coastal stocks, 1 shelf stock, and 1 oceanic stock) are found in areas within the DWH oil spill footprint. The other 20 species of whales and dolphins in the northern Gulf of Mexico are each managed as a single, Gulf-wide stock; the ranges of 18 of these stocks overlap with the DWH oil spill footprint. There are not enough data to make a determination about the overlap between the spill footprint and the ranges of killer whales or Fraser’s dolphins (Table 4.9-1). While the distribution of West Indian manatees overlaps with the DWH oil footprint, none were sighted in oil, and they are not considered further in this assessment. (Throughout the rest of this document, the term “marine mammal” is used in section headings and general references that would include non-cetaceans; references to “cetaceans” are specific to cetaceans.).

For the purposes of the DWH NRDA, the Trustees have considered the impact of the oil spill on the many different species and stocks of cetaceans whose ranges coincide with and/or overlap with the DWH oil spill footprint. Figure 4.9-1 illustrates the ranges of these stocks and species, and the extent of their coincidence and/or overlap with the spill footprint.

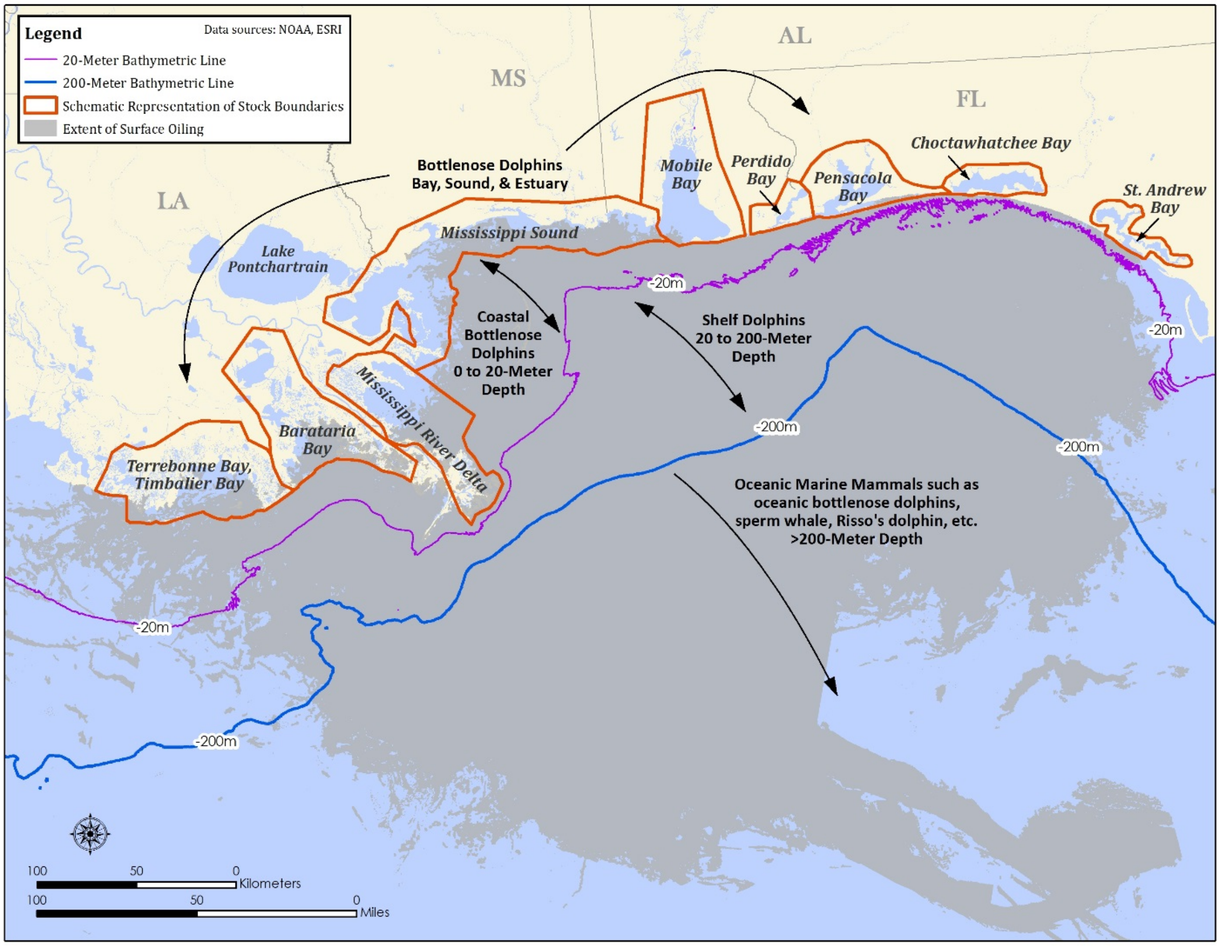


Figure 4.9-1. Thirteen common bottlenose dolphin stocks are found within the cumulative surface oiling footprint from the DWH oil spill, including BSE, coastal, continental shelf, and oceanic stocks. In addition, 18 other oceanic species of marine mammals are found within the oil footprint.

4.9.1.3 Life Histories and Habitats

Cetaceans share many traits, but the 21 species of dolphins and whales in the northern Gulf of Mexico (see Table 4.9-1) have adapted strategies to exploit most of the habitats within the Gulf of Mexico marine ecosystem (Dias & Garrison 2015; Rosel & Mullin 2015). While cetacean behavior can vary greatly, most species generally spend time at the ocean surface to breathe, rest, socialize, and play. When feeding, however, dolphins and whales will swim and dive to various depths throughout the water column, and some feed in sediments on the ocean floor. Cetacean population numbers depend on the quality of habitat, including the quantity and quality of food, weather and environmental conditions, and natural and anthropogenic stressors. Many cetacean species are predators at the top of the food chain (apex predators) that depend on a wide variety of resources within a given habitat. To provide some general description of the physical characteristics, habitat preferences, and life history characteristics of northern Gulf of Mexico cetaceans, two species are described in detail in the following sections. Additional life history information for the other northern Gulf of Mexico cetacean species can be found in *Cetacean Species in the Gulf of Mexico* (Rosel & Mullin 2015).

4.9.1.3.1 Common Bottlenose Dolphins (*Tursiops truncatus*)

Common bottlenose dolphins are large and robust dolphins, averaging 1.9 to 3.8 meters in length with a substantial size variation across populations—in the Gulf of Mexico, most measure 2.7 meters or less (Würsig et al. 2000).

Shown in Figure 4.9-2, bottlenose dolphins are found worldwide in temperate, subtropical, and tropical waters. They can inhabit deep, oceanic waters; nearshore coastal waters; waters over the continental shelf, and BSE waters. BSE dolphins demonstrate strong site fidelity. For example, tagged bottlenose dolphins in Barataria Bay exhibit a high degree of site fidelity to relatively small home ranges within the bay (Wells 2014; Wells & Balmer 2012; Wells et al. 2014a; Wells et al. 2014b).

Source: *Fotosearch/Getty Images.*

Figure 4.9-2. Common bottlenose dolphins inhabit estuarine, nearshore, and offshore habitats throughout the Gulf of Mexico. Bottlenose dolphins are protected under the MMPA.

While bottlenose dolphins in U.S. waters are not listed as threatened or endangered under the ESA, all bottlenose dolphin stocks are protected under the MMPA, and several stocks in the Gulf of Mexico are listed as strategic under the MMPA. Bottlenose dolphins are at risk from entanglement in nets from commercial and recreational fisheries (e.g., shrimp, menhaden, and blue crab); therefore, researchers monitor injuries and deaths from fishery interactions and implement mitigation whenever possible. Bottlenose dolphins are also at risk from illegal feeding and harassment, pollutants, habitat loss and degradation, and intentional harm/injury. However, total human-caused mortality and serious injury is unknown for many stocks. The northern and western coastal stocks are considered strategic due to an ongoing UME (described in Box 1). Certain Gulf of Mexico BSE stocks are also listed as strategic due to unknown (but likely small) stock sizes, as well as potential impacts from the ongoing UME affecting stocks along the coasts of Louisiana, Mississippi, Alabama, and western Florida (Waring et al. 2013).

Box 1: Unusual Mortality Events and the Northern Gulf UME

The MMPA defines an unusual mortality event (UME) as an event that involves a collection of stranded marine mammals that is unexpected, involves a significant die-off of any marine mammal population, and demands an immediate response (16 U.S.C. 1421h). There are seven criteria used to determine if a mortality event is "unusual" (http://www.nmfs.noaa.gov/pr/health/mmume/criteria.htm). UMEs are declared based upon comparisons of a stranding event with historical data, review and recommendation by a federally appointed Working Group for Marine Mammal Unusual Mortality Events (UME Working Group), and input from the NOAA National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS).

From February through April 2010, there was an increase in the number of bottlenose dolphin strandings along the northern Gulf of Mexico coastline: 114 strandings compared to the historical average of 37 for this area during the same time period (http://www.nmfs.noaa.gov/pr/health/mmume/cetacean_gulfofmexico.htm). The increase was particularly notable within Lake Pontchartrain, Louisiana, which had 26 strandings. In March 2010, NMFS consulted the UME Working Group to investigate the increased strandings in the northern Gulf of Mexico, largely motivated by the bottlenose dolphin mortalities in Lake Pontchartrain. The Working Group requested that NMFS reevaluate all cetacean strandings and resubmit their request for consultation.

This consultation with the UME Working Group was subsequently put on hold when the DWH well exploded on April 20, 2010, causing NMFS staff and the stranding network to focus on the crisis at hand and support marine mammal response efforts. After the DWH response phase for cetaceans ended, consultation was reinitiated. On December 13, 2010, the UME Working Group concluded that the high number of cetacean mortalities in the northern Gulf of Mexico in 2010 met the criteria for a UME, or perhaps multiple UMEs. The UME investigation has lasted more than 5 years and, as of August 2, 2015, includes over 1,400 cetacean strandings, of which 86 percent are bottlenose dolphins. Most (94 percent) of the animals stranded dead.

Although the current UME started prior to the DWH incident, most of the strandings (above the historical average) prior to the incident occurred from March to May 2010; were limited to Lake Pontchartrain, Louisiana, and western Mississippi; and were most likely caused by prolonged exposure to cold temperatures and low salinity. Most strandings outside of this March to May 2010 cluster occurred after the DWH blowout, were focused in areas exposed to DWH oil, and could not be attributed to prolonged cold temperatures or low salinity (Litz et al. 2014; Mullin et al. 2015; Venn-Watson et al. 2015c).

Generally females reach sexual maturity between 5 and 13 years of age and males between 9 and 14 years of age (Wells & Scott 2009). Females typically give birth every 3 to 6 years (Wells & Scott 2009). Researchers have identified females up to 57 years old and males up to 48 years old (Wells & Scott 2009).

Prey preferences for bottlenose dolphins are highly variable and depend on their habitat. BSE and coastal animals eat primarily fish (e.g., drums, mullets, and tuna) and some shrimp and crab, while the oceanic animals typically eat fish and squid (Wells & Scott 2009; Würsig et al. 2000; Wynne & Schwartz 1999). Bottlenose dolphins exhibit adaptable feeding behaviors and many different foraging strategies (Krützen et al. 2005). For example, some dolphins root in the sand for submerged prey, burying themselves nearly to their pectoral fins (Rossbach & Herzing 1997).

4.9.1.3.2 Sperm Whales (*Physeter macrocephalus*)

The sperm whale is the largest toothed-whale species (Figure 4.9-3). Adult females can reach 11 to 12 meters in length, while adult males are much larger, measuring as much as 16 to 18 meters in length (Jefferson et al. 1993; Whitehead 2009). Researchers have reported that Gulf of Mexico sperm whales are smaller than those from other areas (Jochens et al. 2008).

Source: *James R.D. Scott/Getty Images.*

Figure 4.9-3. Sperm whales inhabit offshore habitats in the Gulf of Mexico. Sperm whales are protected under the MMPA and ESA.

Sperm whales are listed as endangered under the ESA. A Final Recovery Plan for sperm whales was published and is in effect (NMFS 2010). Sperm whales in the Gulf of Mexico are also listed as strategic under the MMPA. The best available abundance estimate for the northern Gulf of Mexico sperm whales is 763 (coefficient of variation [CV] = 0.38) (Waring et al. 2013). The current levels of human-caused mortality and serious injury for this stock are not known (Waring et al. 2013).

Sperm whales are distributed in deep waters worldwide from the ice edge to the equator (Whitehead 2009). The sexes differ in habitat usage with females distributed primarily in tropical and warm-temperate waters, while adult males have larger ranges and may move from the equator to the ice edge (Whitehead 2009). Sperm whales are found year-round in the northern Gulf of Mexico along the continental slope and in oceanic waters (Waring et al. 2013). There are several areas between Mississippi Canyon and De Soto Canyon where sperm whales congregate at high densities, likely because of localized, highly productive habitats (Biggs et al. 2005; Jochens et al. 2008).

Female sperm whales reach sexual maturity at about 8 to 9 years old, and they give birth about every 5 to 7 years; gestation is 14 to 16 months (Whitehead 2009; Würsig et al. 2000). Males are larger and do not start breeding until their late 20s (Whitehead 2009). Sperm whales consume a wide variety of deep water fish and cephalopods; their primary prey is squid. They forage during deep dives that routinely reach depths of 600 meters and last for about 45 minutes (Whitehead 2009), but they are capable of diving to depths of over 3,200 meters for over 60 minutes (Würsig et al. 2000). After a long, deep dive, sperm whales come to the ocean surface to breathe and recover for approximately 9 minutes (Whitehead 2009).

4.9.2 Approach to the Assessment

In order to investigate the DWH oil spill-related injuries to northern Gulf of Mexico cetaceans, scientists navigated a variety of regulatory, ethical, and logistical challenges that are common for research in cetaceans, including their protection under the MMPA. Evidence of causation was derived by integrating studies from the literature with data from DWH oil spill NRDA field studies and laboratory tests, and critically evaluating alternative potential causes of injury.

Key Points

- Researchers used historical sightings data and conducted surveys in the DWH oil spill footprint to determine the number of animals exposed to DWH oil. Biologists and veterinarians determined the potential routes of exposure of DWH oil to the tissues and organ systems of marine mammals, in order to ascertain potential adverse health effects.
- Separate from the NRDA, the federal government declared a marine mammal UME, now the largest and longest UME on record for the northern Gulf of Mexico. Marine mammal scientists from the NRDA worked with the UME investigators to critically evaluate the stranding data to identify the relationship between the increased number of strandings and the DWH oil spill.
- Researchers temporarily captured dolphins living in Barataria Bay, Louisiana, and in Mississippi Sound (Mississippi and Alabama) to collect medical data on individuals in DWH oil-contaminated habitat. They compared their findings to animals living in areas that did not experience DWH oil contamination, as well as to pathology data from dead, stranded dolphins.
- Scientists conducted surveys to compare survival and reproductive success in dolphin populations living within versus outside of DWH-oil contaminated habitat.
- Researchers analyzed their data in the context of the DWH oil spill and other potential drivers of marine mammal UMEs, in order to determine the likelihood that the injuries observed were caused by the DWH spill.
- Marine mammal scientists synthesized data from DWH field studies and the literature to characterize the adverse effects on the studied populations and extrapolate the magnitude of injury to other populations.

4.9.2.1 Rationale

As oil spilled into the Gulf of Mexico and reached coastlines, response workers rushed to contain the inevitable injuries to Gulf of Mexico resources, and researchers quickly mobilized to characterize injuries in real time. Responders, researchers, and media reports released striking images of cetaceans swimming in the DWH surface oil slick (Figure 4.9-4) and a disturbingly large number of dead cetaceans stranded on coastlines affected by the spill.

Although studies on marine mammals following oil spills are limited, both laboratory and field studies, including research conducted in the wake of the *Exxon Valdez* oil spill, have documented the adverse effects of oil to marine mammals and other wildlife species and their habitats (Peterson 2001; Peterson et al. 2003). Because of the widespread distribution of oil across prime cetacean habitats in the northern Gulf of Mexico, the Trustees developed a suite of studies to assess the extent of DWH oil exposure to northern Gulf of Mexico cetaceans and to identify and characterize potential injuries to these animals as a result of the oil spill.

Source: *Mandy Tumlin, Louisiana Department of Wildlife and Fisheries.*

Figure 4.9-4. A bottlenose dolphin with oil residue on its head swims through a DWH oil surface slick in Barataria Bay, Louisiana (August 5, 2010, authority of section 109h MMPA).

4.9.2.2 Known Risks of Oil to Marine Mammals

While data are sparse, both field and laboratory studies have shown that cetaceans exposed to oil can suffer impaired health, and potentially die, as a result of that exposure. Inference about the impacts of oil exposure on the health of cetaceans is more commonly drawn from the results of laboratory studies on the effects of oil in other marine mammals (e.g., pinnipeds, a group of marine mammals which includes seals, sea lions, and walruses) and surrogate mammalian species. Mink are often used as surrogates for other mammals because they are readily raised in captivity; for example, they have been used as surrogates for sea otters because they have a semi-aquatic life style in the wild. The evidence for injury to cetaceans as a result of exposure to oil is briefly summarized below; presented first is evidence from field studies, followed by evidence from laboratory studies of cetaceans, as well as pinnipeds, mink, and other mammals.

There are only a handful of studies that report on the health or survival of cetaceans following oil spills. Most notably, in the 18 months following the *Exxon Valdez* oil spill, one resident pod and one transient pod of killer whales present in Prince William Sound at the time of the spill experienced an unprecedented number of deaths (30 to 40 percent mortality; Matkin et al. 2008). None of the killer whale carcasses were recovered. As of 2012, NOAA concluded that the pod of resident killer whales still had not reached its pre-spill numbers, while the oil-exposed transient pod numbers have continued to decrease– so much so that they have been listed as a “depleted stock” under the MMPA. Meanwhile, other killer whale populations in southeast Alaska have grown since the mid-1980s (Matkin et al. 2008).

In addition, following the *Exxon Valdez* oil spill, 37 carcasses of other cetaceans were found, which represented the largest number of cetacean strandings ever observed in the region. The cause of death of these stranded animals could not be determined, and the extent to which increased vessel activity might have contributed to increased observations of stranded cetaceans is not known. There is also one report of gray whales showing altered respiratory behavior (increased blow rates) in the presence of surface oiling off the coast of California (Geraci & St. Aubin 1982; Geraci & St. Aubin 1985). A small number of studies have exposed cetaceans to oil (reviewed in Englehardt 1983). Effects from these exposures included the following:

- Liver damage in captive bottlenose dolphins that had oil added to their tank.
- Skin lesions in a number of captive delphinid species where oil was applied to their skin.
- Skin lesions after oil was applied to the skin of a live, stranded sperm whale.

Additional studies in which pinnipeds were exposed to oil via ingestion, inhalation, or application to their fur have shown a wide range of effects, including lung inflammation, increased respiratory rates, respiratory failure, abnormal nervous system function, liver and kidney damage, reproductive impairment, and death (reviewed in Englehardt (1983).

Controlled oil exposure studies with mink documented liver, adrenal, and hematological effects over several months (Mazet et al. 2000; Schwartz et al. 2004). Exposed pregnant mink also had a decrease in the number of live-born offspring (Mazet et al. 2000; Mazet et al. 2001). Subsequent studies confirmed findings of adrenal effects and also determined that the adrenal stress response was impaired in mink chronically exposed to oil (Mohr et al. 2008).

In summary, while data on the effects of oil exposure in cetaceans are scarce, available data point towards the possibility of injury and death following oil spills. Findings from experimental studies in pinnipeds and mink (as a surrogate for marine mammals) also suggest that cetaceans exposed to oil will likely experience adverse health effects and possibly death.

4.9.2.3 Conceptual Model and Studies to Support the Assessment

The Trustees developed a general conceptual model to support the injury assessment (Figure 4.9-5). In order to assess injuries to marine mammals due to the DWH oil spill, the Trustees evaluated the pathways and exposures of marine mammals to DWH oil, characterized the injuries (including death, reproductive failure, and other adverse health effects) associated with DWH oil exposure, and quantified the magnitude of those injuries across northern Gulf of Mexico marine mammal populations. The following subsections describe the Trustees' approach to the assessment, including how scientists across the NRDA-designed studies analyzed NRDA and non-NRDA datasets, and synthesized the results with the published literature, in order to determine marine mammal exposure and injury due to the DWH oil spill.

Figure 4.9-5 illustrates the Trustees' approach.

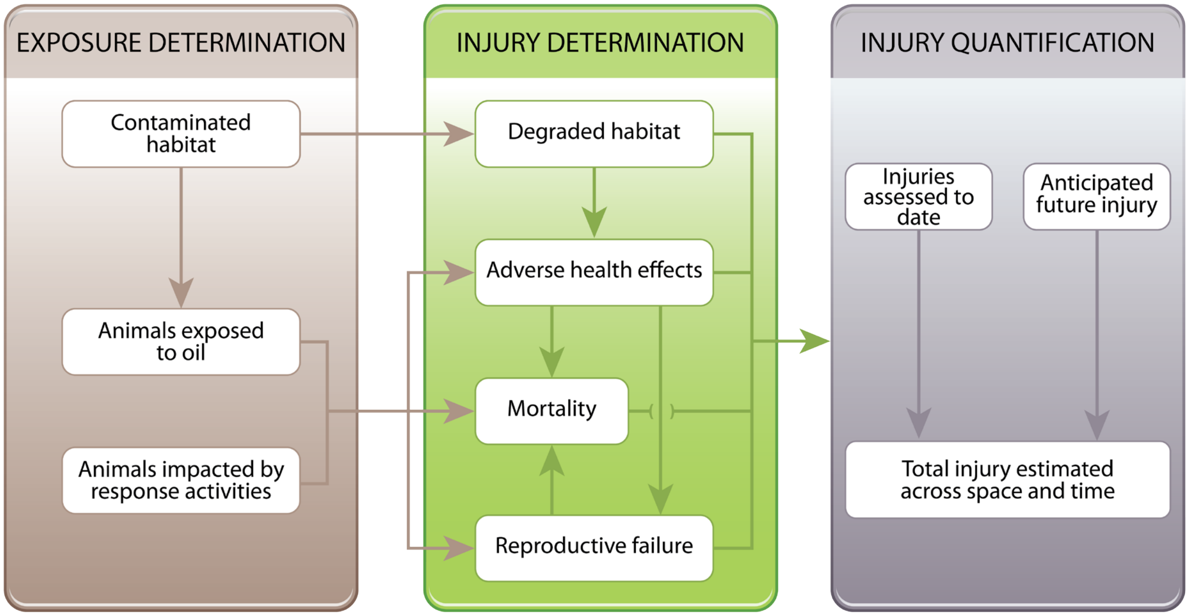


Figure 4.9-5. The Trustees quantified injuries to northern Gulf of Mexico marine mammal populations based on the widespread exposure of marine mammals to DWH oil and the observed injuries documented in the field.

4.9.2.3.1 Exposure Determination

In order to determine the exposure of marine mammals to DWH oil, the Trustees considered the following:

- Biologists conducted aerial and vessel surveys to document and estimate the number of marine mammals within the DWH oil spill surface slick. Researchers also compared marine mammal population distributions and behaviors (based on tracking animals with radio and satellite tags, acoustic tracking of oceanic animals, aerial and vessel-based surveys, and historical data) to the transport of DWH oil (see Section 4.2, Natural Resource Exposure).
- DWH responders reported observations of marine mammals, living or dead, in the surface slick. Stranding response teams collected photographs of any stranded animals with oil on their skin and swabs of the oil for chemical analysis. Some internal tissues (e.g., lung tissue) were also analyzed for DWH oil components.
- Biologists and veterinarians identified and characterized the potential routes of exposure, including inhalation, aspiration, aspiration of oil-induced vomitus, ingestion, and dermal absorption, for DWH oil to various marine mammal tissues and organs. Scientists also identified literature that described the adverse health effects when test organisms encountered oil via these exposure routes.

4.9.2.3.2 Injury Determination

In order to determine the injuries to marine mammals as a result of exposure to DWH oil, the Trustees considered the following:

- In collaboration with researchers studying the ongoing northern Gulf of Mexico marine mammal UME (Box 1), NRDA scientists studied trends in stranding rates over time and geographic space. For example, although the current UME started prior to the DWH incident, most of the strandings (above historical averages) prior to the incident were limited to Lake Pontchartrain, Louisiana, and western Mississippi, and were most likely caused by prolonged exposure to cold temperatures and low salinity. In contrast, most strandings outside of this March to May 2010 cluster occurred after the DWH blowout, were focused in areas exposed to DWH oil, and could not be attributed to prolonged cold temperatures or low salinity (Mullin et al. 2015; Venn-Watson et al. 2015c).

- Veterinary pathologists analyzed tissues from stranded carcasses to understand the health of each animal leading up to its death and potential causes of death (whenever possible, depending on decomposition rates). They compared the results to findings in dolphins that stranded in other areas of the southeastern United States that were unaffected by the DWH oil spill (Venn-Watson et al. 2015a).

- Marine mammal scientists characterized the survival and reproductive success of populations in Barataria Bay, Louisiana, and Mississippi Sound in Mississippi and Alabama (two areas with documented DWH oil contamination) using a combination of photo-identification surveys, mark and recapture analysis, and pregnancy determination via ultrasound or blubber hormone levels.

- In addition to measuring dolphin density/abundance and reproduction, veterinarians and biologists also captured

Box 2: Sarasota Bay and Other Reference Populations

The Trustees compared mortality rates, reproductive parameters, and health indicators for bottlenose dolphins in areas of the northern Gulf of Mexico affected by the DWH oil spill to data from well-studied bottlenose dolphin stocks from other BSEs in the southeastern United States. Researchers have studied bottlenose dolphins extensively at several BSE sites and have documented a common basis of biology, behavior, ecology, and health across sites (Reynolds III et al. 2000; Wells & Scott 1990).

The Trustees used information on mortality and reproduction available from reference sites in Texas, Mississippi, Florida, and South Carolina (Fernandez & Hohn 1998; Mattson et al. 2006; Stolen & Barlow 2003; Wells & Scott 1990).

The Trustees compared dolphin health data with data from Sarasota Bay, Florida, where scientists have conducted health studies for decades, as well as with health data from other long-term studies or repeated, standardized health assessments on BSE dolphins near St. Joseph Bay, Florida; Charleston, South Carolina; Beaufort, North Carolina; and Indian River Lagoon, Florida (Schwacke et al. 2009; Schwacke et al. 2010; Wells et al. 2004).

and released live animals in Barataria Bay and Mississippi Sound to conduct health assessments. They analyzed a suite of medical parameters, including pulmonary health, hormone levels, body condition, blood chemistry, and reproductive status, among many others. Data from these studies were compared to the results from dolphins in Sarasota Bay, Florida, and other similar habitats in the southeastern United States that were unaffected by the DWH oil spill (Schwacke et al. 2014; Smith et al. 2015) (Box 2).

- Scientists specifically designed their studies and analytical methods to account for other factors, such as biotoxins from harmful algal blooms, infectious disease outbreaks, human/fishery interactions, environmental factors, and other chemical contaminants. Whenever possible, researchers evaluated the plausibility, specificity, consistency, and strength of association between the observed adverse effects and oil exposure (or other potentially harmful activities associated with the response to the DWH oil spill). This provided the scientific basis for establishing causality (Venn-Watson et al. 2015b).

- The Trustees found it unrealistic to investigate the health of marine mammals across the entire area of the northern Gulf of Mexico affected by the DWH oil spill. Instead, scientists focused their health assessments on dolphin populations in Barataria Bay and Mississippi Sound to 1) examine potentially sublethal effects in cetaceans exposed to DWH oil, and 2) use these BSE populations as case studies for extrapolating to coastal and oceanic populations that received similar or worse exposure to DWH oil. In addition, marine mammal biologists could compare the injuries identified in health assessments and photo-identification surveys with the pathology findings and statistics from the dead, stranded dolphins in the ongoing UME investigation.

4.9.2.3.3 Injury Quantification

In order to quantify the injuries to marine mammals as a result of exposure to DWH oil, the Trustees considered the following:

- Scientists used data from stranded animals, photo-identification surveys, and live dolphin health assessments to characterize the adverse effects within the observed populations in Barataria Bay and Mississippi Sound, and extrapolated the magnitude of the injury to other populations present within the oil footprint (DWH MMIQT 2015).

- Marine mammal biologists used models that synthesized the extent of oiling over the geographic area and timeline of the spill, the likelihood of lingering adverse health and reproductive effects, and the specific population dynamics of each cetacean species to characterize the effect of the DWH oil spill on cetacean populations.

4.9.2.4 Summary

The Trustees developed a suite of laboratory and field studies to determine the extent of exposure across the northern Gulf of Mexico, characterize the types of injuries suffered by marine mammals, and quantify the total injuries across the marine mammal stocks in the Gulf of Mexico. The results of those studies are presented in the following sections, with interpretations informed by non-NRDA studies and the pre-existing literature.

4.9.3 Exposure

Key Points

- The Trustees have determined that marine mammals were exposed to chemical contaminants resulting from the DWH oil spill in the northern Gulf of Mexico.
- DWH oil contaminated every type of habitat that northern Gulf of Mexico marine mammals occupy.
- During response activities and surveys, workers observed over 1,400 marine mammals in the DWH surface slick. The Trustees estimate that tens of thousands of marine mammals were exposed to DWH oil based on population abundances and the extent of the oil footprint.
- Chemists identified DWH oil on the skin of dead, stranded dolphins and also found constituents consistent with what would be in the breathing zone vapor above oil slicks in the lung tissue of one dead, stranded dolphin.
- Cetaceans were likely exposed to DWH oil via inhalation of contaminated air and/or aspiration of liquid oil. These routes of exposure are consistent with the types of adverse health effects documented in living and dead, stranded dolphins (e.g., effects on the lungs).
- Cetaceans may also have been exposed to DWH oil via ingestion of contaminated sediment, water, or prey, or dermal absorption after contact with contaminated water or sediment.

Source: NOAA. Photo taken under NMFS permit.

Figure 4.9-6. Stenellid dolphins swim through oil on April 29, 2010.

The Trustees have determined that marine mammals were exposed to chemical contaminants resulting from the DWH oil spill in the northern Gulf of Mexico. As demonstrated in this section, as well as in Section 4.2, Natural Resource Exposure; Section 4.4, Water Column; and Section 4.6, Nearshore Marine Ecosystem, sufficient amounts of oil were present, and persisted, in the oil spill footprint to expose marine mammals and their supporting habitats. DWH oil impacted over 112,000 square kilometers of the ocean surface, over 2,100 kilometers of shoreline, at least 1,000 square kilometers of the deep-sea floor, and plumes of deep ocean water that extended over 400 kilometers from the wellhead. While the surface oil and plumes dissipated following the capping of the wellhead, contaminated sediments persisted at least into 2014 (Hayworth et al. 2015) (Section 4.2, Natural Resource Exposure). Critical pathways of exposure include the contaminated water column, where marine mammals swim and capture prey; the surface slick at the air:water interface, where marine mammals breathe, rest, and swim; and contaminated sediment, where marine mammals forage and capture prey.

***Source:** NOAA. Photo taken under NMFS permit.*

Figure 4.9-7. A group of rough-toothed dolphins swim through thick oil offshore on June 16, 2010.

4.9.3.1 Overlap of Marine Mammal Populations and the Surface Oil Footprint

Whether in the area contaminated by the deep-sea plume, at the surface, or in BSE habitats, a variety of cetacean species rely upon the habitat and resources within the DWH oil spill footprint, as shown in Figure 4.9-8 (Dias 2015; Jefferson & Schiro 1997; Waring et al. 2013). Population distributions (as defined by tracking with radio or satellite tags or via acoustic monitoring, aerial and vessel surveys, and historical survey data) demonstrated that the DWH oil spill footprint overlapped with the known ranges of 31 stocks of northern Gulf of Mexico marine mammals (Dias 2015; Waring et al. 2013).

During the spill, response workers and Trustees documented, photographed, and recorded videos of marine mammals present in areas contaminated by oil ranging from light sheen to thick, heavy oil (Dias 2015). Figure 4.9-4, Figure 4.9-6, and Figure 4.9-7 document some of these animals swimming in oil. Between April 28 and September 2, 2010, the Trustees conducted marine mammal surveys in the northern Gulf of Mexico and around the DWH oil spill site. Vessel and aerial marine mammal surveys, as well as reports from response monitoring activities, documented nearly 1,400 marine mammal sightings of at least 11 cetacean species swimming in oil (Table 4.9-2). In addition to the documentation of direct exposure, additional exposure to oil was estimated by overlapping the marine mammal sightings with the oil footprint. A total of 510 cetacean sightings with over 6,400 animals overlapped with the oil footprint between April 28 and August 10, 2010 (Figure 4.9-8) (Dias 2015). In addition, between May 2010

Table 4.9-2. Response workers and scientists observed nearly 1,400 marine mammals swimming in DWH surface oil or with oil on their bodies during response activities and as part of NRDA boat and helicopter surveys (Dias 2015).

Species	Number of Occurrences	Number of Individuals
Atlantic spotted dolphin	1	71
Clymene dolphin	1	1
Common bottlenose dolphin	35	329
Cuvier's beaked whale	1	1
Pantropical spotted dolphin	3	205
Pygmy sperm whale	1	2
Risso's dolphin	3	127
Rough-toothed dolphin	4	75
Sperm whale	19	33
Spinner dolphin	2	283
Striped dolphin	2	130
Unidentified dolphin	10	130
Unidentified mammal	3	7
Total	**85**	**1,394**

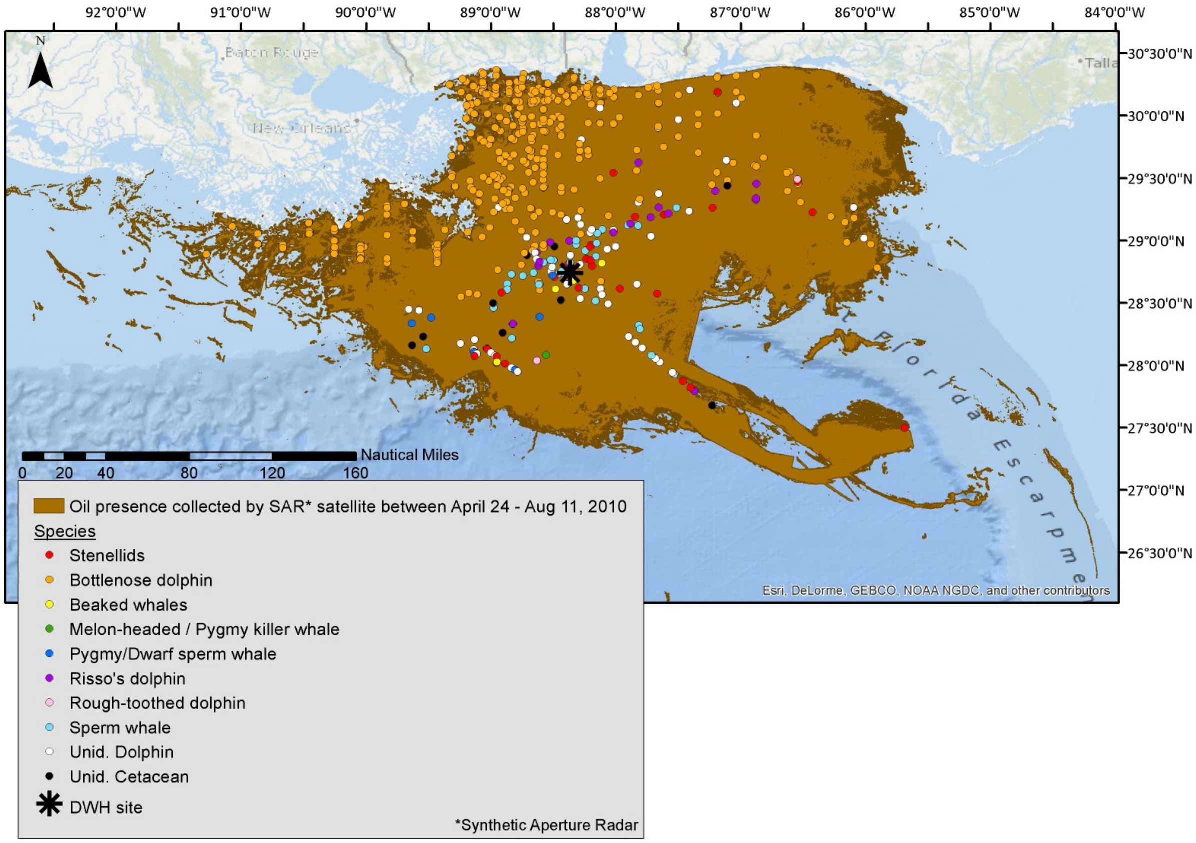


Figure 4.9-8. Marine mammals of at least eight cetacean groups were sighted within the DWH oil spill footprint during vessel and aerial surveys between April 28 and August 10, 2010. Individual points can represent more than one animal. Surveys continued through September 2, 2010, but those data are not shown here.

and as late as February 2012, 14 stranded marine mammals were found with DWH oil on their skin (as confirmed by analytical chemistry) (Stout 2015a).

4.9.3.2 Routes of Exposure

Due to the long timeframe and expansive area of the DWH oil spill, marine mammals may have experienced different exposures depending on their intersection(s) with the oil transport pathways. Oceanic animals near the wellhead were likely exposed to fresher oil for longer periods of time, while BSE bottlenose dolphins were exposed to intermittent, but lingering, doses of more weathered oil (see Section 4.2, Natural Resource Exposure). Near the wellhead, rising oil was relatively fresh and contained a wide range of toxic components, including polycyclic aromatic hydrocarbons (PAHs, e.g., naphthalene, phenanthrene, and benzo[a]pyrene). In the nearshore, however, oil could have spent days to weeks in the subsurface mixing zone and on the surface before moving into the nearshore environment and exposing the BSE bottlenose dolphins. This oil may have contained a more weathered PAH profile (Stout 2015b, 2015c). Thus, marine mammals would have experienced a heterogeneous set of exposure scenarios.

Marine mammals encountering DWH oil would have experienced multiple routes of exposure that could result in injuries through inhalation, ingestion, ingestion leading to secondary aspiration, direct aspiration, and dermal absorption. Response workers and scientists routinely saw marine mammals swimming in surface oil both offshore and nearshore, like the rough-toothed dolphins pictured in Figure 4.9-7. These animals would likely have inhaled contaminated air, ingested contaminated water, aspirated liquid oil, and contacted oil with their skin and mucus membranes.

4.9.3.2.1 Exposure Through Inhalation

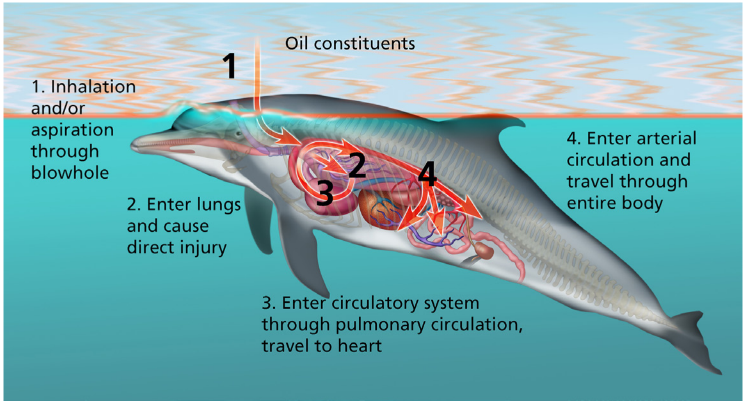
When cetaceans surfaced to breathe within the footprint of the DWH oil surface slick, they would have likely inhaled toxic volatile and aerosolized oil components (see Section 4.2, Natural Resource Exposure). When taking a breath, cetaceans break the surface of the water and exhale/inhale through their blowhole near the air:water interface. If they are surfacing after a long dive, they may breathe at the surface for a long period of time in order to replenish their oxygen supply. During long dives, oceanic marine mammals may retain inhaled toxic chemicals for as long as an hour, allowing for long exposures to lung tissue or absorption into the blood when their lungs collapse, a typical physiological response during long dives (Piscitelli et al. 2010; Ponganis 2011).

The chemical components of oil will escape from a surface slick (at various rates depending on the compound and its concentration) and become available to cetaceans in a variety of forms. In surface slicks, they may escape as volatile organic compounds (VOCs), intermediate volatile organic compounds (iVOCs), or semivolatile organic compounds (sVOCs) (de Gouw et al. 2011; Stout 2015a) and be suspended in the air column (de Gouw et al. 2011; Haus 2015; Murphy et al. 2015). Chemical components may also enter small seawater droplets that can become indefinitely suspended in the air column due to the breaking of waves, wind, raindrops, animals breaking the surface, or other disruptions to the air:water interface (primary aerosols). Finally, volatiles and particles in the air column can undergo chemical transformations and coalesce to form suspended particulates (secondary aerosols) (de Gouw et al. 2011; Haus 2015; Murphy et al. 2015).

Inhalation exposures were a concern for any organism near the DWH surface slick, including response workers, birds, sea turtles, and marine mammals. For health and safety purposes, DWH spill responders were required to wear dosimetry badges to document their potential inhalation exposure to oil compounds. Chemical analyses of these badges demonstrated that some of these workers inhaled oil compounds and suffered adverse respiratory effects (Groth et al. 2014; Ramachandran et al. 2014; Sandler et al. 2014; Stenzel et al. 2014).

Inhaled contaminant exposure and lung injury to cetaceans were likely amplified compared to other mammals and wildlife because cetaceans breathe at the air:water interface where volatile contaminants would be at their highest concentrations; lack nasal structures that filter air prior to reaching the lung; have deep lung air exchange (80 to 90 percent of lung volume compared to 10 to 20 percent for humans); have extended breath hold time; and have an extensive, rich blood supply in their lungs (Green 1972; Irving et al. 1941; Ridgway 1972; Ridgway et al. 1969). As shown in Figure 4.9-9, all of these factors would facilitate the transport of inhaled contaminants into the blood, directly to the heart, and then throughout the body, without first going through the liver (which is a major organ involved in metabolizing toxicants absorbed via ingestion).

The Trustees were able to directly document inhalation exposure to DWH oil vapors in a deceased bottlenose dolphin stranded on May 24, 2010, near Barataria Bay. This dolphin had DWH oil on its exterior, and its lung tissue contained volatile petroleum constituents. The lung tissue data strongly suggest that the dolphin inhaled DWH oil vapors prior to death (Stout 2015a). Researchers ruled out aspiration of liquid or aerosolized oil due to the absence of nonvolatile PAHs and biomarkers in the lung tissue (see Section 4.2, Natural Resource Exposure).

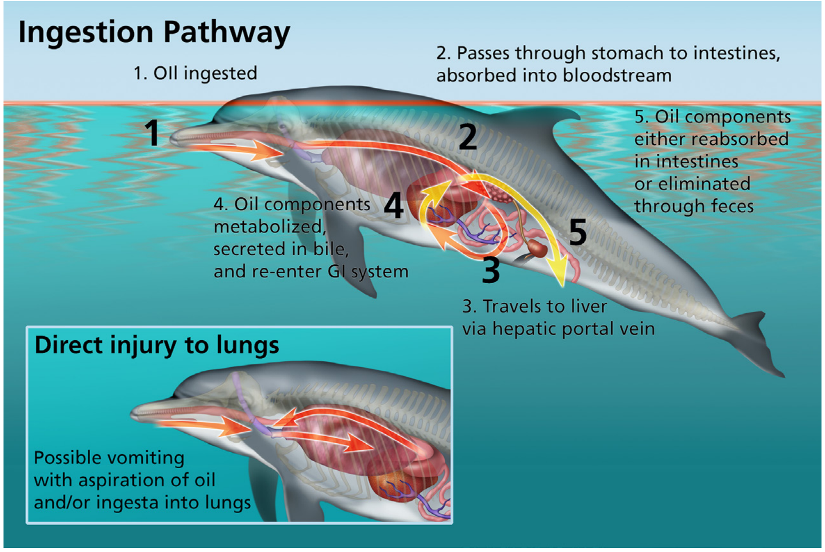


Source: *Kate Sweeney for NOAA.*

Figure 4.9-9. Chemical components become available to cetaceans through inhalation and aspiration exposure pathways.

4.9.3.2.2 Exposure Through Ingestion

Depending on the species, habitat, and prey, marine mammals can use many different feeding techniques, from baleen whales straining large volumes of water for krill, to bottlenose dolphins foraging in shallow, turbid BSE waters. In the presence of oil-contaminated resources, it is inevitable that animals will inadvertently ingest oil from water, sediment, or prey. Some specific examples of such ingestion include suction feeding, where animals quickly suck water and prey into their mouths, and crater feeding, where dolphins burrow into the sediment with their rostrum (sometimes as far as their pectoral fins) in search of submerged prey (Rossbach & Herzing 1997). Contaminated water, sediment, and prey will transit from the oral cavity to the esophagus and the rest of the gastrointestinal tract, where oil can cause mucosal irritation, vomiting, and regurgitation (Edwards 1989; Rowe et al. 1951) (see Section 4.7, Birds, and Section 4.8,



Source: *Kate Sweeney for NOAA.*

Figure 4.9-10. Cetaceans become exposed to chemicals through ingestion of contaminated water, soil, or prey. After ingestion, some animals can become nauseous and vomit, presenting an opportunity for aspiration of oil-contaminated oil and/or ingesta into the lungs.

Sea Turtles). Figure 4.9-10 illustrates these ingestion pathways. Oil components can transit to the bloodstream (and then to the rest of the body) from the gastrointestinal tract, but chemicals that make it to the liver will likely be metabolized. Ingested toxicants will affect a variety of organs, including the adrenal glands and the reproductive tract, but the detoxification pathway associated with ingestion is likely to reduce any potential effect to the lungs except through vomiting and aspiration. Bodkin et al. (2012) reported long-term effects from the *Exxon Valdez* oil spill on sea otters *(Enhydra lutris)* in Alaska. The pathway for these long-term effects was attributed to oil exposure via ingestion during intertidal foraging and the presence of oil near otter foraging pits (the authors ruled out exposure by inhalation).

Animals that ingest petroleum are likely to experience nausea and vomiting; after vomiting, they are at risk of aspirating the vomitus (a collection of food, petroleum, and stomach acids) into the lungs (Coppock et al. 1995; Coppock et al. 1996; Lifshitz et al. 2003; Siddiqui et al. 2008). The aspiration of vomitus results in pneumonias that can eventually lead to lung abscesses and secondary infections (Coppock et al. 1995; Coppock et al. 1996). In regions of the world where children frequently mistake bottles of kerosene for water during hot summer months, numerous reports have documented that kerosene ingestion leads to aspiration pneumonia (Lifshitz et al. 2003; Sen et al. 2013; Siddiqui et al. 2008). Although aspiration pneumonia is considered rare in dolphins, a recent study from stranded animals in the northern Gulf of Mexico has documented aspiration pneumonia (Venn-Watson et al. 2015a).

4.9.3.2.3 Exposure Through Aspiration

When cetaceans surface, some water remains around the blowhole, or water droplets may be splashed onto the blowhole just prior to or during a breath. In oil-contaminated waters, when marine mammals surface to breathe, they can aspirate (i.e., draw in liquid via suction) oily water or droplets directly into their blowhole, through the larynx and trachea, and potentially into their lungs. Figure 4.9-11 shows a spray of oily water droplets from exhalation, which could lead to the animal's aspiration of oily water. Aspiration of liquid oil results in physical injuries, where oil damages tissues and membranes along the respiratory tract and lungs. Exposure to toxic oil components will results in chemical injury to tissues, as well as delivery of oil components to the blood, and then throughout the body (Coppock et al. 1995; Coppock et al. 1996; Prasad et al. 2011). Aspiration of petroleum products in cattle most commonly leads to a severe inflammatory response and lung disease, including pneumonia, fibrosis, and pulmonary dysfunction (Coppock et al. 1995; Coppock et al. 1996).

Source: *NOAA. Taken under NMFS permit.*

Figure 4.9-11. A rough-toothed dolphin surfaces through an oil slick during the DWH oil spill (June 16, 2010). The spray of oily water droplets is generated from the animal's exhalation, leading to the potential for inhalation of contaminated aerosols and aspiration of oily water.

4.9.3.2.4 Exposure Through Dermal Contact

Cetaceans have developed a thick epidermis that prevents absorption and leakage, which is critical in a high-salinity environment. Their thick skin is thought to protect against oil absorption. However, if DWH oil-exposed animals had any skin lesions, cuts, rake marks, or abrasions, or were in low-salinity water for a period of time, oil could be absorbed more readily and delivered into the bloodstream. In addition, in areas of skin ulceration, direct exposure to oil could damage the exposed tissues. Oil can also irritate and erode mucous membranes, for example, in the eyes and mouth (Dutton 1934; Hansborough et al. 1985)

4.9.3.3 Summary

DWH oil contaminated marine mammal habitats throughout the northern Gulf of Mexico, resulting in the exposure of 31 stocks of dolphins and whales. Cetaceans would have been exposed to toxic oil components through inhalation, aspiration, ingestion, and dermal exposure. Similar routes of exposure have been characterized in field and laboratory studies of a variety of species and resulted in physical and toxicological damage to organ systems and tissues, reproductive failure, and death. However, unlike in laboratory exposures, cetaceans exposed in the northern Gulf of Mexico would have suffered from multiple routes of exposure at the same time, over intermittent timeframes and at varying rates, doses, and chemical compositions of oil, thus complicating the severity and combinations of injuries.

4.9.4 Injury Determination

Key Points

- The Trustees determined that exposure to DWH chemical contaminants resulted in death, reproductive failure, and adverse health effects in northern Gulf of Mexico marine mammal populations.
- The adverse health effects include lung disease, adrenal disease, and poor body condition. Other factors contributing to poor health include tooth loss, anemia, and liver injury.
- In addition to injuries from direct exposure to DWH oil, marine mammal habitat was degraded.
- Marine mammals were affected by DWH oil spill response activities.
- The Trustees ruled out other potential causes of the observed injuries and have concluded that the impacts from the DWH oil spill are the only reasonable cause for the suite of observed adverse health effects and the patterns of observed reproductive failure and mortalities.

The Trustees have determined that exposure to DWH chemical contaminants resulted in death, reproductive failure, and adverse health effects in northern Gulf of Mexico marine mammal populations. The determination is based on the integrated analysis of field-based studies of marine mammal health and reproduction, Gulf-wide stranding numbers, histopathology analysis of tissues from dead dolphins, and oil toxicity literature from field and laboratory studies. The nature of the adverse health effects in living marine mammals, the lesions identified in stranded marine mammals, and the spatial extent and timing of marine mammal injuries are consistent 1) across the DWH marine mammal injury determination studies; 2) with known toxic effects described in the literature and Section 4.3, Toxicity; and 3) with the timing and pathway of oil and chemical contaminants during and after the DWH oil spill.

Oil toxicity exerts a variety of negative effects on marine mammals (Geraci & St. Aubin 1990; Loughlin 2013), including molecular, cellular, and developmental effects; tissue and organ malfunction; reproductive failure and death; impacts on population dynamics; and secondary effects from the oil's harm on critical marine mammal habitats. Oil is a complex chemical mixture, and oil toxicity will likely manifest differently in each individual. In laboratory and field studies with a variety of taxa (including fish, birds, invertebrates, and turtles), DWH oil exposure resulted in anemia, immunosuppression, growth inhibition, hormone dysregulation, reproductive failures, and many other adverse effects, often in various combinations depending on dose, species, and other factors (see Section 4.3, Toxicity). In this section, the Trustees describe the marine mammal injuries from the DWH spill and discuss how the adverse effects to each tissue or organ can influence marine mammal fitness and survival.

While marine mammals in the northern Gulf of Mexico face a variety of pathogens, environmental insults, and anthropogenic stressors on a routine basis, the likely cause of the suite of adverse effects described in this section is exposure to DWH oil (Venn-Watson et al. 2015b). Marine mammal scientists and veterinarians designed field studies and conducted data analysis so as to explicitly examine other potential explanations for marine mammal injuries, including biotoxins from harmful algal blooms, infectious diseases that have been implicated in previous marine mammal UMEs, human and fishery interactions, and other potential contaminants unrelated to the DWH oil spill (Venn-Watson et al. 2015b). Based on the results of these analyses, the scientists ruled out each of these other factors as a primary cause for the high prevalence of adverse health effects, reproductive failures, and disease in stranded animals. When all of the data are considered together, the DWH oil spill is the only reasonable cause for the full suite of observed adverse health effects.

4.9.4.1 Effects on Habitat

Marine mammals live for decades and use a wide range of behaviors and feeding strategies to exploit available resources. Oiled marine mammal habitats in the northern Gulf of Mexico include the shallow embayments with fringing wetlands and beaches, coastal and shelf waters, and oceanic habitats including the deeper waters used by deep divers such as sperm whales. Thus, any detrimental impacts on their habitat will most likely have some negative effect on their typical behavior, especially as interpreted through the lens of the MMPA (which prohibits the "disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering"). Herbivorous/planktivorous marine mammals will suffer from the effects of DWH oil on seagrass and plankton populations (see Section 4.3, Toxicity, and Section 4.4, Water Column). Carnivorous cetaceans such as dolphins and sperm whales, which are typically apex predators, will suffer from DWH oil's effects on fish and invertebrate populations. At a more subtle, but still crucial, level, the summed negative effects of the DWH oil spill on the Gulf of Mexico ecosystem across resources, up and down the food web, and among habitats, will especially impact marine mammals due to their long lives and their strong dependence on a healthy ecosystem (Bossart 2011; Moore 2008; Reddy et al. 2001; Ross 2000; Wells et al. 2004).

4.9.4.2 Response Injury

In addition to direct impacts to habitat, response activities (including removal of oil from the environment, use of dispersant, response vessel activity, and protection of shorelines and other critical areas) may have had unintended negative consequences for marine mammal habitats and marine mammals themselves. Oceanic animals that were proximate to response activities would have been at risk for:

- Breathing in smoke from burning surface slicks (enriched in the higher molecular weight pyrogenic PAHs and particulate matter).
- Blocked habitat from skimming and burning operations.
- Disturbance by increased vessel traffic and noise.
- Increased bioavailability and mobilization of PAHs by dispersant application.
- Direct breathing and dermal exposure to aerially applied dispersants.
- Increased inhalation and aspiration of dispersed oil at the air:water interface.
- Increased noise from seismic and other assessment or drilling activities near the wellhead.

For deep-diving whales, subsurface dispersant application at the wellhead likely had negative impacts on prey resources and habitat, and may have led to incidental ingestion of dispersant.

Similarly, response activities adversely affected BSE and coastal dolphin habitat, including:

- Disturbance of shallow feeding and resting habitat by containment boom (including boom deployment, stranded boom, and boom maintenance).
- Introduction of dispersants to the habitat (particularly off Grand Isle and Chandeleur Sound, Louisiana, but also in Barataria Bay, where a pilot was forced to release a full load of dispersant during an emergency).
- Increased vessel traffic and noise associated with response operations.

BSE and coastal animals were also at increased risk of entanglement and entrapment in boom and sampling gear.

Overall, the large and varied efforts initiated in response to the spill impacted marine mammals and their habitats and may have contributed to injury.

4.9.4.3 Mortality from Oil Exposure

Starting in 2010, the number of stranded cetaceans (primarily bottlenose dolphins) along the coastlines of Louisiana, Mississippi, and Alabama increased dramatically (Litz et al. 2014; Venn-Watson et al. 2015c). The ongoing, high rate of dead, stranded animals in the region prompted a federally declared northern Gulf of Mexico UME (see Box 1; Litz et al. 2014). The long duration, high number of strandings, and large area affected suggest that the overall event could include multiple overlapping incidents or one incident with multiple contributing factors. To investigate the potential lethal effects of the DWH oil spill on northern Gulf of Mexico marine mammals, researchers analyzed the stranding data in the context of DWH oil transport and conducted surveys to track the survival of animals in BSEs exposed to DWH oil(Lane et al. In Press). Based on the combined results of these studies, the Trustees have determined that the DWH oil spill resulted in the death of marine mammals and is the primary underlying cause of the persistent, elevated stranding numbers and reduced survivorship in the northern Gulf of Mexico.

More than 1,000 cetaceans stranded in Alabama, Mississippi, and Louisiana from 2010 to 2014 (an average of more than 200 per year) compared to an average of 54 per year prior to 2010 for the same region (Figure 4.9-12).

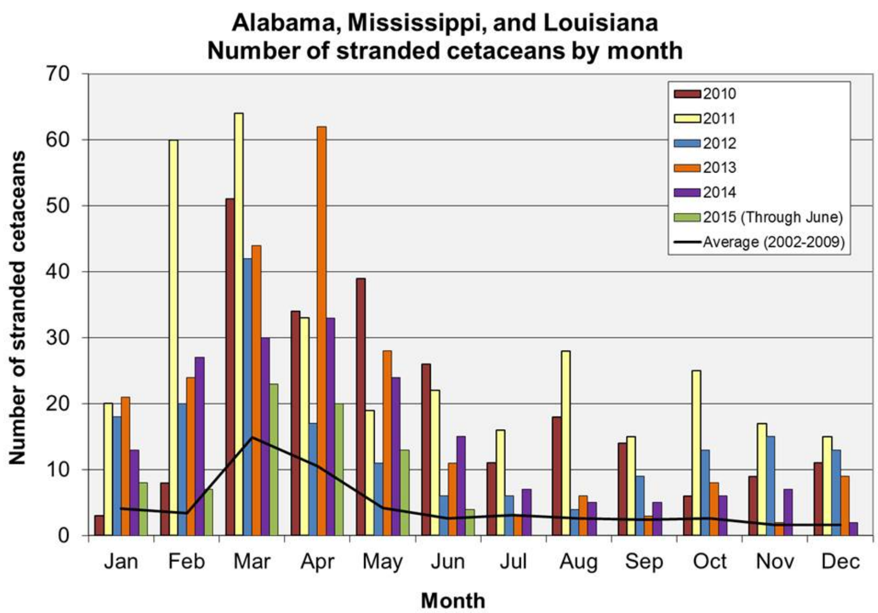


Figure 4.9-12. The DWH oil spill contributed to a large increase in monthly marine mammal strandings in the northern Gulf of Mexico. The number of monthly cetacean strandings along the coastlines of Louisiana, Mississippi, and Alabama from January 2010 to June 2015 is compared to the average number of strandings from 2002 to 2009 (solid line). Stranding data for this figure were extracted from the NOAA Marine Mammal Health and Stranding Response Program Database on August 24, 2015.

The majority of stranded cetaceans were common bottlenose dolphins (87 percent), and more than 95 percent were dead (Venn-Watson 2015). The geographic and temporal patterns of the elevated strandings overlap with the DWH oil footprint. The largest increase in stranding rates occurred in Barataria Bay, which sustained the longest period of consecutive months with unusually high numbers of stranded dolphins (August 2010 through December 2011). The numbers of stranded animals in 2010 and 2011 in Louisiana were the highest in recorded history; 2011 was also one of the highest stranding years for both Mississippi and Alabama. In contrast, beaches in Florida and Texas did not see a similar sustained increase in the number of dead, stranded dolphins (Venn-Watson et al. 2015c). Although there were elevated strandings in western Mississippi and outside of Lake Pontchartrain before the DWH oil spill, the Trustees' analyses show that these are likely due to a cold winter and mortality among dolphins from Lake Pontchartrain (DWH MMIQT 2015; Mullin et al. 2015; Venn-Watson et al. 2015c). Barataria Bay did not have elevated pre-spill strandings (Venn-Watson et al. 2015c).

The total number of stranded animals is likely an underestimate. The detection of marine mammal carcasses is very low, because many carcasses do not make it to shore and those that do may not be found or reported to stranding networks (DWH MMIQT 2015). For offshore animals, currents and time of drift preclude the beaching of a carcass onshore, where they are more likely to be reported to a stranding network than in the open ocean (DWH MMIQT 2015).

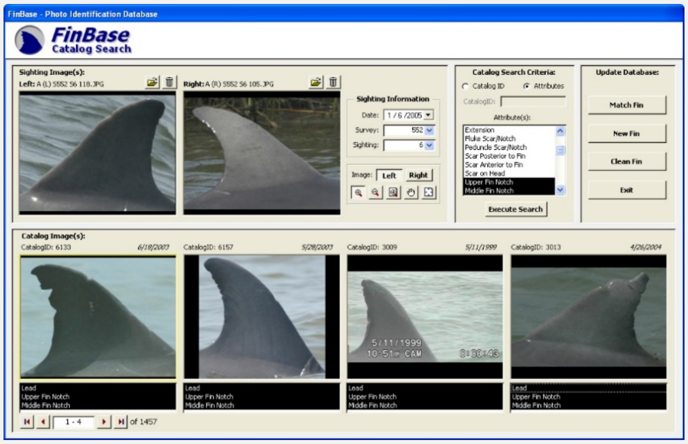
Consistent with the increased number of strandings in heavily oiled coastal locations, researchers conducting photo-identification studies observed high apparent mortality (i.e., unexplainable disappearances) among dolphin populations in Barataria Bay and Mississippi Sound (27 percent per year; confidence interval of 22 to 33 percent) in the year following the DWH spill (DWH MMIQT 2015). (Box 3 describes and illustrates the use of photo-identification studies, also known as photo-ID studies.) This mortality is referred to as "apparent," because photo-ID studies can only document the loss of individuals from the population, and both mortality and permanent emigration may contribute to those losses. These apparent mortality rates, however, are very high compared to those reported for other populations from the southeast U.S. coast measured using similar photo-ID study methods (3.9 and 4.9 percent per year for Sarasota Bay and Charleston, South Carolina stocks, respectively) (Speakman et al. 2010; Wells & Scott 1990)). Furthermore, dolphin satellite tracking data obtained in the years following the spill did not indicate long-range movements from either Barataria Bay or Mississippi Sound (Wells & Balmer 2012; Wells et al. 2014a; Wells et al. 2014b), making it unlikely that a significant number of dolphins were emigrating from these areas. Furthermore, a variety of marine mammal species suffered increased mortality rates after the *Exxon Valdez* oil spill, including sea otters, harbor seals, and killer whales (Frost et al. 1999; Garrott et al. 1993; Matkin et al. 2008).

In summary, the Trustees have determined that DWH oil exposure resulted in increased mortality to marine mammals. While UMEs can be caused by a variety of factors, including infection, biotoxins, cold stress, and other anthropogenic toxin exposures, researchers have ruled out these alternative hypotheses and determined that the most likely causes of the increased strandings and the observed high mortality rates are the adverse health effects caused by the DWH oil spill (Venn-Watson et al. 2015b).

Box 3: Photo ID Studies and Apparent Mortality

Source: *NOAA.*

Dolphins naturally acquire nicks and notches on their dorsal fins. These markings, along with unique fin shapes, can be used to identify individual animals. Photo-identification (photo-ID) studies use dorsal fin photographs to recognize and track individual animals over time. The data from photo-ID studies can be used to estimate survival rates and track reproductive events for females with marked fins. In addition, a mark-recapture model (a type of statistical model) can be used to estimate population size based on the re-sighting of previously identified dolphins relative to sighting of dolphins that had not been previously identified.



Source: *NOAA.*

FinBase is a customized Access database that allows photo-ID researchers to search an existing catalog of identifiable dorsal fins to find matches for newly photographed fins. Fins are assigned attributes such as "upper fin notch," which allows FinBase to present the existing dorsal fins in a sorted order to improve the searching efficiency.

4.9.4.4 Reproductive Failure

Marine mammal reproductive traits vary from species to species, the best understood being those of the bottlenose dolphin. Female bottlenose dolphins reach reproductive maturity between 5 and 12 years old and typically give birth every 3 to 5 years until they are as old as 48 years (Wells 2014). Gestation lasts about 380 days, and the mother, along with potentially other socially associated females, are critical for the successful recruitment of the calf into the population (Wells 2014). Stressors and injuries that affect the reproductive success of female marine mammals can have deleterious effects at the individual and population levels.

Twenty-two percent of all bottlenose dolphins that stranded between January 2010 and December 2013 were perinates, that is, pre-birth or very young newly born animals (Venn-Watson et al. 2015c). These dead animals showed a higher prevalence of fetal distress, failure of their lungs to inflate with air, and *in utero* pneumonia (not due to lungworm infection), when compared to reference perinates from outside the spill footprint.

During health assessments in Barataria Bay and Mississippi Sound, researchers identified a number of pregnant females, but their reproductive success (measured through follow-up surveys at and after the birth due date) was unusually low. Only 19.2 percent of pregnant females studied in Barataria Bay and Mississippi Sound between 2010 and 2014 gave birth to a viable calf. In contrast, dolphin populations in Florida and South Carolina have a pregnancy success rate of 64.7 percent (DWH MMIQT 2015). Pregnant females within the DWH oil spill footprint were more likely to abort a calf (due to infection or other factors), and if a calf was born, it was less likely to survive more than a few weeks (Smith et al. 2015). Figure 4.9-13 shows a Barataria Bay female with a dead near-term or newly born calf. The poor reproductive success was seen over successive years, not just for females that were pregnant during the spill (DWH MMIQT 2015; Smith et al. 2015).

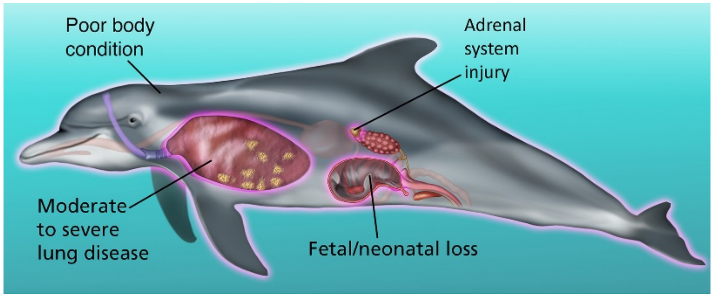
Source: *Jeremy Hartley, Louisiana Department of Wildlife and Fisheries.*

Figure 4.9-13. In August 2011, a female dolphin (Y01) was examined in Barataria Bay and found pregnant with an expected due date in May 2012. In March 2013, just 10 months after her due date, she was seen supporting a dead calf. The timing and size of the dead calf (which appears to be near- or full-term) suggests that Y01 must have lost the calf due in 2012, become pregnant again, and suffered a second loss. Seventy-eight percent of pregnant females in Barataria Bay lost their calves, either as fetuses, neonates, or calves less than 1 year old.

The reproductive failure in DWH oil-exposed animals is consistent with field and laboratory study results present in the literature. After the *Exxon Valdez* oil spill, sea otters suffered from high rates of maternal, fetal, and neonatal loss (Tuomi & Williams 1995). In laboratory studies in which mink were exposed to bunker C oil, mothers had fewer offspring and neonates were less likely to survive (Mazet et al. 2001). Oil exposure has also been linked to spontaneous abortions and infant mortality in rats, mink, and humans (Mazet et al. 2001; Merhi 2010; Thiel & Chahoud 1997).

The DWH oil spill is the most plausible explanation for the overall reproductive failure effects seen in dolphins exposed to DWH oil (Venn-Watson et al. 2015b). Evidence of *in utero* deaths and poor reproductive success was found in both live and dead dolphins within the DWH oil spill footprint (Colegrove et al. 2015; DWH MMIQT 2015). The confirmed high prevalences of adult dolphins with hypoadrenocorticism, moderate to severe primary bacterial pneumonia, and poor body weight–all adverse health effects attributed to DWH oil exposure (Schwacke et al. 2014; Venn-Watson et al. 2015b) (see Section 4.3, Toxicity)–were the most likely primary causes for the high rates of failed pregnancies in dolphins living within the oil spill footprint. Other potential causes of the reproductive failures, such as other contaminants, infectious diseases, or harmful algal toxins, were ruled out as primary causes for all of the failed pregnancies across the oil spill footprint. However, the effects of these stressors may have been exacerbated by the oil spill (particularly infectious pathogens such as *Brucella* that are known to cause abortions) (Colegrove et al. 2015; Smith 2015; Venn-Watson et al. 2015b).

The Trustees have determined that DWH oil exposure resulted in increased reproductive failure for marine mammals. There is also evidence that the prevalence of poor reproductive success in live dolphins is persisting 4 years after the DWH oil spill (DWH MMIQT 2015; Schwacke et al. 2014).



Source: *Kate Sweeney for NOAA.*

Figure 4.9-14. Bottlenose dolphins within the DWH oil spill footprint had a variety of oil-related health issues, including lung disease, adrenal system disruption, poor body condition, and reproductive failure, including fetal/neonatal loss. These health effects were seen in dead, stranded bottlenose dolphins; in health assessments conducted on live bottlenose dolphins in 2011, 2013, and 2014; and in reproductive follow-up studies conducted in 2012, 2014, and 2015.

4.9.4.5 Adverse Health Effects

The toxic components of oil, including, but not limited to, PAHs, cause a variety of adverse health effects (Figure 4.9-14). Toxic effects may manifest in different ways in different species, or even individuals. Thus, the Trustees relied upon a combination of field-based studies, analyses of tissues from stranded animals, and the toxicity literature in order to determine and characterize the injuries to marine mammals in the northern Gulf of Mexico. The DWH NRDA studies revealed a suite of adverse health effects that are both injuries in and of themselves (symptoms that will affect the quality of life for animals) and health effects that will lead to decreased survival and/or reproductive failure in northern Gulf of Mexico marine mammals.

Based on data from the live health assessments and post-mortem analyses of stranded animals, researchers have identified three primary adverse health effects that are the main contributors to the increased prevalence of sick animals, dead animals, and reproductive failure within the DWH oil spill footprint: lung disease, abnormal stress response, and poor body condition. They also identified a suite of other adverse health effects that compound the primary injuries, including anemia, liver disease, and dental disease. This section will describe each adverse health effect in further detail and put it into the context of an overall clinical prognosis for an animal's survival and reproductive potential.

4.9.4.5.1 Lung and Respiratory Impairments

The cetacean respiratory system is critical for individual fitness; reduced lung capacity will lead to less effective foraging, swimming, and diving. Since cetaceans breathe at the ocean surface, they are particularly at risk from exposure to the toxic chemicals in oil surface slicks via inhalation of volatiles and aerosols, as well as aspiration of liquid oil (see Section 4.9.3.2). The lungs of cetaceans, more than those of humans, are designed to maximize surface area to allow for efficient oxygen uptake and to expand diving capability. Physical disruptions, whether from oil coating tissues and membranes or organ/cellular damage due to chemical toxicity, can result in serious effects on animal health, including infections (Coppock et al. 1995; Coppock et al. 1996).

Through post-mortem examinations, marine mammal pathologists identified unusually high numbers of bottlenose dolphins with primary bacterial pneumonia and severe pneumonias (Venn-Watson et al. 2015a). Similarly, researchers found evidence of lung damage and disease in dolphins in Barataria Bay and Mississippi Sound during live dolphin health assessments (Schwacke et al. 2014).

From 2010 to 2012 (data beyond 2012 are not yet available), the prevalence of primary bacterial pneumonia was higher in animals that stranded within the footprint of the DWH oil spill (22 percent) than in animals that stranded outside of that footprint (2 percent) (Venn-Watson et al. 2015a). Many of these pneumonias were unusual in severity and caused or contributed to death (Venn-Watson et al. 2015a). Several types of bacteria were responsible for the large numbers of pneumonia cases, making it unlikely that all of these dolphins succumbed to a single infectious disease outbreak (Venn-Watson et al. 2015a). More likely, oil exposure weakened their lung health and systemic immune functions, resulting in infection by whatever bacteria could exploit the injury in each individual.

In 2011, 2013, and 2014, marine mammal veterinarians performed ultrasound examinations on dolphins captured for health assessments. They found unusually high rates of moderate to severe lung disease in dolphins from Barataria Bay and Mississippi Sound compared to animals from Sarasota Bay. Specifically, dolphins in areas contaminated with DWH oil were three to six times more likely to have moderate to severe lung disease compared to those in the Sarasota Bay reference site (Schwacke et al. 2014; Smith et al. 2015). Severe cases of lung disease were only found in oiled locations, and the severity and prevalence of lung disease decreased from 2011 to 2014 (Smith et al. 2015).

The lung injuries affecting animals in the DWH oil spill footprint were consistent with those reported in the human and mammalian oil toxicology literature, including research on inhaled, aspirated, and ingested oil and other chemicals (Akira & Suganuma 2014; Franzen et al. 2013; Lifshitz et al. 2003). In humans, inhalation exposure to the *Prestige*, *Hebei Spirit,* and *Tasman Spirit* oil spills resulted in increased respiratory symptoms (Carrasco et al. 2006; Janjua et al. 2006; Jung et al. 2013; Sim et al. 2010; Suarez et al. 2005; Zock et al. 2007; Zock et al. 2012), and DWH response workers reported increased respiratory symptoms (Sandler et al. 2014). Ingestion of oil, followed by aspiration of petroleum products, has led to pneumonia in both animals and humans (Bystrom 1989; Coppock et al. 1995; Coppock et al. 1996; Edwards 1989; Lifshitz et al. 2003; Sen et al. 2013). These studies are consistent with the exposure and injuries to northern Gulf of Mexico marine mammals as a result of the DWH oil spill. Lung disease in exposed dolphins likely contributed to the increased mortality within the DWH oil spill footprint, as well as to some of the other adverse health effects reported here, including poor body condition and reproductive failure. The lung disease seen in marine mammals is also consistent with the adverse health effects reported in other wildlife species exposed to DWH oil, including cardiac dysfunction, poor body condition/growth, anemia, immune system abnormalities, decreased flight/swimming performance in birds and fish, and reproductive failure (see Section 4.3, Toxicity). Based on the consistency between DWH marine mammal studies, DWH toxicity studies, and the literature, the Trustees have determined that DWH oil exposure likely caused lung disease in cetaceans, which contributed to the adverse health effects, mortality, and reproductive failure observed in exposed northern Gulf of Mexico marine mammals.

4.9.4.5.2 Adrenal Gland Impairments

The adrenal glands, part of the endocrine system, are responsible for secreting hormones (e.g., cortisol and aldosterone) into the blood stream that are essential for mounting an appropriate response to stress, maintaining glucose and electrolyte levels, modulating the immune system, and altering behavior. Hypoadrenocorticism is a disease state resulting from a deficiency in adrenal gland hormones (Klein & Peterson 2010), which, if left untreated, greatly increases an animal's risk of death from adrenal crisis, particularly during times of stress, such as illness or pregnancy. Toxigenic chemicals, including PAHs, have been shown to affect the adrenal gland in vertebrates (Nichols et al. 2011; Ribelin 1984). Therefore, dolphins within the DWH oil spill footprint were examined for evidence of adrenal disease and dysfunction.

Marine mammal veterinarians and pathologists found evidence of adrenal gland disease in bottlenose dolphins that stranded within the DWH oil spill footprint, and evidence of hypoadrenocorticism in live bottlenose dolphins from Barataria Bay and Mississippi Sound. Specifically, stranded dolphins had adrenal cortical atrophy (Venn-Watson et al. 2015a), and live dolphins had low cortisol and aldosterone hormone levels (Schwacke et al. 2014; Smith et al. 2015). Adrenal cortical atrophy (thinning of the cortex) is sometimes associated with adrenal insufficiency, or low hormone levels (Capen 2007).

An unusually high number of dead dolphins (33 percent) stranding in Louisiana, Mississippi, and Alabama between 2010 and 2012 had a low corticomedullary ratio (indicating a thin cortex); 50 percent of these animals stranded within Barataria Bay (Venn-Watson et al. 2015a). Approximately 7 percent of reference dolphins (outside the spill area) had a low corticomedullary ratio. A specific cause of death could not be identified for many of the dolphins with a thin adrenal cortex; the dolphins likely died of an adrenal crisis (Venn-Watson et al. 2015a). Histological evaluation of the adrenal glands in dead, stranded dolphins did not produce evidence of other possible causes of adrenal cortical atrophy (Venn-Watson et al. 2015a).

Normally, blood concentrations of cortisol are expected to increase following the initiation of a stressor, for example, the chase-capture process used in the health assessments (Thomson & Geraci 1986). This was not the case, however, for dolphins living in Barataria Bay and Mississippi Sound. Cortisol concentrations were lower than expected in dolphins captured in Barataria Bay (Schwacke et al. 2014; Smith et al. 2015). In fact, nearly half (44 percent) of the dolphins captured in Barataria Bay in 2011 had cortisol measures that were lower than the minimum values previously measured in other dolphin populations (Schwacke et al. 2014). Abnormal cortisol concentrations were also seen in Mississippi Sound dolphins in 2013, and continued to be seen in Barataria Bay dolphins in 2013 and 2014, although the number of individuals affected was lower than the number observed in 2011 (Smith et al. 2015). These findings are consistent with hypoadrenocorticism (Klein & Peterson 2010). Some bottlenose dolphins from Barataria Bay suffered from low blood sugar, high potassium, and/or low sodium, which are symptoms likely associated with hypoadrenocorticism (Schwacke et al. 2014).

The results from both the live and stranded bottlenose dolphin studies are unprecedented in terms of previous observations with marine mammals and consistent with literature concerning the effects of oil exposure on other species. Preliminary DWH NRDA studies using Gulf toadfish (*Opsanus beta*) and human adrenal cell lines demonstrated that DWH oil can disrupt stress hormone pathways (see Section 4.3, Toxicity; Takeshita et al. 2015). Laboratory studies exposing animals to other crude oils have demonstrated impaired stress responses and poorly functioning adrenal glands (Lattin et al. 2014; Mohr et al. 2008). The adrenal hormone abnormalities observed among Barataria Bay and Mississippi Sound dolphins were unexpected and significantly different than normal when compared to reference intervals, a reference site, and other dolphins previously sampled in the southeastern United States (Hart et al. 2015; Schwacke et al. 2014; Smith et al. 2015). Prior to this investigation, adrenal cortical atrophy had not been described in free-ranging cetaceans (Clark et al. 2005).

If left untreated, hypoadrenocorticism is life threatening and can lead to adrenal crisis and death in mammals (Arlt & Allolio 2003). Adrenal crises may have caused death in dolphins with damaged adrenal glands and contributed to death in dolphins exposed to factors to which a healthy dolphin would have otherwise adapted. Thus, adrenal cortical atrophy leading to adrenal insufficiency likely contributed to the increased mortality in areas affected by the DWH oil spill (Smith et al. 2015; Venn-Watson et al. 2015a; Venn-Watson et al. 2015c). It also likely contributed to other marine mammal adverse health effects, including reproductive failure in pregnant females (Colegrove et al. 2015); liver enzyme, glucose, potassium, and sodium imbalances; and poor body condition (Schwacke et al. 2014). The adrenal injury in marine mammals is also consistent with the injuries to other DWH oil-affected wildlife, including impaired stress response, immunosuppression, poor body condition/growth, and glucose/electrolyte imbalances (see Section 4.3, Toxicity; Section 4.7, Birds; and Section 4.8, Sea Turtles).

Based on the measures of adrenal function in health assessments, adrenal cortical thinning in dead dolphins, and a review of the available literature, the Trustees have determined that adrenal gland impairment is a result of exposure to DWH oil and likely contributed to the adverse health effects, mortality, and reproductive failure observed in exposed northern Gulf of Mexico marine mammals.

4.9.4.5.3 Body Condition

Many factors can be measured to assess the health of wildlife, but the fitness of an animal can be represented effectively and with little effort by assessing body condition, as measured by the ratio between weight and length. Poor body condition could be a symptom of many types of stressors, including exposure to toxins, illness, low feeding success, poor diet quality, or detrimental changes in behavior. Any increase in the severity or duration of decreased body condition will result in further reductions in the animal’s fitness, including its ability to feed, reproduce, and deal with other environmental stressors, and, in the worst scenarios, will eventually lead to death.

Dolphins exposed to DWH oil were more likely to be underweight than dolphins outside the oil spill footprint (Schwacke et al. 2014). In 2011, 25 percent of live animals captured in Barataria Bay had a low mass-to-length ratio, compared to only 4 percent of Sarasota animals. Furthermore, dolphins with poor body condition from Barataria Bay were more severely underweight, especially males, than the underweight animals from Sarasota Bay. The most likely cause of the low mass-to-length ratio in Barataria Bay dolphins was illness, in particular, the lung disease or hypoadrenocorticism associated with DWH oil exposure (Venn-Watson et al. 2015b).

The decreased body condition in dolphins exposed to DWH oil is consistent with the oil toxicity studies described in Section 4.3 (Toxicity) and with other adverse health effects observed in the marine mammal health assessments. In laboratory studies of DWH oil exposures, birds, fish, and invertebrates all suffered from reduced growth rates (Section 4.3, Toxicity).

A low mass-to-length ratio, depending on its severity, would likely contribute to the increased prevalence of reproductive failure and mortality in marine mammals exposed to DWH oil. Poor body condition is a possible effect of lung disease and adrenal disease.

Based on the poor body condition of animals in Barataria Bay and the prevalence of decreased body condition in oil-exposed wildlife in the literature, the Trustees have determined that poor body condition as a result of exposure to DWH oil likely contributed to the adverse health effects, mortality, and reproductive failure observed in exposed northern Gulf of Mexico marine mammals.

4.9.4.5.4 Other Adverse Health Effects

In addition to the adverse health effects described above, marine mammal veterinarians and biologists also documented the following:

- **Anemia**. While health assessments in 2011 showed that 13 percent of dolphins in Barataria Bay were anemic, no dolphins in Sarasota Bay showed signs of anemia (Schwacke et al. 2014). Anemia was also seen in birds and fish exposed to DWH oil, and is well documented in field and laboratory studies of oil exposure on a variety of species (Section 4.3, Toxicity; Section 4.7, Birds(Bursian et al. 2015a; Bursian et al. 2015b; Dorr et al. 2015; Harr et al. 2015)).

- **Tooth loss.** Some of the bottlenose dolphins in Barataria Bay had excessive tooth loss. Three animals had fewer than half of their teeth remaining, and three animals had lost all or nearly all of their teeth (eight, two, and zero teeth remaining) (Schwacke et al. 2014). Bottlenose dolphins typically have 76 to 108 teeth. All of these animals suffered from overgrown tissue in their gums. Beluga whales and pinnipeds exposed to toxic chemicals in the St. Lawrence estuary and the Baltic, respectively, exhibited tooth loss (Beland et al. 1993; Bergman et al. 1992). Laboratory exposures of rodents to PAHs resulted in inflammation, hyperplasia, and cell proliferation in the mouth (Brandon et al. 2009; Guttenplan et al. 2012; Wester et al. 2012). Mink exposed to chemical toxicants that act by a similar toxic mechanism to PAHs demonstrated bone loss, extreme tooth loss, and lesions in the jaw (Haynes et al. 2009; Render et al. 2000).

- **Liver damage.** Some Barataria Bay dolphins had signs of liver injury, including increased liver enzyme concentrations in the blood (Schwacke et al. 2014). Similar hepatic dysfunction was seen in sea otters following the *Exxon Valdez* spill and in laboratory exposures with mammals (Mazet et al. 2000; Rebar et al. 1995; Schwartz et al. 2004). Captive bottlenose dolphins exposed to oil showed signs of liver damage (Englehardt 1983). Abnormal liver function can lead to decreased fitness and, importantly in the context of exposure to DWH oil, reduce an animal's ability to cope with exposure to toxic chemicals.

4.9.4.5.5 Overall Prognosis

For each animal captured during the Barataria Bay and Mississippi Sound health assessments (as well as those captured during the health assessments for the Sarasota Bay reference site), veterinarians synthesized the results from the physical examinations, ultrasounds, hematology, and serum chemistry in order to generate an overall prognosis (Schwacke et al. 2014; Smith et al. 2015). Dolphins could be assigned one of the following prognoses: good (favorable outcome expected), fair (favorable outcome possible), guarded (outcome uncertain), poor (unfavorable outcome expected), or grave (death considered imminent). Dolphins examined in Barataria Bay in 2011, 1 year following the DWH oil spill, had a high prevalence of guarded or worse prognosis scores (48 percent) when compared to animals sampled in Sarasota Bay in 2011 (7 percent) or over both survey years (2011 and 2013) in Sarasota Bay (11 percent). In the years following, the prevalence of guarded or worse prognoses in Barataria Bay decreased but remained elevated (37 percent in 2013, 28 percent in 2014). Dolphins examined in Mississippi Sound, an area also impacted by DWH oil, had a higher prevalence of guarded or worse prognoses scores (35 percent in 2013) as well. Unfortunately, health assessments could not be conducted in earlier years for Mississippi Sound. Thus, it is not surprising that the constellation of adverse health effects documented in marine mammals exposed to DWH oil happened synchronously with the increased number of marine mammal strandings in the northern Gulf of Mexico.

4.9.4.6 Causation

The Trustees have determined that exposure to DWH-related petroleum products caused a suite of adverse health effects, including lung disease, adrenal disease, reproductive failure, mortality, and poor body condition, in bottlenose dolphins. This conclusion was based on the overlap of observed disease conditions with the DWH oil spill footprint (in terms of time, space, and severity), the consistent evidence of abnormalities in both live and dead animals, the coherence and interconnectedness of the particular adverse health effects observed (Figure 4.9-15), and the elimination of other plausible causes (Venn-Watson et al. 2015b).

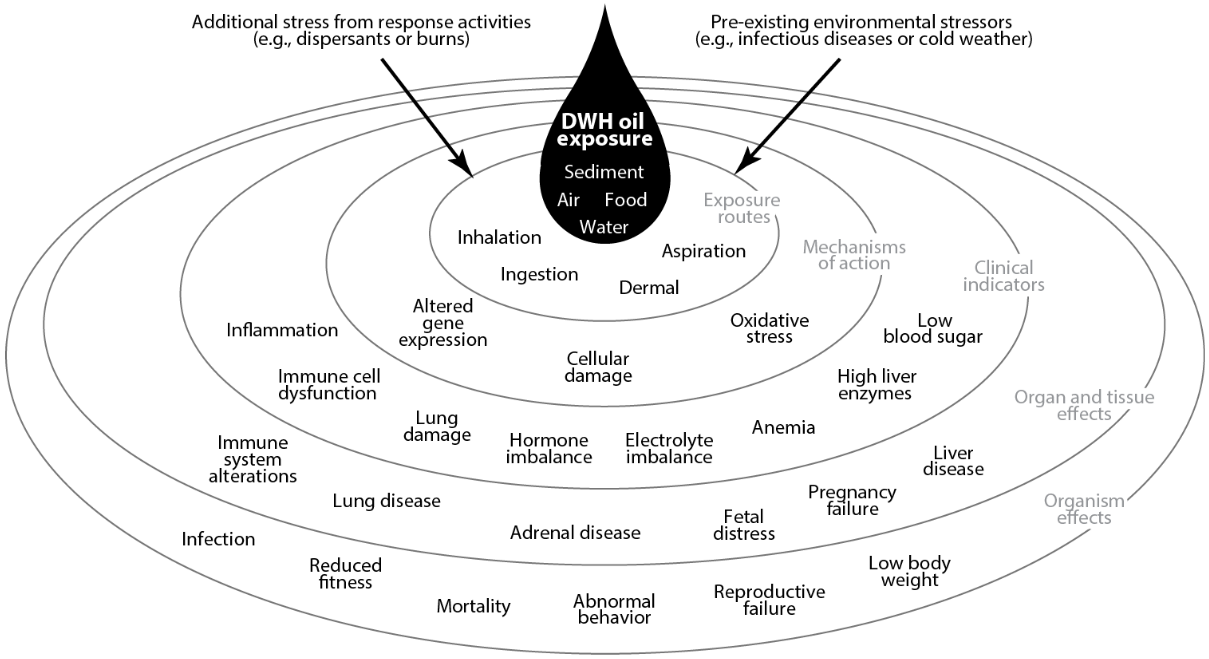


Figure 4.9-15. This conceptual model shows the interactions among oil exposure, exposure routes, mechanisms of action, clinical indicators, organ and tissue effects, and organism effects in marine mammals within the DWH oil spill footprint. Oil toxicity will manifest differently in each exposed individual, leading to different combinations of the outcomes listed in this diagram. The diagram includes information from DWH NRDA studies, as well as from the literature.

Very few factors can cause the suite of adverse health effects observed, and the most likely cause is exposure to DWH-related petroleum products. Researchers ruled out a variety of alternative explanations for the observed injuries, including morbillivirus, brucellosis, biotoxins, environmental stressors (e.g., cold temperatures or salinity changes), and fisheries/human interactions (Venn-Watson et al. 2015b). In addition, the adverse health effects seen in affected dolphins were consistent with what has been reported in other animal species experimentally exposed to petroleum and petroleum-associated products. These health effects, if left untreated, greatly increase an animal's risk of death, particularly during times of stress, illness, or pregnancy. Therefore, this constellation of adverse health effects almost certainly contributed to the poor survival rate and high reproductive failure of dolphins living in heavily oiled Barataria Bay and to the high stranding rates within the DWH oil spill footprint (DWH MMIQT 2015; Venn-Watson et al. 2015c).

Based on these findings and on the extent of exposure to DWH oil in the northern Gulf of Mexico, the Trustees concluded that it is reasonable to expect that marine mammals from oil-exposed species or stocks suffered similar effects as observed in Barataria Bay and Mississippi Sound.

4.9.5 Injury Quantification

Key Points

- The Trustees synthesized DWH NRDA exposure and injury data with the existing scientific literature to quantify the magnitude of injuries across the marine mammal populations in the northern Gulf of Mexico.
- The Trustees have determined that the majority of cetacean stocks within the DWH oil spill footprint were injured by some combination of increased mortality, increased reproductive failure, and/or adverse health effects, leading to reduced populations that will take decades to recover naturally.
- The injury quantification presented here builds from the measured injuries in Barataria Bay and Mississippi Sound bottlenose dolphins to other BSE stocks of bottlenose dolphins, and then to the coastal and oceanic stocks of bottlenose dolphins and other cetacean species within the DWH oil spill footprint.
- The Trustees quantified injuries to four BSE stocks: Barataria Bay, Mississippi River Delta, Mississippi Sound, and Mobile Bay. For example, in the Barataria Bay bottlenose dolphin stock, the DWH oil spill caused 35 percent (confidence interval of 15 to 49 percent) excess mortality, 46 percent (confidence interval of 21 to 65 percent) excess failed pregnancies, and a 37 percent (confidence interval of 14 to 57 percent) higher likelihood that animals would have adverse health effects (DWH MMIQT 2015).
- Shelf and oceanic stocks were also affected. Of these stocks, Bryde's whales were the most impacted, with 17 percent (confidence interval of 7 to 24 percent) excess mortality, 22 percent (confidence interval of 10 to 31 percent) excess failed pregnancies, and an 18 percent (confidence interval of 7 to 28 percent) higher likelihood of having adverse health effects (DWH MMIQT 2015).
- To more completely quantify the injury to each stock, statisticians and biologists used a population modeling approach to capture the overlapping and synergistic relationships among the three injuries, and to quantify the entire scope of DWH marine mammal injury to populations into the future.
- Based on the results of the population model, the Barataria Bay stock of bottlenose dolphins suffered 30,347 (confidence interval of 11,511 to 89,746) lost cetacean years due to the DWH oil spill. In the absence of active restoration, the population will take 39 (confidence interval of 24 to 80) years to recover. This represents a 51 percent (confidence interval of 32 to 72 percent) maximum reduction in the population size due to the DWH oil spill (DWH MMIQT 2015).

The DWH oil spill resulted in an increased number of dead, stranded marine mammals on the shorelines of Louisiana, Mississippi, and Alabama; reduced survival and increased reproductive failure in bottlenose dolphins surveyed in Barataria Bay and Mississippi Sound; and a constellation of adverse health effects, including lung disease, adrenal disease, and poor body condition in bottlenose dolphins examined in

Barataria Bay and Mississippi Sound. Given the severity of these observed injuries and the wide distribution of marine mammals throughout the DWH oil spill footprint, the Trustees quantified the degree, and spatial and temporal extent, of marine mammal injuries within the entire footprint.

In the wake of the DWH oil spill, and in the midst of the northern Gulf of Mexico UME, the bottlenose dolphin stocks in Barataria Bay and Mississippi Sound offered an opportunity to study the effects of DWH oil exposure on cetaceans, in a situation that could be both logistically feasible (given the difficulties studying dolphins and whales in the open ocean) and serve as a reasonable case study for other cetacean species (with adjustments for differences in behavior, anatomy, physiology, life histories, and population dynamics among species). Thus, the injury quantification presented here builds from the measured injuries in Barataria Bay and Mississippi Sound bottlenose dolphins to other BSE stocks of bottlenose dolphins, and then to the coastal and oceanic stocks of bottlenose dolphins and other cetacean species within the DWH oil spill footprint.

A total of 31 cetacean stocks (9 BSE, 2 coastal, 2 shelf, and 18 oceanic) overlap with the oil spill footprint. As a first step, scientists used stranding data to calculate excess mortality above baseline for the BSE stocks (DWH MMIQT 2015). Two oceanic stocks, Fraser's dolphins and killer whales, were not considered; although they are present in the Gulf of Mexico, sightings are very rare and there were no historical sightings in the oil spill footprint on the surveys used in the quantification. Four BSE stocks (Perdido Bay, Pensacola Bay, Choctawhatchee Bay, and St. Andrews Bay) did not show evidence of excess mortality based on the quantification approach described in Section 4.9.5.1, despite some oiling along beaches and barrier islands of these stock areas. As a result, injury quantification was not pursued for these four BSE stocks. Table 4.9-3 presents a list of all marine mammal stocks that the Trustees considered for injury quantification in the assessment and, for those that were not included, an explanation of why they were not included.

Table 4.9-3. Thirty-two marine mammal stocks, 31 cetacean and 1 sirenian, were considered for injury quantification.

Common Name/Species	Stock	Injury Quantified?	Explanation
BSE			
West Indian manatee	Gulf-wide	No	Exposure unlikely
Common bottlenose dolphin	Terrebonne-Timbalier Bay	No	Unable to distinguish between injury due to DWH oil versus cold temperatures
Common bottlenose dolphin	Barataria Bay	Yes	Included
Common bottlenose dolphin	Mississippi River Delta	Yes	Included
Common bottlenose dolphin	Mississippi Sound	Yes	Included
Common bottlenose dolphin	Mobile Bay	Yes	Included
Common bottlenose dolphin	Perdido Bay	No	No excess strandings attributed to DWH oil using regression model
Common bottlenose dolphin	Pensacola Bay	No	No excess strandings attributed to DWH oil using regression model
Common bottlenose dolphin	Choctawhatchee Bay	No	No excess strandings attributed to DWH oil using regression model

Common Name/Species	Stock	Injury Quantified?	Explanation
Common bottlenose dolphin	St. Andrews Bay	No	No excess strandings attributed to DWH oil using regression model
Coastal			
Common bottlenose dolphin	Western coastal	Yes	Included
Common bottlenose dolphin	Northern coastal	Yes	Included
Shelf			
Atlantic spotted dolphin	Gulf-wide	Yes	Included in “Shelf dolphins” summation[a]
Common bottlenose dolphin	Gulf-wide	Yes	Included in “Shelf dolphins” summation[a]
Oceanic			
Blainville's beaked whale	Gulf-wide	Yes	Included in “Beaked whales” summation[a]
Bryde’s whale	Gulf-wide	Yes	Included
Clymene dolphin	Gulf-wide	Yes	Included
Common bottlenose dolphin	Gulf-wide	Yes	Included
Cuvier's beaked whale	Gulf-wide	Yes	Included in “Beaked whales” summation[a]
Dwarf sperm whale	Gulf-wide	Yes	Included in "Pygmy/Dwarf sperm whales" summation[a]
False killer whale	Gulf-wide	Yes	Included
Gervais’ beaked whale	Gulf-wide	Yes	Included in “Beaked whales” summation[a]
Melon-headed whale	Gulf-wide	Yes	Included
Pantropical spotted dolphin	Gulf-wide	Yes	Included
Pygmy killer whale	Gulf-wide	Yes	Included
Pygmy sperm whale	Gulf-wide	Yes	Included in "Pygmy/Dwarf sperm whales" summation[a]
Risso’s dolphin	Gulf-wide	Yes	Included
Rough-toothed dolphin	Gulf-wide	Yes	Included
Pilot whale (short-finned)	Gulf-wide	Yes	Included
Sperm whale	Gulf-wide	Yes	Included
Spinner dolphin	Gulf-wide	Yes	Included
Striped dolphin	Gulf-wide	Yes	Included

[a] It can be difficult to distinguish between species during vessel and aerial surveys. Thus, for the purposes of the injury quantification, some species were grouped together.

Marine mammal biologists, veterinarians, and statisticians worked together to integrate DWH NRDA exposure and injury data with the existing scientific literature to most appropriately quantify the magnitude of injuries across the marine mammal populations in the northern Gulf of Mexico. The Trustees have determined that the majority of cetacean stocks within the DWH oil spill footprint were injured by some combination of increased mortality, increased reproductive failure, and/or adverse health effects—leading to reduced populations that will take decades to recover naturally. In the Terrebonne/Timbalier Bay stock (one of the stocks for which the Trustees did not quantify injury), it is still likely that DWH oil exposure resulted in injuries; however, there are not enough data about the populations and injuries before and after the spill to make a determination at this time.

The quantification approaches are summarized more thoroughly in the DWH Marine Mammal Injury Quantification Team Technical report (DWH MMIQT 2015). Summaries are tabulated in Table 4.9-12 and Table 4.9-13 at the end of Section 4.9.5.4.

4.9.5.1 Mortality

The Trustees have determined that DWH oil exposure resulted in the increased number of dead cetaceans in the northern Gulf of Mexico following the DWH blowout. The numbers of deaths were above the expected baseline mortality. These significantly increased mortality rates in cetaceans exposed to DWH oil will have a negative impact on each population stock for years to come (Table 4.9-4 and Table 4.9-6).

Statistical analysis of data from photo-ID surveys conducted from 2010 to 2013 in Barataria Bay and from 2010 to early 2012 in Mississippi Sound estimated the proportion of each stock that succumbed to the effects of DWH oil exposure. Statisticians estimated the annual mortality rates in the 3 years in Barataria Bay and 1 year in Mississippi Sound following the DWH oil spill and compared them to previously reported annual mortality rates from other southeast U.S. dolphin populations (hereafter referred to as baseline or expected mortality). They used these comparisons to estimate excess mortality attributable to the DWH oil spill (hereafter referred to as excess mortality). Excess mortality was calculated as the difference between the expected annual mortality and the estimated annual mortalities for Barataria Bay and Mississippi Sound, and the differences were then summed over the multiyear period. Based on these calculations, there was 35 percent (confidence interval of 15 to 49 percent) excess mortality in Barataria Bay and 22 percent (confidence interval of 13 to 29 percent) excess mortality in Mississippi Sound (DWH MMIQT 2015).

Table 4.9-4. The DWH oil spill caused the deaths of BSE and coastal bottlenose dolphins throughout the surface slick footprint. This table presents the estimated percentage of each stock that died due to the DWH oil spill (above baseline).

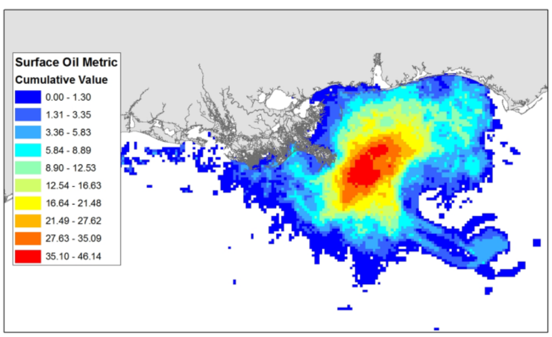
Bottlenose Dolphin Stock	Population Killed (%)	95% CI
Barataria Bay	35	15-49
Mississippi River Delta	59	29-100
Mississippi Sound	22[a]	13-29
Mobile Bay	12	5-20
Western coastal	1	1-2
Northern coastal	38	26-58

[a] Based on 1 year of surveys; all other stocks based on 3 years of observations.

For the other five potentially affected BSE and two coastal bottlenose dolphin regions, statisticians used a linear regression model to compare the actual monthly strandings for each region during and after the spill to the predicted monthly strandings in a scenario assuming the oil spill never took place. This “no oil spill” scenario is modeled based on the average historical monthly strandings and weather/environmental conditions during and after the spill (DWH MMIQT 2015). Statisticians then calculated the number of excess strandings above baseline for the 3 years following the spill for each region exposed to DWH oil, after accounting for other potential contributing factors, such as cold temperatures. Stranded dolphins were tested using genetics and stable isotopes to determine if they belonged to a BSE or coastal stock. Once it was determined what percentage of the strandings came from the BSE or adjacent coastal stock, the number of strandings was scaled to estimate the number of mortalities for each stock, including those that did not wash ashore or were not detected. Based on these calculations, the Mississippi River Delta stock experienced 59 percent (confidence interval of 29 to 100 percent) excess mortality, and the Mobile Bay stock had 12 percent (confidence interval of 5 to 20 percent) excess mortality attributable to the DWH spill (DWH MMIQT 2015).

The Terrebonne-Timbalier Bay stock had higher stranding rates in the spring and summer of 2010 compared to baseline, but the statistical model could not distinguish mortalities due to oil exposure from mortalities due to cold weather. In addition, the marshy habitat in Terrebonne Bay is remote, and many strandings in the estuarine system may not be found and therefore go unreported. Thus, it was not possible to distinguish excess strandings from baseline in Terrebonne-Timbalier Bay, and the Trustees did not perform further injury quantification for this stock.

For the other BSE stocks (Perdido Bay, Pensacola Bay, Choctawhatchee Bay, and St. Andrews Bay), there was no evidence of excess mortality in the post-spill period, based on the results of the linear regression model (DWH MMIQT 2015). DWH oil reached the coastal shores of these Florida bays, but with intermittent frequency (especially compared to the Louisiana, Mississippi, and Alabama coastal areas), and the oil did not penetrate very far into the estuarine waters.



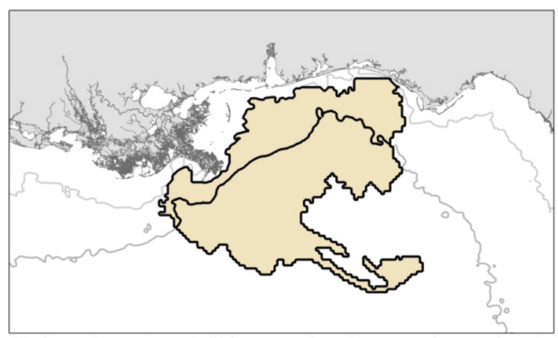


Figure 4.9-16. Marine mammals in coastal, shelf, and oceanic communities in waters with equal or greater DWH surface oiling (measured in days with at least a thin sheen) than in Barataria Bay would be subject to equal or greater DWH oil toxicity (DWH MMIQT 2015). **Figure 4.9-16a** (left) shows the entire oil footprint with a metric presenting comparative surface oiling across the Gulf of Mexico on the days of imaging. Based on the average of this metric in Barataria Bay, Barataria Bay mortality rates and reproduction rates were applied to areas of the shelf/oceanic area that had at least the same amount of oil. In **Figure 4.9-16b** (right)**,** the polygon shows the area of shelf (20 to 200 meters isobaths) and oceanic habitats (less than 200 meters isobaths) that exceed the Barataria Bay oiling metric.

To calculate the increase in percent mortality for the shelf and oceanic marine mammal stocks, the Barataria Bay percent mortality was applied to the percentage of animals in each stock that was exposed to oil (DWH MMIQT 2015). For the purposes of calculating the percentage of the population exposed, this quantification assumes that animals experiencing a level of cumulative surface oiling similar to or greater than that in Barataria Bay (Table 4.9-5) would have been likely to suffer a similar or greater degree and magnitude of injury (Figure 4.9-16). For example, Barataria Bay dolphins experienced 35 percent excess mortality; 47 percent of the spinner dolphin stock range in the northern Gulf of Mexico experienced oiling equal to or greater than Barataria Bay, and, therefore, would have experienced at least a 35 percent mortality increase. Thus, the entire northern Gulf of Mexico spinner dolphin stock experienced a 16 percent mortality increase (0.35 x 0.47 = 0.16). The results of these calculations for each shelf and oceanic stock are presented in Table 4.9-6.

Table 4.9-5. This table presents estimates of pre-spill abundance and percentage of population exposed to DWH oil for each northern Gulf of Mexico cetacean stock with quantifiable injury (DWH MMIQT 2015). Cetaceans experiencing a level of surface oiling similar to or greater than that experienced by bottlenose dolphins in Barataria Bay would likely have suffered a similar or greater degree and magnitude of injury.

Cetacean Stock	Pre-spill Abundance Estimate	95% CI	Population Exposed to Oil (%)	95% CI
Bottlenose dolphin Barataria Bay	2,306	1,973-2,639	NA	NA
Bottlenose dolphin Mississippi River Delta	820	657-984	NA	NA
Bottlenose dolphin Mississippi Sound	4,188	3,617-4,760	NA	NA
Bottlenose dolphin Mobile Bay	1,393	1,252-1,535	NA	NA
Bottlenose dolphin western coastal	20,161	14,482-28,066	23	16-32
Bottlenose dolphin northern coastal	7,185	4,800-10,754	82	55-100
Continental shelf dolphins[a]	63,361	42,898-87,417	13	9-19
Bottlenose dolphin oceanic	8,467	4,285-16,731	10	5-20
Sperm whale	1,635	1,132-2,359	16	11-23
Bryde's whale	26	12-56	48	23-100
Beaked whales[b]	1,167	643-2,117	12	7-22
Clymene dolphin	3,228	1,558-6,691	7	3-15
False killer whale	316	121-827	18	7-48
Melon-headed whale	1,696	709-4,060	15	6-36
Pantropical spotted dolphin	33,382	25,489-43,719	20	15-26
Short-finned pilot whale	1,641	710-3,790	6	4-9
Pygmy killer whale	281	131-601	15	7-33
Pygmy/dwarf sperm whales[c]	6,690	3,482-12,857	15	8-29
Risso's dolphin	1,848	1,123-3,041	8	5-13
Rough-toothed dolphin	2,414	964-6,040	41	16-100
Spinner dolphin	6,621	3,386-12,947	47	24-91
Striped dolphin	2,605	1,537-4,415	13	8-22

[a] Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

[b] Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

[c] Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

For coastal stocks, the excess mortality estimates are 1 percent (confidence interval of 1 to 2 percent) and 38 percent (confidence interval of 26 to 58 percent) for the western and northern coastal stocks, respectively (DWH MMIQT 2015). The increase in mortality due to DWH oil exposure in the shelf and oceanic stocks ranges from 2 to 17 percent of each population (DWH MMIQT 2015).

Table 4.9-6. The DWH oil spill caused the deaths of shelf and oceanic cetaceans throughout the surface slick footprint. This table presents the estimated percentage of each stock that died due to the DWH oil spill (above baseline).

Cetacean Stock	Population Killed (%)	95% CI
Continental shelf dolphins[a]	4	2-6
Bottlenose dolphin oceanic	3	1-5
Sperm whale	6	2-8
Bryde's whale	17	7-24
Beaked whales[b]	4	2-6
Clymene dolphin	2	1-4
False killer whale	6	3-9
Melon-headed whale	5	2-7
Pantropical spotted dolphin	7	3-10
Short-finned pilot whale	2	1-3
Pygmy killer whale	5	2-8
Pygmy/dwarf sperm whales[c]	5	2-7
Risso's dolphin	3	1-4
Rough-toothed dolphin	14	6-20
Spinner dolphin	16	7-23
Striped dolphin	5	2-7

a Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

b Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

c Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

4.9.5.2 Reproductive Failure

The Trustees have determined that DWH oil exposure resulted in the increased number of dead, stranded perinates and the unexpected number of unsuccessful pregnancies documented during Barataria Bay and Mississippi Sound surveys. The numbers of reproductive failures in these health assessment studies were above the expected baseline failures, based on historical monthly perinate stranding averages and reproductive failure rates in the bottlenose dolphin reference stocks in the southeastern United States. The increased reproductive failure rates in pregnant females exposed to DWH oil will have a negative impact on each population stock.

From 2011 to 2014, researchers tracked the numbers of pregnant females and successful pregnancies identified during health assessments and by measuring hormone levels in blubber biopsies (DWH MMIQT 2015). The Trustees pooled data from Barataria Bay and Mississippi Sound to achieve a reasonable sample size. Researchers found an excess of 46 percent (confidence interval of 21 to 65 percent) failed pregnancies in Barataria Bay and Mississippi Sound compared to the expected rate of reproductive failure based on reported observations from the Charleston, South Carolina; Indian River Lagoon, Florida; and Sarasota Bay, Florida, bottlenose dolphin populations (DWH MMIQT 2015). In other words, exposure to DWH oil caused 46 percent of pregnant females in Barataria Bay and Mississippi

Sound to lose their calves. No reproductive failure data are available for other stocks exposed to the DWH oil spill. Thus, the percentage of females with reproductive failure in Barataria Bay and Mississippi Sound (46 percent) is the best estimate of excess failed pregnancies for marine mammals in the oil spill footprint (Table 4.9-7) (DWH MMIQT 2015).

The quantification of reproductive failure in coastal, shelf, and oceanic stocks is analogous to the percent mortality calculations described in the previous section. The reproductive failure rate from the Barataria Bay and Mississippi Sound stocks (46 percent) is applied to the percentage of each stock that experienced levels of DWH oil exposure similar to or greater than animals in Barataria Bay. The increase in percentage of failed pregnancies due to DWH oil exposure in the two coastal stocks ranged from 10 to 37 percent, and in the shelf and oceanic stocks from 3 to 22 percent of each stock (Table 4.9-7) (DWH MMIQT 2015).

Table 4.9-7. The DWH oil spill caused reproductive failure in cetaceans throughout the surface slick footprint. This table presents the estimated percentage of females in each stock that suffered from reproductive failure due to the DWH oil spill (above baseline).

Cetacean Stock	Females with Reproductive Failure (%)	95% CI
Bottlenose dolphin Barataria Bay	46	21-65
Bottlenose dolphin Mississippi River Delta	46	21-65
Bottlenose dolphin Mississippi Sound	46	21-65
Bottlenose dolphin Mobile Bay	46	21-65
Bottlenose dolphin western coastal	10	5-15
Bottlenose dolphin northern coastal	37	17-53
Continental shelf dolphins[a]	6	3-8
Bottlenose dolphin oceanic	5	2-6
Sperm whale	7	3-10
Bryde's whale	22	10-31
Beaked whales[b]	5	3-8
Clymene dolphin	3	2-5
False killer whale	8	4-12
Melon-headed whale	7	3-10
Pantropical spotted dolphin	9	4-13
Short-finned pilot whale	3	1-4
Pygmy killer whale	7	3-10
Pygmy/dwarf sperm whales[c]	7	3-10
Risso's dolphin	3	2-5
Rough-toothed dolphin	19	9-26
Spinner dolphin	21	10-30
Striped dolphin	6	3-9

a Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

b Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

c Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

4.9.5.3 Adverse Health Effects

The Trustees have determined that DWH oil exposure resulted in the constellation of adverse health effects documented in health assessments of bottlenose dolphins in Barataria Bay and Mississippi Sound. Veterinarians assigned a general prognosis to each individual animal (based on its various combinations of adverse health effects) in order to characterize the dolphin's likely future outcome. The percentage of the populations within a given prognosis category is meaningful and predictive. For example, two dolphins that were given a grave prognosis in August 2011 died within 6 months, and the percentage of the population with the two lowest prognoses (17 percent poor and grave) essentially predicted the percentage of dolphins that disappeared and presumably died the following year based on photo-ID surveys (17 percent, confidence interval of 14 to 21 percent) (DWH MMIQT 2015). To quantify the magnitude of animals with adverse health effects in each stock, the Trustees used the overall prognosis to encapsulate the various combinations and severities of adverse health effects in each individual. In other words, they asked, what is the likely outcome for dolphins exposed to DWH oil compared to unexposed animals?

In Barataria Bay and Mississippi Sound, the percentage of the population with a guarded or worse prognosis was 37 percent and 24 percent higher, respectively, compared with dolphins sampled in Sarasota Bay (DWH MMIQT 2015). Biologists applied these numbers to the Mississippi River Delta and Mobile Bay stocks, respectively, based on the similar habitat and exposure levels in each pair (Table 4.9-8). (The Mississippi River Delta stock is most similar to the Barataria Bay stock; the Mobile Bay stock is most similar to the Mississippi Sound stock.) The quantification of adverse health effects for coastal, shelf, and oceanic stocks uses the same logic as the reproductive failure quantification: the Barataria Bay adverse health effects metric (37 percent) is applied to the percentage of each stock that experienced a level of DWH oil exposure equal to or greater than the stock in Barataria Bay. The percentage range for each of these stocks that suffer from an increase in adverse health effects is 2 to 30 percent (Table 4.9-9) (DWH MMIQT 2015).

Table 4.9-8. The DWH oil spill caused adverse health effects in BSE and coastal bottlenose dolphins throughout the surface slick footprint. This table presents the estimated percentage of each stock that suffered adverse health effects due to the DWH oil spill (above baseline).

Bottlenose Dolphin Stock	Population with Adverse Health Effects (%)	95% CI
Barataria Bay	37	14-57
Mississippi River Delta	37	14-57
Mississippi Sound	24	0-48
Mobile Bay	24	0-48
Western coastal	8	3-13
Northern coastal	30	11-47

Table 4.9-9. The DWH oil spill caused adverse health effects in shelf and oceanic cetaceans throughout the surface slick footprint. This table presents the estimated percentage of each stock that suffered adverse health effects due to the DWH oil spill (above baseline).

Cetacean Stock	Population with Adverse Health Effects (%)	95% CI
Continental shelf dolphins[a]	5	2-7
Bottlenose dolphin oceanic	4	1-6
Sperm whale	6	2-9
Bryde's whale	18	7-28
Beaked whales[b]	4	2-7
Clymene dolphin	3	1-4
False killer whale	7	3-11
Melon-headed whale	6	2-9
Pantropical spotted dolphin	7	3-11
Short-finned pilot whale	2	1-3
Pygmy killer whale	6	2-9
Pygmy/dwarf sperm whales[c]	6	2-9
Risso's dolphin	3	1-4
Rough-toothed dolphin	15	6-23
Spinner dolphin	17	6-27
Striped dolphin	5	2-8

[a] Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

[b] Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

[c] Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

4.9.5.4 Overall Effects on Populations

The increases in mortality, reproductive failure, and adverse health effects represent a snapshot of how DWH oil exposure impacted each northern Gulf of Mexico population stock from 2011 to 2013. They do not, however, capture the overlapping and synergistic relationships among the three injuries, and fail to quantify the entire scope of the DWH oil spill injury to marine mammal populations into the future. To more completely quantify the injury to each stock, statisticians and biologists used a population modeling approach. Cetaceans are long-lived, slow-maturing species. Thus, populations have difficulty recovering from the loss of reproductive adults, whether from illness, death, or a decrease in reproductive success. A population model allows consideration of long-term impacts resulting from individual losses, adverse reproductive effects, and persistent impacts on survival for exposed animals (Figure 4.9-17).

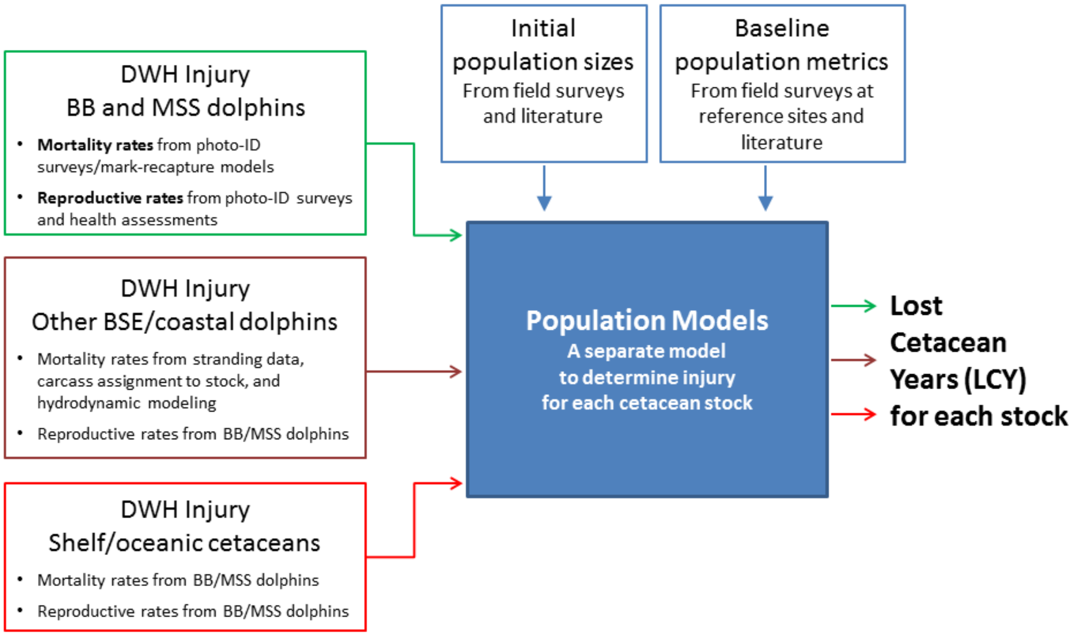


Figure 4.9-17. Population models were used to calculate the number of lost cetacean years (for each stock) due to the DWH oil spill. A lost cetacean year (LCY) is analogous to production foregone. Marine mammal scientists ran a separate population model for each stock; however, for convenience, we use the term lost cetacean years to describe the output of each model.

The model for the DWH marine mammal injury quantification (DWH MMIQT 2015) is run using baseline mortality and reproductive parameters to determine what the population trajectory of each stock would have been if the DWH spill had not happened. The same model is then run a second time, with estimates for excess mortality, reproductive failures, and adverse health effects due to the DWH oil spill. Figure 4.9-18 shows the result of a population model for an example cetacean population. The number of years predicted for the DWH oil-impacted population to recover (without active restoration) is the number of years until the DWH oil-injured population trajectory catches up with the baseline population trajectory, reported as years to recovery (YTR). In addition, the difference between the two trajectories summed over the years until the DWH oil-impacted population recovers is the total number of lost cetacean years (LCY) due to the DWH oil spill. This measure of LCY is the sum of all of the years of life lost, from animals that died earlier than they would have to animals that were never born due to reproductive failure. The output from the population model also predicts the largest proportional decrease in population size (i.e., the difference between the two population trajectories when the DWH oil-impacted trajectory is at its lowest point).

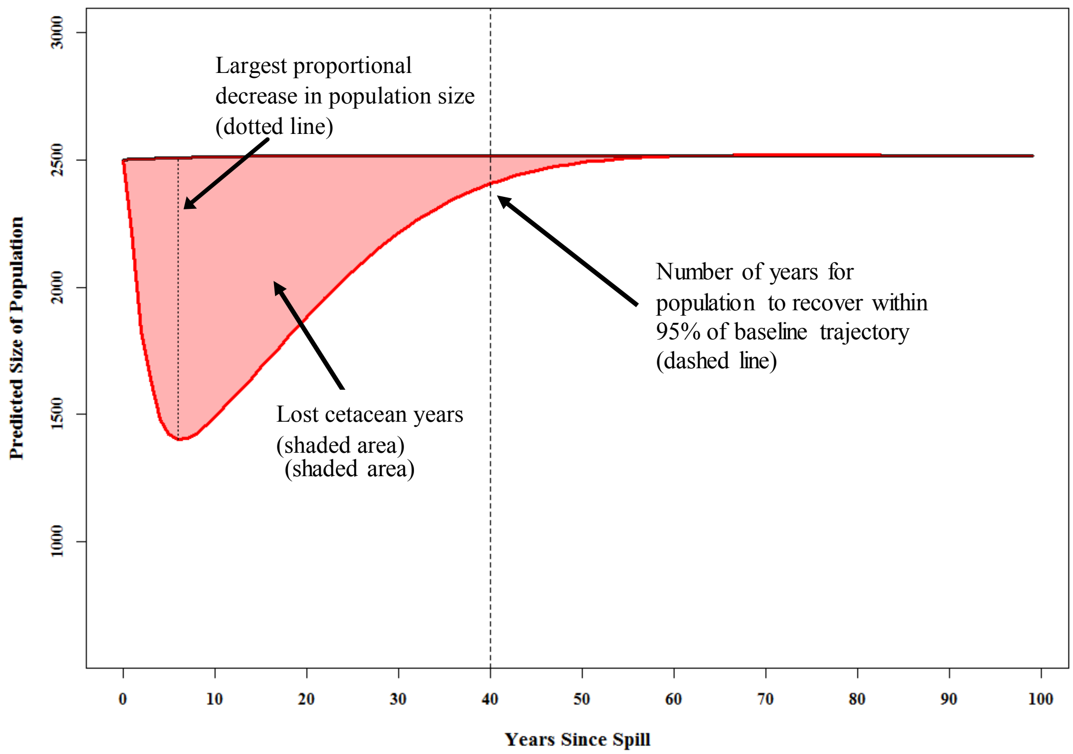


Figure 4.9-18. Assuming a stable baseline population size, this population model demonstrates that this example cetacean population will continue to decline for approximately 7 years, then will begin a slow recovery period lasting approximately 40 years. To quantify injury to each cetacean stock, the Trustees report the years to recovery (YTR) as the time it takes for the population to recover to 95 percent of the baseline trajectory (dashed line); the lost cetacean years (LCY) as the summed difference between the two trajectories (shaded area); and the maximum reduction in population size as the largest population difference in the two trajectories (dotted line).

A separate population model is run for each cetacean stock (Table 4.9-13). The inputs for the population models are restricted to the available data for each stock. For inputs without empirical data, the values are extrapolated from other stocks or incorporate additional modeling efforts (DWH MMIQT 2015). The Barataria Bay and Mississippi Sound population models mostly rely upon empirical data from the health assessments and population surveys.

The Barataria Bay stock of bottlenose dolphins suffered 30,347 LCY (confidence interval of 11,511 to 89,746) due to the DWH oil spill (Table 4.9-10) (DWH MMIQT 2015). In the absence of active restoration, the population will take 39 YTR (confidence interval of 24 to 80). This represents a 51 percent (confidence interval of 32 to 72 percent) maximum reduction in the population size due to the DWH oil spill. The Mississippi Sound bottlenose dolphin stock experienced 78,266 LCY (confidence interval of 38,858 to 219,602), with 46 YTR (confidence interval of 27 to 89) and a 62 percent (confidence interval of 43 to 83 percent) maximum reduction in the population size due to the DWH oil spill. The higher range of LCY in Mississippi Sound is due to the higher abundance of animals in Mississippi Sound compared to Barataria Bay (Table 4.9-10) (DWH MMIQT 2015).

Table 4.9-10. The DWH oil spill negatively impacted BSE and coastal bottlenose dolphin stocks throughout the surface slick footprint. This table presents the results of population models for each stock.

Bottlenose Dolphin Stock	Lost Cetacean Years	95% CI	Years to Recovery[a]	95% CI	Maximum Population Reduction (%)	95% CI
Barataria Bay	30,347	11,511-89,746	39	24-80	-51	32-72
Mississippi River Delta	20,065	4,896-62,355	52	27-106	-71	40-97
Mississippi Sound	78,266	30,858-219,602	46	27-89	-62	43-83
Mobile Bay	9,362	3,429-32,356	31	18-65	-31	20-51
Western coastal	19,041	6,869-64,245	NA	NA	-5	3-9
Northern coastal	92,069	36,427-264,716	39	23-76	-50	32-73

[a] It was not possible to calculate YTR for stocks with maximum population reductions of ≤ 5% (see DWH MMIQT 2015 for details).

The Mississippi River Delta stock of bottlenose dolphins suffered 20,065 LCY (confidence interval of 4,896 to 62,355) due to the DWH oil spill, including a 71 percent (confidence interval of 40 to 97 percent) maximum population reduction (Table 4.9-10) (DWH MMIQT 2015). In the absence of active restoration, the population will take 52 YTR (confidence interval of 27 to 106). The Mobile Bay bottlenose dolphin stock experienced 9,362 LCY (confidence interval of 3,429 to 32,356), with 31 YTR (confidence interval of 18 to 65 years) and a 31 percent (confidence interval of 20 to 51 percent) maximum reduction in population (Table 4.9-10) (DWH MMIQT 2015).

The values for LCY for the shelf and oceanic stocks varied widely due to differences in population sizes and proportions of the populations impacted by DWH oil (Table 4.9-11) (DWH MMIQT 2015). For the two stocks with the greatest abundances, the shelf dolphins lost 359,996 cetacean years and pantropical spotted dolphins lost 363,780 cetacean years. Of the two, the pantropical spotted dolphins had the greatest change in population size with a 9 percent maximum decrease requiring 39 years to recover. The shelf dolphins experienced a 3 percent maximum decline in population size; this decline was not significantly lower than 95 percent of the original population size and so YTR could not be determined (Table 4.9-11) (DWH MMIQT 2015).

Spinner dolphins, rough-toothed dolphins, pygmy and dwarf sperm whales, and oceanic bottlenose dolphins had LCY values ranging between 37,688 and 188,713 (Table 4.9-11) (DWH MMIQT 2015). Spinner dolphins and rough-toothed dolphins had the highest maximum reductions in population size at 23 percent (105 YTR) and 17 percent (54 YTR). As with the shelf dolphins, several oceanic stocks had declines of less than 5 percent of the original population size, so YTR could not be determined (Table 4.9-11) (DWH MMIQT 2015).

Table 4.9-11. The DWH oil spill negatively impacted coastal and oceanic cetacean stocks throughout the surface slick footprint. This table presents the results of population models for each stock.

Cetacean Stock[a]	Lost Cetacean Years	Years to Recovery[b]	Maximum Population Reduction (%)
Continental shelf dolphins[c]	359,996	NA	-3
Bottlenose dolphin oceanic	37,668	NA	-4
Sperm whale	13,197	21	-7
Bryde's whale	705	69	-22
Beaked whales[d]	7,838	10	-6
Clymene dolphin	12,167	NA	-3
False killer whale	3,422	42	-9
Melon-headed whale	14,887	29	-7
Pantropical spotted dolphin	363,780	39	-9
Short-finned pilot whale	5,304	NA	-3
Pygmy killer whale	2,501	29	-7
Pygmy/dwarf sperm whales[e]	49,100	11	-6
Risso's dolphin	6,258	NA	-3
Rough-toothed dolphin	50,464	54	-17
Spinner dolphin	188,713	105	-23
Striped dolphin	18,647	14	-6

[a] Confidence intervals for shelf and oceanic animals were not calculated (see DWH MMIQT 2015 for details).

[b] It was not possible to calculate YTR for stocks with maximum population reductions of ≤ 5% (see DWH MMIQT 2015 for details).

[c] Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

[d] Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

[e] Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

Two species of particular concern are the endangered sperm whales and Bryde's whales. For sperm whales, DWH oil exposure resulted in 13,197 LCY and a 7 percent maximum decline in population size, requiring 21 YTR (Table 4.9-11) (DWH MMIQT 2015). For Bryde's whales, 48 percent of the population was impacted by DWH oil, resulting in an estimated 22 percent maximum decline in population size that will require 69 YTR. Due to the very small Bryde's whale population size (26 animals, confidence interval of 12 to 56), the number of LCY is only 705. These results, however, should be interpreted with caution, for Bryde's whales, in particular. Small populations are highly susceptible to stochastic, or unpredictable, processes and genetic effects that can reduce productivity and resiliency to perturbations. The population models do not account for these effects, and, therefore, the capability of the Bryde's whale population to recover from this injury is unknown (Table 4.9-11) (DWH MMIQT 2015).

Table 4.9-12. This table summarizes the injuries to northern Gulf of Mexico cetaceans caused by the DWH oil spill, including the percentage of each stock killed, the percentage of each stock with reproductive failure, and the percentage of each stock with adverse health effects.

Cetacean Stock	Pre-spill Abun-dance Estimate	95% CI	Popula-tion Exposed to Oil (%)	95% CI	Popula-tion Killed (%)	95% CI	Females with Repro-ductive Failure (%)	95% CI	Population with Adverse Health Effects (%)	95% CI
Bottlenose dolphin Barataria Bay	2,306	1,973-2,639	NA	NA	35	15-49	46	21-65	37	14-57
Bottlenose dolphin Mississippi River Delta	820	657-984	NA	NA	59	29-1	46	21-65	37	14-57
Bottlenose dolphin Mississippi Sound	4,188	3,617-4,760	NA	NA	22[a]	13-29	46	21-65	24	0-48
Bottlenose dolphin Mobile Bay	1,393	1,252-1,535	NA	NA	12	5-20	46	21-65	24	0-48
Bottlenose dolphin western coastal	20,161	14,482-28,066	23	16-32	1	1-2	10	5-15	8	3-13
Bottlenose dolphin northern coastal	7,185	4,800-10,754	82	55-100	38	26-58	37	17-53	30	11-47
Continental shelf dolphins[b]	63,361	42,898-87,417	13	9-19	4	2-6	6	3-8	5	2-7
Bottlenose dolphin oceanic	8,467	4,285-16,731	10	5-10	3	1-5	5	2-6	4	1-6
Sperm whale	1,635	1,132-2,359	16	11-23	6	2-8	7	3-10	6	2-9
Bryde's whale	26	12-56	48	23-100	17	7-24	22	10-31	18	7-28
Beaked whales[c]	1,167	643-2,117	12	7-22	4	2-6	5	3-8	4	2-7
Clymene dolphin	3,228	1,558-6,691	7	3-15	2	1-4	3	2-5	3	1-4
False killer whale	316	121-827	18	7-48	6	3-9	8	4-12	7	3-11
Melon-headed whale	1,696	709-4,060	15	6-36	5	2-7	7	3-10	6	2-9
Pantropical spotted dolphin	33,382	25,489-43,719	20	15-26	7	3-10	9	4-13	7	3-11
Short-finned pilot whale	1,641	710-3,790	6	4-9	2	1-3	3	1-4	2	1-3
Pygmy killer whale	281	131-601	15	7-33	5	2-8	7	3-10	6	2-9
Pygmy/dwarf sperm whales[d]	6,690	3,482-12,857	15	8-29	5	2-7	7	3-10	6	2-9

Cetacean Stock	Pre-spill Abun-dance Estimate	95% CI	Popula-tion Exposed to Oil (%)	95% CI	Popula-tion Killed (%)	95% CI	Females with Repro-ductive Failure (%)	95% CI	Population with Adverse Health Effects (%)	95% CI
Risso's dolphin	1,848	1,123-3,041	8	5-13	3	1-4	3	2-5	3	1-4
Rough-toothed dolphin	2,414	964-6,040	41	16-100	14	6-20	19	9-26	15	6-23
Spinner dolphin	6,621	3,386-12,947	47	24-91	16	7-23	21	10-30	17	6-27
Striped dolphin	2,605	1,537-4,415	13	8-22	5	2-7	6	3-9	5	2-8

[a] Based on 2 years of observations; all other stocks based on 3 years of observations.

[b] Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

[c] Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

[d] Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

Table 4.9-13. This table summarizes the injuries to northern Gulf of Mexico cetaceans caused by the DWH oil spill, including lost cetacean years, years to recovery, and maximum population reductions.

Cetacean Stock	Lost Cetacean Years	95% CI[a]	Years to Recovery[b]	95% CI[a]	Maximum Population Reduction (%)	95% CI[a]
Bottlenose dolphin Barataria Bay	30,347	11,511-89,746	39	24-80	-51	32-72
Bottlenose dolphin Mississippi River Delta	20,065	4,896-62,355	52	27-106	-71	40-97
Bottlenose dolphin Mississippi Sound	78,266	30,858-219,602	46	27-89	-62	43-83
Bottlenose dolphin Mobile Bay	9,362	3,429-32,356	31	18-65	-31	20-51
Bottlenose dolphin western coastal	19,041	6,869-64,245	NA	NA	-5	3-9
Bottlenose dolphin northern coastal	92,069	36,427-264,716	39	23-76	-50	32-73
Continental shelf dolphins[c]	359,996	NA	NA	NA	-3	NA
Bottlenose dolphin oceanic	37,668	NA	NA	NA	-4	NA
Sperm whale	13,197	NA	21	NA	-7	NA
Bryde's whale	705	NA	69	NA	-22	NA
Beaked whales[d]	7,838	NA	10	NA	-6	NA
Clymene dolphin	12,167	NA	NA	NA	-3	NA

Cetacean Stock	Lost Cetacean Years	95% CI[a]	Years to Recovery[b]	95% CI[a]	Maximum Population Reduction (%)	95% CI[a]
False killer whale	3,422	NA	42	NA	-9	NA
Melon-headed whale	14,887	NA	29	NA	-7	NA
Pantropical spotted dolphin	363,780	NA	39	NA	-9	NA
Short-finned pilot whale	5,304	NA	NA	NA	-3	NA
Pygmy killer whale	2,501	NA	29	NA	-7	NA
Pygmy/dwarf sperm whales[e]	49,100	NA	11	NA	-6	NA
Risso's dolphin	6,258	NA	NA	NA	-3	NA
Rough-toothed dolphin	50,464	NA	54	NA	-17	NA
Spinner dolphin	188,713	NA	105	NA	-23	NA
Striped dolphin	18,647	NA	14	NA	-6	NA

a Confidence intervals for shelf and oceanic animals were not calculated (see DWH MMIQT 2015 for details).

b It was not possible to calculate YTR for stocks with maximum population reductions of ≤ 5% (see DWH MMIQT 2015 for details).

c Continental shelf dolphins is a combination of shelf bottlenose dolphins and Atlantic spotted dolphins.

d Beaked whales is a combination of Blainville's beaked whales, Cuvier's beaked whales, and Gervais' beaked whales.

e Pygmy/dwarf sperm whales is a combination of pygmy sperm whales and dwarf sperm whales.

4.9.5.5 Qualitative Considerations

Due to the scope of the spill, the magnitude of potentially injured populations, and the difficulties and limitations of working with marine mammals (e.g., MMPA regulations), it is impossible to quantify injury without uncertainty. In evaluating these injuries, however, the Trustees have considered the following:

- Marine mammals were clearly observed in DWH oil.
- Population boundaries (i.e., stock ranges) overlap with the spill.
- Oil has negative physiological effects on wildlife, including marine mammals.
- The Trustees documented negative physiological effects on live dolphins examined in Barataria Bay and Mississippi Sound and dead, stranded dolphins from Louisiana, Mississippi, and Alabama.

The Trustees have additionally supplemented these observations and findings with the best available scientific literature, as well as their professional judgment and expert opinions in identifying the most appropriate assumptions and extrapolations to characterize the magnitude of injury (for each species and the temporal and spatial extent of the spill). Wherever possible, the quantification results presented herein represent ranges of values that encapsulate the uncertainty inherent in the underlying datasets. However, a variety of qualitative information cannot be captured in uncertainty estimates or ranges.

Even taking into account the inevitable uncertainty associated with assumptions and extrapolations, as well as the protected status of all marine mammals under the MMPA (and sperm whales under the ESA), the Trustees believe that the results of the injury assessment conducted over the past 5 years, coupled with the professional judgment used in quantifying these injuries, provides sufficient basis for identifying restoration activities. These restoration activities will appropriately compensate the public for injuries to marine mammals as a result of the DWH oil spill.

The Trustees have determined that estimating the total number of lost cetacean years, or LCY, for each stock is the best metric of the damage to marine mammals in the northern Gulf of Mexico resulting from the DWH oil spill. This value best reflects the long-term injury to each stock. A single calculation of dead dolphins could be misconstrued to represent injuries that occurred in a narrow time frame and, therefore, could be restored in a narrow time frame. The population model outputs best represent the temporal magnitude of the injury and the potential recovery time from the injury. As is evident from these calculations, some stocks will suffer from the effects of the spill for decades.

The quantification of injury is based on the extent of surface oiling across the Gulf of Mexico, because 1) many of the adverse health effects and causes of death for stranded animals were likely related to inhalation or aspiration of oil components in the surface slick, and 2) little is known about the fate and transport of DWH deep-sea oil plumes in relation to deep-diving marine mammals, such as sperm whales. The characterization of marine mammal exposures to contaminated sediments, both nearshore and offshore, is similarly uncertain.

4.9.6 Conclusions and Key Aspects of the Injury for Restoration Planning

Immediately following the DWH blowout, it was clear that the oil spill would jeopardize the marine mammal communities in the northern Gulf of Mexico. The Trustees developed a suite of studies and analyses to assess the exposure and injuries to dolphins and whales from the DWH oil spill. Response workers and scientists observed marine mammals swimming in the DWH surface slick, but animals would have also been exposed via contaminated air and sediments, as well as oil in the water column. After inhaling, ingesting, aspirating, and potentially absorbing oil components, animals suffered from physical damage and toxic effects to a variety of organs and tissues. Marine mammals living in areas contaminated with DWH oil suffered from lung disease, adrenal disease, poor body condition, and a suite of other adverse health effects. Unsurprisingly, at the same time as veterinarians were documenting these illnesses, NOAA declared and was investigating the largest and longest Gulf of Mexico cetacean UME on record. In the years following the spill, many of these stranded animals were fetuses from unsuccessful pregnancies, and more than 80 percent of cetacean pregnancies in Barataria Bay and Mississippi Sound were unsuccessful (DWH MMIQT 2015).

Based on their scientific findings, and after carefully ruling out other potential causes for the unprecedented number of dead, stranded marine mammals, the Trustees concluded that the DWH oil spill is the most likely explanation for the combination of injuries and mortalities seen across the marine mammal populations in the northern Gulf of Mexico. The routes of exposure, adverse health effects, increased mortality, reproductive failure, and causes of death for stranded animals form a coherent pathological narrative, consistent with the chemistry and toxicity of oil transport and exposure, and the relationship between the levels of oil exposure and the severities of injuries across the spatial and temporal extent of the DWH oil spill event.

Common bottlenose dolphins in the Barataria Bay, Mississippi River Delta, Mississippi Sound, and Mobile Bay BSE stocks suffered some of the most severe injuries to both individual animals and their populations going forward. The northern and western coastal bottlenose dolphin stocks and most of the shelf and oceanic marine mammal stocks suffered quantifiable injuries. Other bottlenose dolphin stocks, including the Terrebonne-Timbalier Bay stock, which saw extensive DWH oiling, and the Perdido Bay and Pensacola Bay stocks, which saw less oil, but were within the DWH oil spill footprint, were likely injured, but data were too sparse to determine the relationship between DWH oil exposure and potential injuries.

In summary, the injury assessment found that:

- The DWH oil spill resulted in the contamination of prime marine mammal habitat in the estuarine, nearshore, and offshore waters of the northern Gulf of Mexico.
- Nearly all of the marine mammal stocks that overlap with the DWH oil spill footprint have demonstrable, quantifiable injuries. The remaining stocks within the footprint were also likely injured, but there is not enough information to make a determination at this time.
- The Barataria Bay and Mississippi Sound bottlenose dolphin stocks were two of the most severely injured populations, with a 51 percent and 62 percent maximum reduction in their population sizes, respectively. Dolphins are long-lived animals, and slow to reach reproductive

maturity, and these stocks will take approximately 40 to 50 years to recover, without any active restoration (DWH MMIQT 2015).

- Smaller percentages of the oceanic stocks were exposed to DWH oil. However, they still experienced increased mortality (as high as 17 percent), reproductive failure (as high as 22 percent), and other adverse health effects (as high as 18 percent) (DWH MMIQT 2015).

The Trustees considered all of these aspects of the injury in restoration planning, and also considered the ecosystem effects and recovery information described below.

4.9.6.1 Ecosystem Effects

Carnivorous cetaceans such as dolphins and sperm whales, which are typically apex predators, will suffer from DWH oil's effects on fish and invertebrate populations. At a more subtle, but still crucial, level, the summed negative effects of the DWH oil spill on the Gulf of Mexico ecosystem across resources, up and down the food web, and among habitats, will especially impact marine mammals due to their long lives and their strong dependence on a healthy ecosystem (Bossart 2011; Moore 2008; Reddy et al. 2001; Ross 2000; Wells et al. 2004).

4.9.6.2 Recovery

In the absence of active restoration, marine mammal stocks across the northern Gulf of Mexico will take many years to recover. Whales and dolphins are slow to reach reproductive maturity, only give birth to a single offspring every 3 to 5 years, and are long lived (with lifespans up to 80 years). Therefore, it will take decades for the Gulf of Mexico stocks to recover from losses following the spill. The ability of the stocks to recover and the length of time required for that recovery are tied to the carrying capacity of the habitat, and to the degree of other population pressures. The fact that not enough is known about the pressures, or stressors such as human impacts and natural events, that may adversely affect these animals makes understanding the timeframe required for stocks to recover even more challenging.

4.9.6.3 Restoration Considerations

As described in Chapter 5 (Section 5.5.11), the Trustees have identified an integrated portfolio of restoration approaches that collectively address all stocks, species, and geographies that were injured by the spill, taking into account the long-lived nature of these animals and the stressors that adversely affect them. The restoration portfolio for marine mammals will also include robust monitoring, analysis, and scientific support for an adaptive management approach to restoration planning and implementation to ensure that restoration goals are met.

4.9.7 References

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4.10 Lost Recreational Use

What Is in This Section?

Executive Summary

The Gulf of Mexico is a popular destination for many types of recreation. The *Deepwater Horizon* (DWH) oil spill resulted in losses to the public's use of natural resources for outdoor recreation, such as boating, fishing, and going to the beach. These spill impacts in the Gulf of Mexico started in May 2010 and lasted through November 2011. The Trustees conducted a number of studies to measure the lost recreational value to the public due to the spill. Recreational use was evaluated in coastal areas of Texas, Louisiana, Mississippi, Alabama, and Florida, and losses were evaluated for residents throughout the contiguous United States.

The Trustees estimated that the public lost 16,857,116 user days of boating, fishing, and beach-going experiences as a result of the spill. Losses occurred across multiple years, and the final estimates were compounded to 2015 using a 3 percent interest rate and adjusted to 2015 price levels.[1] Total recreational use damages due to the spill are estimated to be $693.2 million with uncertainty ranging

[1] All dollar values in the section are presented in July 2015 price levels unless otherwise noted.

from $527.6 million to $858.9 million.[2,3] This lost value does not include losses to private businesses/individuals or lost tax revenue to municipalities, which are not compensable under NRDA regulations.

This section provides an overview of the Trustees' assessment approach for recreational losses. Additional details are available in a set of technical memos provided in the Administrative Record.[4]

4.10.1 Introduction

The Gulf of Mexico is a popular destination for a wide variety of recreational activities, which draw people not only from the region but from all across the country. Activities including boating, fishing, and beach-going depend directly on the environmental quality of the Gulf of Mexico's natural resources and the ability to access them. When an event such as the DWH oil spill degrades the quality of shorelines and water resources, there are severe impacts to recreational use.

Key Findings

- The DWH oil spill resulted in losses to the public's use of natural resources for outdoor recreation, such as boating, fishing, and going to the beach.
- These spill impacts in the Gulf of Mexico started in May 2010 and lasted through November 2011.
- The Trustees estimate the value of recreational losses to the public from the oil spill to be $693.2 million with uncertainty ranging from $527.6 million to $858.9 million.
- As a result of the spill, the public lost over 16 million user days of boating, fishing, and beach-going experiences.

Following the spill, the presence of oil on beaches or in the water degraded that quality and/or accessibility. For example, some beaches were closed due to oiling or cleanup activities while others remained open with posted advisories. Furthermore, even the expectation of oiling caused individuals to cancel planned trips to coastal areas. The oil spill affected recreation in the Gulf of Mexico through people cancelling trips, people choosing alternate sites for recreation, and people experiencing a reduction in the quality of their recreational activities.

4.10.1.1 Recreational Use

In assessing the lost use damages due to the spill, the Trustees measured impacts to two broad categories of recreation: shoreline use and boating.

[2] An approximation of the 95% confidence interval for this estimate is derived by adding a point estimate for the Tier 2 subset of total recreational use damages to the upper and lower 95% confidence interval of the Tier 1 recreational use damages, recognizing that the statistical uncertainty of the Tier 2 estimates is unknown.
[3] In comparison, in the 2007 *Cosco Busan* oil spill in San Francisco Bay, Trustees assessed approximately 1 million lost recreational user days valued at $22.2 million (2010 U.S.). In the 1990 *American Trader* oil spill in southern California, Trustees assessed 0.76 million lost recreation user days valued at $12.2 million (1990 U.S.).
[4] The data collected by Trustees for the lost recreational use assessment is available at *https://dwhdiver.orr.noaa.gov*.

- **Shoreline use** refers to recreational activities conducted by individuals at locations near beaches and other shoreline areas. These activities include swimming, sunbathing, surfing, walking, kayaking, and fishing that takes place from the shore or shoreline structures such as piers.

- **Boating** includes a variety of recreational boating activities that begin at sites providing access to salt water near the Gulf Coast, including marinas, unimproved launches, and private residences. Boat-based fishing is included in this category.

The Trustees measured the severity, extent, and duration of the adverse effects to recreational activities by evaluating the public's recreation patterns and determining how those changed because of the spill.

A "user day" refers to any time an individual visits a beach, goes fishing, or goes boating for the purpose of recreation for at least part of the day.

The Trustees began conducting studies on recreational use in the Gulf of Mexico shortly after the spill began and continued to do so for up to 3 years for many types of activities and locations. These studies yielded estimates of visitation which, when compared to the amount of visitation that would have occurred without the spill, determined the number of lost user days. This measure is an indicator of the severity of the impact from the spill across both space and time. Trustees evaluated factors other than the spill that could have affected coastal recreation (such as weather and macroeconomic conditions) and accounted for them with adjustments to predictions of baseline when necessary. The result is an estimate of lost user days that is exclusively attributable to the spill.

4.10.1.2 Defining Public Losses

To determine recreation damages, the Trustees measured the loss in recreation value provided by Trust resources to the public. The Oil Pollution Act of 1990 (OPA) defines value as "the maximum amount of goods, services, or money an individual is willing to give up to obtain a specific good or service" (15 CFR §990.30). This reduction in value is calculated for all individuals who potentially use the public resources for recreational purposes. The Trustees used the estimates of lost user days combined with models of recreational demand to measure the lost recreational value due to the spill. This approach generated a value per lost user day that includes three behavioral responses to the oil spill presented in Figure 4.10-1:

- Canceling trips (lost trips).

- Taking trips to different locations (substitute trips).

- Continuing to take trips to the same locations, but under conditions that reduce enjoyment (diminished-value trips).

This approach also takes into account the fact that some individuals are unaffected by the spill and do not contribute to the estimate of damages.

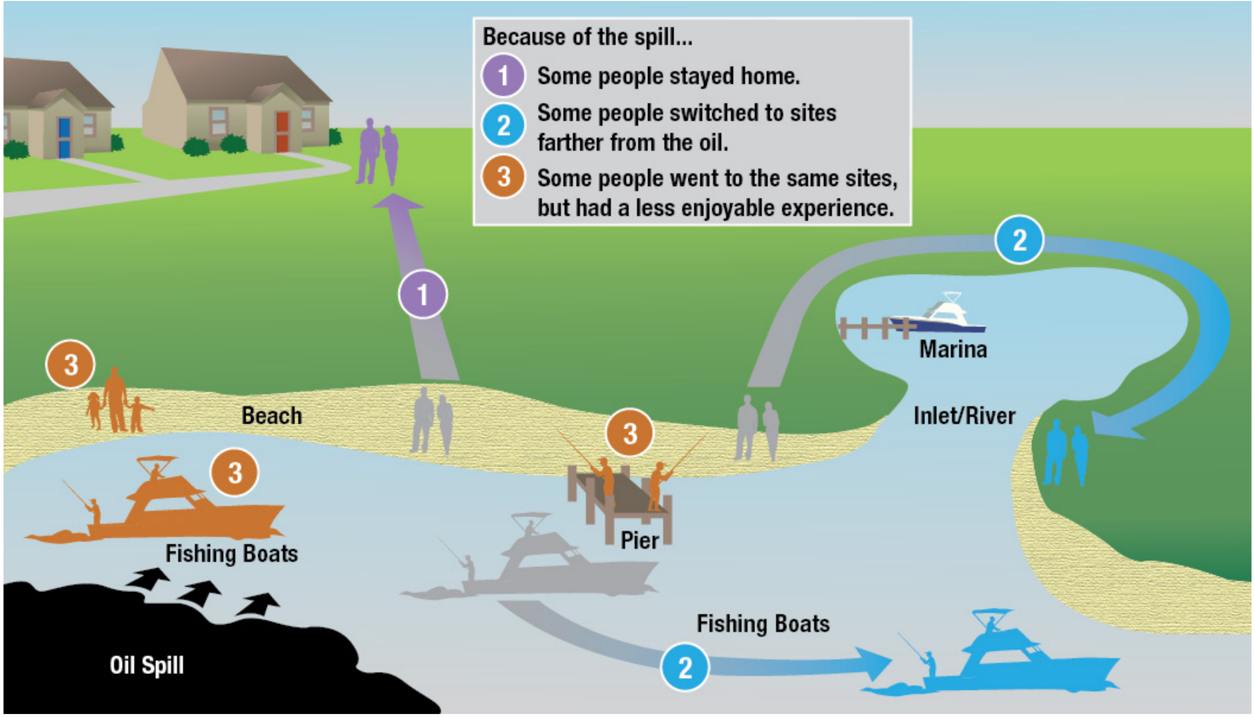

Figure 4.10-1. Recreational users reacted to the oil spill in different ways.

4.10.2 Economic Damages

Marine and coastal natural resources of the Gulf of Mexico provide recreational services to individuals from across the United States and around the world. These services have economic value, just like services from private amusement facilities such as Walt Disney World. Many natural resources are publicly owned, so the services they provide do not have observable prices as do goods and services in monetary economic transactions. For example, goods in a grocery store have a dollar price, and a monetary transaction takes place when those goods are sold. Although recreational services from public resources are not sold on markets, they still have value.

When an oil spill impacts natural resources, it diminishes the value of the recreational services they provide. The reduction in the number of people using the affected resources is an important determinant of that lost value. Another important indicator of lost value is the distance people travel and the expenses they incur to reach the resources under unoiled conditions. Economic models can use this type of information to estimate the lost recreational value attributable to the spill.

A spill can also cause other types of economic losses, both public and private, which are outside the scope of the Trustees' assessment. Private losses are generally associated with declines in business profits, lost wages, or costs of repairing property damage caused by the spill. Public economic losses that are outside the scope of the Trustees' assessment include losses in revenue from taxes or fees collected by local municipalities. Impacts to habitats and wildlife represent losses in value that are part of the Trustees' assessment; these impacts are addressed in other sections of Chapter 4. Table 4.10-1 lists examples of economic losses potentially recoverable under OPA (Natural Resource Trustee Claims) and those that are not (Other Claims). (This is not an exhaustive list.)

Table 4.10-1. OPA categories of potential economic damages from an oil spill.

Natural Resource Trustee Claims	Other Claims
Lost recreational use	Lost business profits
	Lost government taxes
Impacts to habitat and wildlife	Royalties, fees, rents
	Lost wages
	Injury to real or personal property
	Costs of increased public services to address the spill

4.10.3 Characterization of Injury

The damage assessment conducted by the Trustees for the DWH oil spill combines two studies addressing different components of lost recreational use damages. One study measures lost user days through a series of in-person surveys, infield counts, and counts of users in aerial photographs. The second estimates lost value per lost user day by modeling demand for recreation, using information collected in mail and telephone surveys. Figure 4.10-2 illustrates the overall assessment approach, described in detail throughout Section 4.10. As background, Section 4.10.3 describes the information gathered on the extent of activities potentially affected, restrictions on recreation activity, and public awareness of the spill.

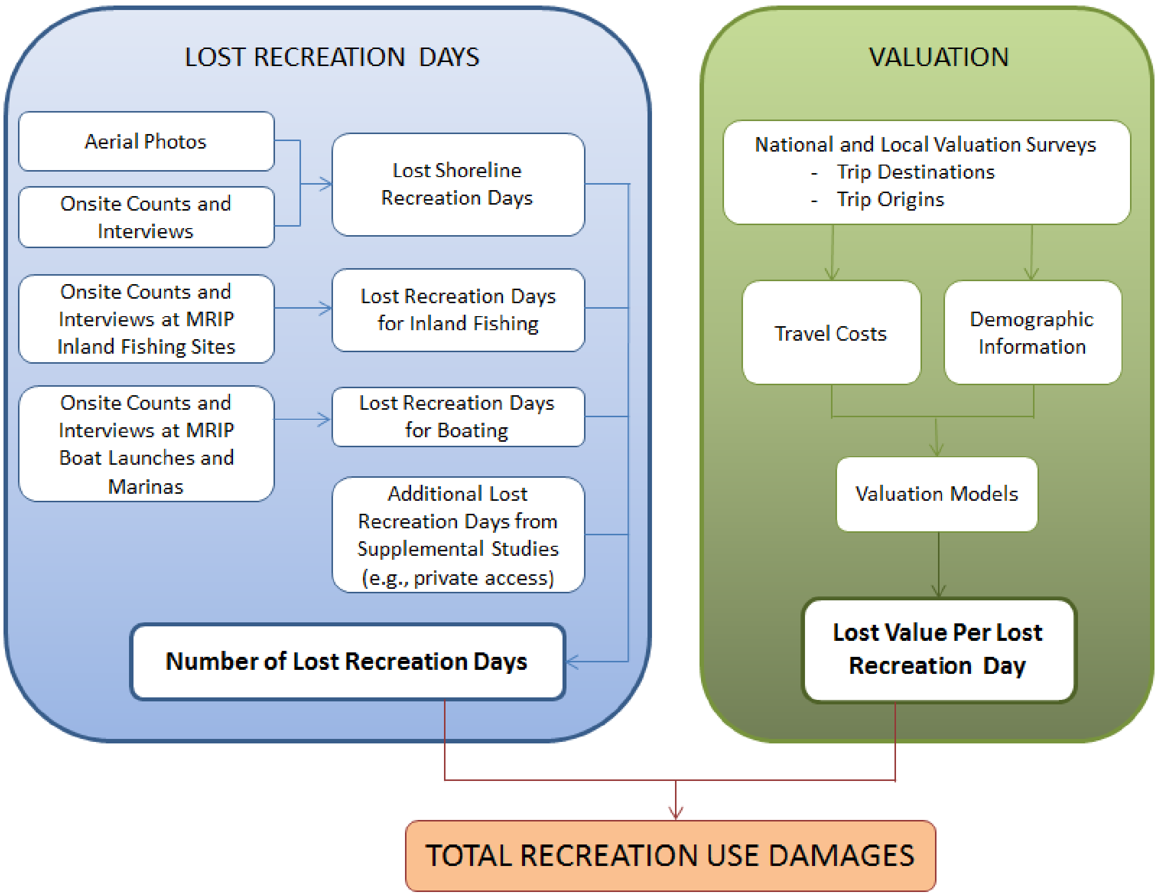

Figure 4.10-2. The Trustees used a number of different data sources to estimate the total recreational use damages.

4.10.3.1 Scope of Spill Impacts

4.10.3.1.1 Recreation in the Gulf Before the Spill

To help understand the levels of recreational activities prior to the spill, the Trustees turned to several federal government studies that have been ongoing for decades.

Data collected by NOAA as part of the Marine Recreational Information Program (MRIP) indicate that from 2000 to 2009 there were on average about 13.6 million annual saltwater boat fishing trips and 8.7 million annual saltwater shore fishing trips to locations along the Gulf of Mexico from Louisiana to Florida (Welsh 2015a).

The U.S. Fish and Wildlife Service periodically conducts a National Survey of Fishing, Hunting, and Wildlife-Associated Recreation. This survey provides an estimate of the number of anglers taking trips to saltwater locations and the number of days spent fishing at saltwater locations on a state-by-state basis. The last phase of the survey prior to the spill estimated that in 2006 there were 2.5 million anglers who spent a total of 27.4 million user days visiting saltwater locations in Louisiana, Mississippi, Alabama, and Florida.[5]

Data from the National Survey on Recreation and the Environment, conducted between 2005 and 2007, indicate that there were about 191.8 million saltwater-related recreation user days in Louisiana, Mississippi, Alabama, and the Gulf Coast of Florida annually. A wide range of activities were associated with these saltwater recreation trips, including water contact activities (swimming, snorkeling, scuba diving, and surfing), boating (motorboating, sailing, and using personal watercraft), fishing, hunting, and nature/wildlife viewing.

4.10.3.1.2 Public Awareness of the Spill

Due to widespread press coverage, there was a high level of public awareness of the spill. Beginning in April 2010, spill-related information was provided by a Joint Information Center managed by the U.S. Coast Guard. From April 21, 2010, through February 2013, the Joint Information Center issued over 600 press releases about the spill, with the great majority of these in 2010.

These press releases were provided to media outlets and, along with other sources of information, formed the basis for numerous television, newspaper, radio, and online reports about the spill. A general indication of public awareness can be seen in a Pew Research Center study conducted in late-July 2010, in which 59 percent of respondents said they were following news about the spill “very closely” (Welsh 2015c).

A CNN report on April 20, 2011, the first anniversary of the spill, discussed the popularity of the “spillcam” showing live feeds of the oil spill. This feed was first made available by the U.S. House of Representatives Select Committee for Energy Independence and Global Warming. CNN reported that a day after the live feed became available it had already been viewed a million times, and that traffic was so heavy it “temporarily crashed the House of Representative’s Web system” (Welsh 2015c).

[5] This estimate includes all saltwater locations in Florida, including the Gulf and Atlantic coasts.

The Vanderbilt Television News Archive provides a searchable database of news coverage, including a count of news items and the number of minutes of coverage related to particular topics. Information from this database indicates that television coverage in each of the 3 months following the spill included over 150 reports totaling over 900 minutes of airtime on six major national networks, as shown in Figure 4.10-3 below.

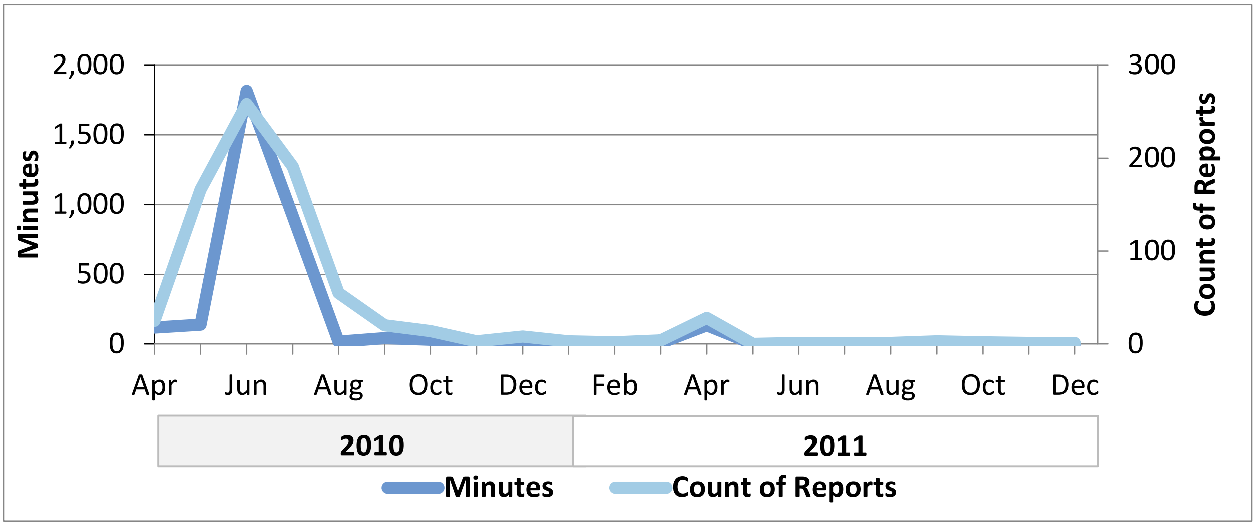

Figure 4.10-3. Television coverage of the oil spill, shown here in minutes and number of reports, helped create a high level of awareness following the spill. Data include ABC, CBS, CNN, FOX, MSNBC, and NBC (Welsh 2015c).

Other information about awareness of the spill was collected in a number of Trustee-directed studies, including the Infield Surveys and the Local and National Coastal Activity Surveys further described in the following section. In the Infield Surveys, over 99 percent[6] of shoreline recreators interviewed reported that they were aware of the oil spill. The Local and National Coastal Activity Surveys also found that over 99 percent of respondents either mentioned the oil spill during the course of the interview or said they had been aware of the spill when asked directly.

4.10.3.2 Impacts on Recreational Activities in the Gulf

Two survey-based investigations conducted by the Trustees provided evidence of the impact of the spill. The Local and National Coastal Activity Surveys measured spill awareness and the potential for spill impacts. The Infield Surveys were used to estimate lost user days, but they also provided supporting evidence of impact through brief onsite surveys of recreational users.

4.10.3.2.1 Evidence from Coastal Activity Surveys

The Trustees implemented two phone surveys in 2011 to collect information on the effects of the oil spill on Gulf Coast recreation. The Local Coastal Activity Survey collected information from residents of Louisiana, Mississippi, Alabama, and Florida. The National Coastal Activity Survey collected information from residents in the remaining contiguous United States.

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[6] The actual percentage varied over time and by activity.

In the Local Coastal Activity Survey, about 26 percent of respondents indicated that they had either stopped going to some Gulf Coast sites or had visited them less often because of the oil spill (Welsh 2015c). By extrapolating to the relevant populations, Trustees estimated that the recreation activities of about 2.5 million individuals in the four states surveyed had been affected by the spill.

In the National Coastal Activity Survey, about 10 percent of respondents said they had canceled a planned trip to the Gulf Coast as a result of the spill. By extrapolating to the population covered by the survey, Trustees estimated that about 2.3 million people had cancelled a trip to the Gulf Coast because of the oil spill. Of the subset of respondents who did take trips to the Gulf Coast, 10 percent said that the oil spill had affected their activities during at least one of their trips. By extrapolating to the relevant populations, Trustees estimated that about 1.2 million people had activities during Gulf Coast trips affected by the oil spill.

4.10.3.2.2 Evidence from Infield Surveys

The Trustees also conducted in-person surveys of individuals participating in recreation on the Gulf Coast after the oil spill. Survey respondents were asked whether the oil spill had affected their choice of location to visit and the activities in which they had participated.

A significant percentage of respondents reported that the spill had affected the location they chose and the activities in which they engaged. Typically, the percentage of respondents reporting these impacts was highest in the area ranging west from Gulf County, Florida, to the Louisiana/Texas border and was highest in the summer of 2010 (Welsh 2015c). For example, in July 2010 in this area, about 24 percent of shoreline recreators reported that the spill had affected the location they visited, the activities in which they chose to engage, or their enjoyment of the site they visited. This percentage fell to less than 10 percent by September 2010 and remained in the 0.3 to 7 percent range until the end of May 2013, when the Infield Surveys concluded. Since these surveys were conducted at recreation sites, they do not include any information about people who did not recreate because of the spill.

4.10.3.3 Extent of Injury

4.10.3.3.1 Activities Included in the Assessment

The lost recreational use assessment covered two broad categories of recreation: shoreline use and boating. Shoreline use refers to recreational activities conducted by individuals at locations near beaches and other shoreline areas, and includes swimming, sunbathing, surfing, walking, kayaking, and fishing from the shore or shoreline structures such as piers. It also includes fishing at sites that are considered coastal but are not directly on the beach. Specifically excluded from the shoreline use assessment are recreational boating, commercial activities, and oil spill response.

The second broad category, boating, includes individuals engaged in recreational boating activities that begin at sites providing access to salt water near the Gulf Coast. The term “sites” encompasses a wide variety of locations providing boat access to coastal waters, including marinas, unimproved launches, and private residences. Non-recreational boating activities, including commercial fishing, law enforcement/safety, and oil spill response, are excluded from this category. Figure 4.10-4 illustrates examples of affected shoreline and boating activities.

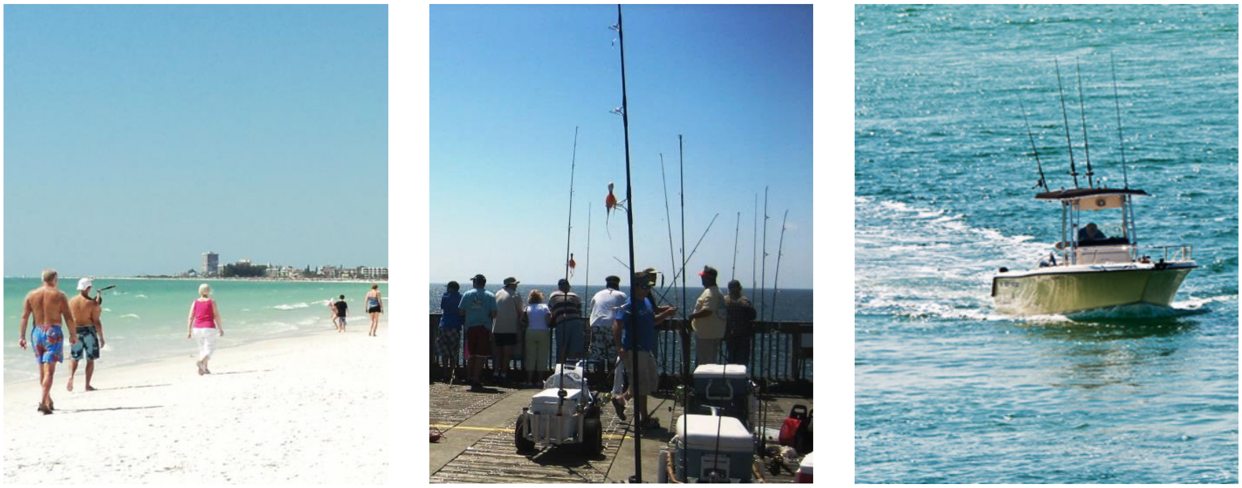

Figure 4.10-4. Beach use, fishing, and boating are examples of activities affected by the spill.

4.10.3.3.2 Activities Investigated but Not Included in the Assessment

Additional recreational activities, including guided hunting, commercially operated diving trips, and nighttime shoreline use, were investigated but not included in the assessment. Discussions with commercial operators of guided hunting and commercial diving ventures suggested some losses may have occurred, but since private business data are often confidential and difficult to obtain, Trustees did not assess these categories of loss. Trustees also did not assess a variety of nighttime shoreline uses that are difficult to measure. While Trustees used existing data to correct for night fishing (McConnell 2015a), non-fishing nighttime beach use was not assessed because of logistical and safety considerations. The fact that potential damages associated with these recreational activities are not included in the estimated total may indicate that the Trustee estimate of lost recreational use is a lower-bound estimate.

4.10.3.3.3 Geographical Extent of Assessment Activities

The oil spill directly affected a wide area of the Gulf Coast. Investigations conducted during and after the spill showed evidence of shoreline oiling from Texas through the Panhandle of Florida. Evidence of oiling was found more than 2 years after the spill. In addition, there was considerable concern that oil could come ashore along the Florida Peninsula. This possibility was identified in a press release provided by NOAA on July 20, 2010. During the spill, there was public awareness and a reasonable expectation that oiling could potentially occur as far south as the Florida Keys (Welsh 2015c).

This information indicated that potential recreational impacts could occur throughout the Gulf Coast from the Louisiana/Texas border to the Florida Keys. Trustees subsequently decided to sample recreation sites within this geographic area. For assessment purposes, the region was divided into the North Gulf (Louisiana/Texas border through Gulf County, Florida) and the Florida Peninsula (Franklin County, Florida, through the Florida Keys), as illustrated in Figure 4.10-5. Due to initial uncertainty on the geographic extent of impacts, the Trustees did not collect primary use data at recreational sites in Texas.

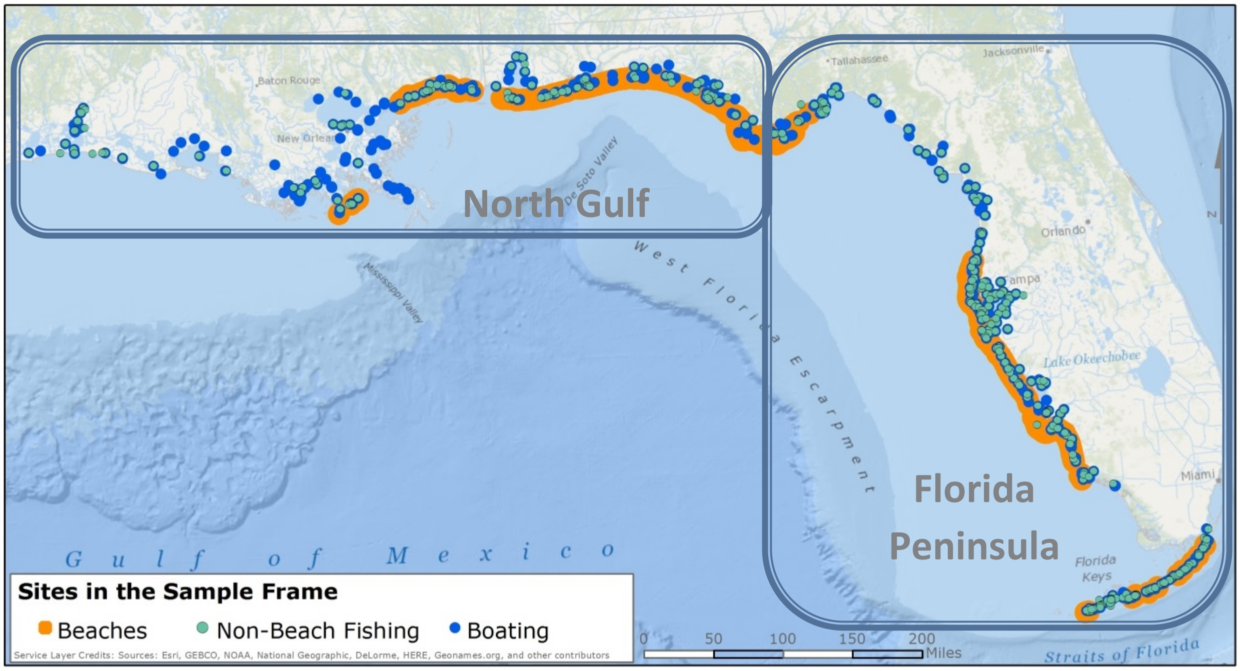

Figure 4.10-5. Trustees sampled a large number of recreation sites across the Gulf of Mexico. Each dot represents a separate site that was included in the sampling.

4.10.3.3.4 Duration of Spill Impact

The spill affected both the quantity and the quality of recreational trips to the Gulf Coast. A remaining critical issue is the time period, or duration, over which the recreational impacts occurred. The responses to the Infield Surveys show an expected pattern: a higher percentage of respondents surveyed in time periods closer to the spill reported impacts of the spill than respondents in time periods more removed from the spill. However, responses also suggested that the duration of the spill impacts could extend beyond the months and locations at which oiling was experienced.

The Trustees determined the duration of spill impacts by observing when recreational use data collected during the assessment (e.g., counts of shoreline users) suggested that recreational activities had returned to pre-spill levels. The impacts of both weather and general economic conditions were considered in evaluating the return to baseline conditions (Tourangeau & English 2015b). Based on this analysis, the Trustees observed that spill impacts for shoreline activities in the North Gulf started in May 2010 and continued through November 2011. In the Florida Peninsula, shoreline impacts started in June 2010 and continued through January 2011. For non-beach saltwater fishing (from piers and other non-beach shoreline areas), the spill impacts started in May 2010 and lasted through March 2011 and occurred only in the North Gulf. Spill impacts on boating (including boat-based fishing) started in May 2010 and lasted through August 2010 and occurred only in the North Gulf (Tourangeau & English 2015b). Figure 4.10-6 summarizes these impacts.

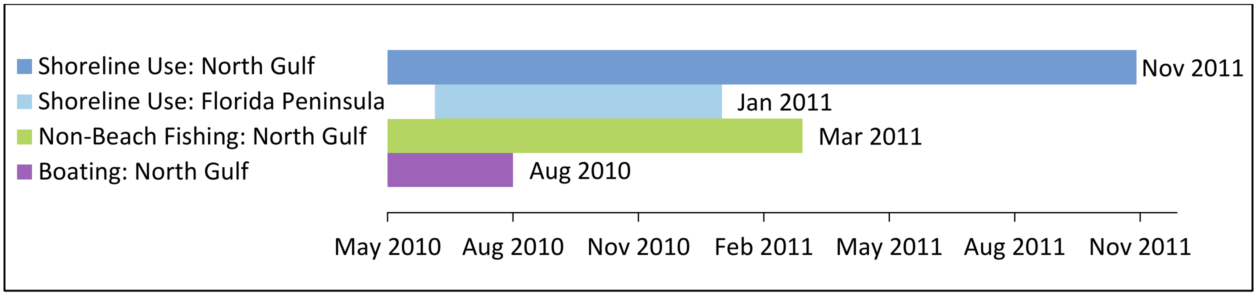

Figure 4.10-6. The duration of losses to recreational use differed depending on the region and activity.

4.10.4 Measurement of Lost User Days

Measurement of lost user days requires a comparison of the number of user days during the spill to the baseline number of user days that would have occurred without the spill. While several data collection efforts on recreational use in the Gulf of Mexico exist (Welsh 2015a), they are not comprehensive enough in space or time to establish a pre-spill baseline or to measure the full extent of losses caused by the spill. The Trustees determined that the best way to measure lost user days was to collect primary data on recreational use from the start of the spill until losses were no longer evident. The Trustees conducted three primary studies to measure the number of lost user days: the Shoreline Study, the Inland Fishing Study, and the Boating Study.

4.10.4.1 Direct Measurement

4.10.4.1.1 Shoreline Study

Trustees measured the number of recreators on sandy beach areas from Grand Isle, Louisiana, through the Florida Keys. Beach user days were estimated using a carefully designed systematic and random sample of overflights, onsite interviews, and onsite counts at 743 predefined beach segments generally less than 1 mile (1.6 kilometers) in length. Overflights were conducted by low-flying airplanes, while onsite interviews and counts were conducted by survey teams on foot.

Overflights of beaches along the Gulf of Mexico were scheduled for half of all weekdays and two thirds of all weekend days over a 3-year period from June 2010 through May 2013. Aerial photographs were taken of almost the entire beach coastline. Analysts used the photographs from each overflight to count the number of people on the beach and in the water for a sample of beach segments. Overflight photographs covered Gulf Coast beaches from Waveland, Mississippi, through Marco Island, Florida. Sampling teams on the ground counted beachgoers at segments in Grand Isle, Louisiana, and the Florida Keys.

Field teams conducted onsite interviews at selected beach segments. The interviewers randomly sampled recreators at preassigned beaches and asked questions about their visit, including the duration of their stay. Onsite count and interview data were collected on about one-fourth of the days selected for overflights. Given the absence of overflights in Grand Isle and the Florida Keys, onsite sampling was conducted more frequently in these locations.

The period of the day covered by interviews and overflights varied by time of year and other factors, but was generally either 7.5 hours or 9.5 hours. Throughout the Shoreline Study, 97,062 people were selected for interviews, with a response rate of 63.5 percent, and 84,687 segment-day pairs were selected for counts, with a response rate of 97.4 percent (Tourangeau et al. 2015b).

4.10.4.1.2 Inland Fishing Study

Trustees estimated the number of recreational anglers at fishing sites along the Gulf of Mexico from western Louisiana through the Florida Keys. Since anglers at sandy beaches were included in the Shoreline Study above, the Inland Fishing Study focused on recreational fishing taking place at non-beach saltwater access points (such as those on bays, inlets, and other tidal areas).

To select a sample of inland fishing sites, Trustees relied on a list of saltwater fishing access sites provided by MRIP. Of the 323 sites on the MRIP list, the sample included 49 sites in the North Gulf and 68 sites in the Florida Peninsula. Infield interview teams visited each sampling site an average of 1 weekday and 1 weekend day every 4 weeks from June 2010 through March 2013. At each site, teams counted the number of recreational anglers and randomly selected individual anglers for interviews. Throughout the Inland Fishing Study, 19,463 people were selected for interviews, with a response rate of 65.1 percent, and 9,202 site-day pairs were selected for counts, with a response rate of 96.4 percent (Tourangeau et al. 2015b).

4.10.4.1.3 Baseline and Shoreline Use Estimates

The Trustees used information from aerial counts and infield interviews to calculate monthly estimates of the number of people on the beaches and at fishing sites throughout the study. These estimates were then adjusted for weather and used to compare the actual level of visitation on the beaches with baseline (the level of visitation that would have occurred, but for the spill). Baseline can be determined by evaluating data on recreational use prior to the spill, or by directly measuring recreational use after observable spill impacts have ended. Trustees evaluated existing data collected by local, state, and national parks and resource management agencies throughout the Gulf of Mexico. While many of these data sources are useful at measuring use at specific sites, they do not provide uniform coverage across the entire Gulf of Mexico or for all years. Trustees thus opted to continue the Infield Surveys beyond the end of spill effects (Tourangeau & English 2015a, 2015b).

Post-spill recreation data were used as the basis for predicting baseline recreational use during the spill; however, weather, economic conditions, and other factors may have caused differences in recreational use across years. Trustees evaluated a number of factors that can affect recreational use and determined that only a weather adjustment was necessary to predict baseline use (Siikamaki 2015; Tourangeau & English 2015b).

The weather adjustment is a statistical procedure that makes the baseline and spill periods comparable. This adjustment ensures that the estimate of lost user days is fully attributable to the spill and does not reflect potential differences in weather.

The difference between the baseline user days and the user days that took place during the spill is the number of lost user days, according to the following equation.

Baseline User Days − User Days During Spill = Lost User Days

Table 4.10-2 shows estimated shoreline use during baseline and spill conditions, and the resulting lost user days, for the North Gulf and Florida Peninsula.

Table 4.10-2. Estimates of shoreline user days by region. (Standard errors, a statistical measure of uncertainty, are shown in parentheses.)

Spill Period /Region	Estimated Baseline User Days (a)	Estimated User Days During Spill (b)	Estimated Lost User Days (c=a-b)	Percent Decline Due to Spill (d=c/a x 100%)
Jun 2010–Jan 2011				
North Gulf	**14,207,507**	**7,782,270**	**6,425,237**	**45.2%**
	(737,483)	(565,853)	(944,623)	
Peninsula	**17,471,871**	**13,601,695**	**3,870,176**	**22.2%**
	(701,090)	(701,037)	(1,014,982)	
Feb 2011–Nov 2011				
North Gulf	**21,754,732**	**19,580,582**	**2,174,149**	**10.0%**
	(873,894)	(639,215)	(1,068,929)	
Total	**53,434,109**	**40,964,547**	**12,469,562**	**23.3%**
	(1,582,834)	(1,109,725)	(1,894,098)	

4.10.4.1.4 Boating Study

Trustees estimated the number of recreational boaters entering the Gulf of Mexico at sites along the coast from western Louisiana through the Florida Keys. As with the Inland Fishing study, Trustees relied on the MRIP list of saltwater boating access points open to the public. Of the 534 boating sites on the MRIP list, Trustees sampled 103 sites in the North Gulf and 90 sites in the Florida Peninsula. At each selected site, counts and interviews occurred on 1 weekend day and 1 weekday every 4 weeks from June 2010 through August 2012.[7] Throughout the Boating Study, 65,556 people were selected for interviews, with a response rate of 83 percent, and 11,488 site-days were selected for counts, with a response rate of 91.7 percent (Tourangeau et al. 2015b).

Table 4.10-3 shows estimated boating use during baseline and spill conditions, and the resulting lost user days, from the Boating Study.

Table 4.10-3. Estimates of boating user days. (Standard errors are shown in parentheses.)

Spill Period /Region	Estimated Baseline User Days (a)	Estimated User Days During Spill (b)	Estimated Lost User Days (c=a-b)	Percent Decline Due to Spill (d=c/a x 100%)
Jun 2010 – Aug 2010	**759,605**	**544,231**	**215,374**	**28.4%**
North Gulf	(53,556)	(49,880)	(72,944)	

[7] Sampling ended in June 2012 in the Florida Peninsula.

4.10.4.2 Adjustments for Coverage

The Shoreline, Inland Fishing, and Boating Studies provide reliable estimates of lost recreational use for the majority of times and geographical areas where spill impacts occurred. The coverage of these studies, however, does not include all relevant months, places, and times of day impacted by the spill. To address missing coverage, the Trustees performed a number of targeted supplemental analyses.

4.10.4.2.1 Early Data Collection

The primary Shoreline, Fishing, and Boating Studies were not fully implemented by the Trustees until June of 2010. In order to evaluate impacts to recreation in the weeks immediately following the spill but before the full studies were implemented, Trustees collected information at selected boating, fishing, and beach sites during May of 2010 (English 2015a). An additional 1,572,845 lost shoreline and 72,871 lost boating user days were added to the estimates from the primary studies to account for May 2010.

4.10.4.2.2 Supplemental Shoreline Study

The daily coverage of the primary Shoreline Study began at 10 a.m. and ended between 5:30 and 7:30 p.m., depending on location and time of year. The Trustees conducted a supplemental study to account for shoreline recreational use occurring during daylight hours outside of these time limits. Averaged over 2010 and 2011, the resulting adjustment factor increased the estimate of user days by 5.0 percent in the North Gulf and 18.9 percent in the Florida Peninsula (English 2015c). An additional 1,234,821 lost user days were added to the estimate from the primary studies.

4.10.4.2.3 Backyard Boating

The primary boating study did not include coastal waterfront residences or private marinas inaccessible to the public (or what is termed "backyard boating"). To capture boating user days originating from these locations, the Trustees performed a survey of registered boat owners in counties near coastal areas in Louisiana, Mississippi, Alabama, and Florida (Lupi 2015). The Trustees estimated that the spill resulted in a loss of 22,895 backyard boating user days.

4.10.4.2.4 Night Fishing

Neither the primary Shoreline and Inland Fishing Studies nor the supplemental shoreline study covered fishing at night. To correct for this undercoverage, the Trustees developed an adjustment factor that they applied to the estimates of lost fishing user days in the Shoreline and Inland Fishing Studies. The adjustment was based on data obtained from MRIP's Coastal Household Telephone Survey, which includes estimates of hourly fishing activity in the Gulf of Mexico over a 24-hour period (McConnell 2015a). Using this approach, an additional 152,517 lost user days were added to the estimate from the primary studies.

4.10.4.2.5 For-Hire Fishing

The estimates of lost user days from the primary studies explicitly excluded any boat-based trips that individuals take for a fee, such as for-hire boat fishing. The Trustees relied on data collected by MRIP to estimate lost user days associated with for-hire boat fishing (McConnell 2015b). An additional 216,089 lost user days for for-hire fishing were added to the estimate from the primary studies.

4.10.4.2.6 Federal Lands Outside of Sample Area

The primary Shoreline Study did not include estimates of losses at Ship Island, Fort Barrancas, and Advanced Redoubt, all of which are part of Gulf Islands National Seashore. These sites were excluded from the primary studies for logistical reasons. Data available from the National Park Service was used to measure spill impacts at these sites, resulting in an additional 23,276 lost user days (English 2015b).

4.10.5 Measurement of Value

To determine the total recreational losses, the Trustees had to measure the value of a lost user day. The Trustees estimated this value using the travel cost method, a technique that is common in the economics literature and is frequently applied in damage assessments (Herriges 2015). Travel cost valuation models were used to calculate the average value of a lost user day specific to the time periods and activities described in Table 4.10-2 and Table 4.10-3. While these user day values were applied specifically to lost user days, they incorporate value from all three types of responses to the spill described at the beginning of this section: lost trips, substitute trips, and diminished-value trips. Multiplying the number of lost user days by the value of a lost user day provides an estimate of the total recreational losses.

4.10.5.1 Valuation Surveys

The Trustees implemented two separate general population surveys to gather the data for the valuation models. The Local Valuation Survey targeted adults living in Louisiana, Mississippi, Alabama, Florida, and selected counties in Texas and Georgia (Lupi & Welsh 2015). The National Valuation Survey targeted adults living in the contiguous 48 United States, excluding the states and counties targeted in the Local Valuation Survey (Welsh 2015b). Both the Local and the National Valuation Surveys gathered data on respondents' recreation trips and demographic characteristics. Trustees received 296,842 completed mail surveys and conducted 43,335 follow-up telephone interviews during the course of the two valuation surveys. Figure 4.10-7 illustrates the sample areas for the two surveys.

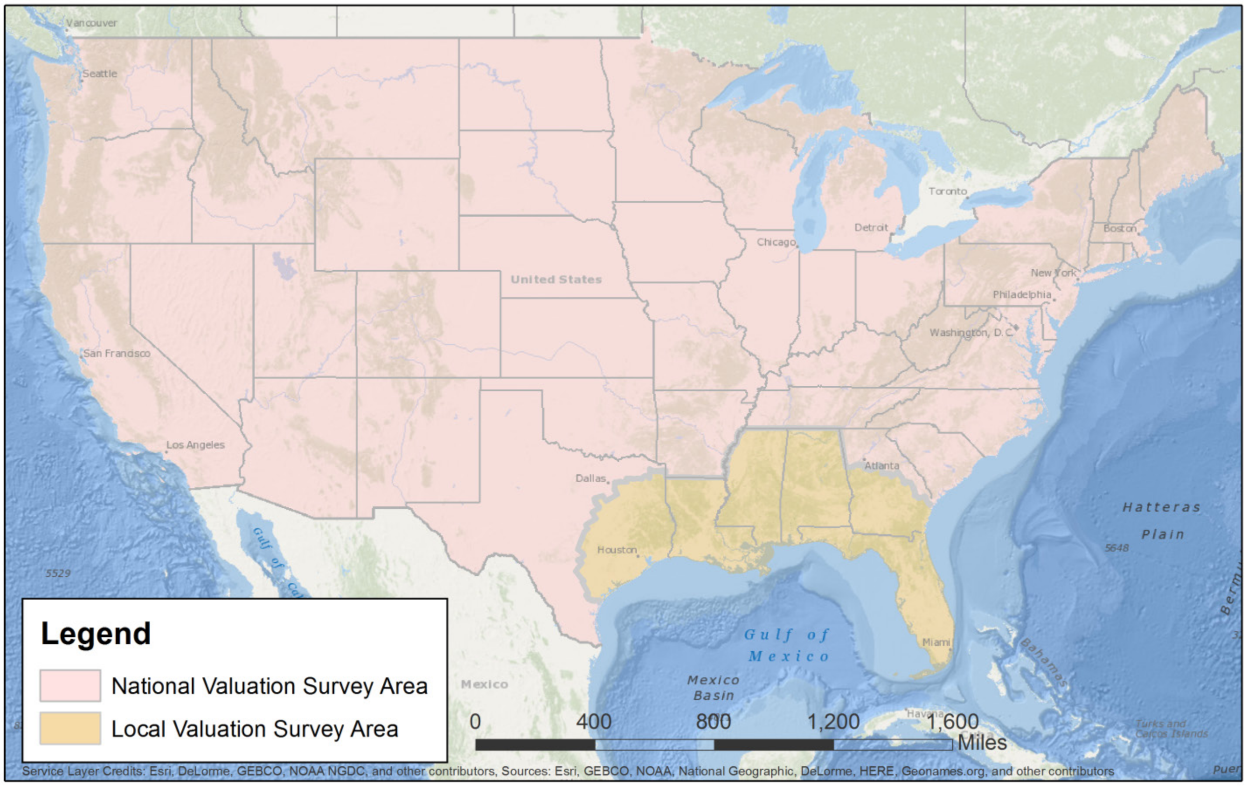

Figure 4.10-7. Trustees used two different valuation surveys covering the entire contiguous 48 United States to learn about people's trips to the Gulf of Mexico.

The National Valuation Survey was a combination mail and telephone survey that was implemented during 2012 and 2013. The telephone survey gathered data on all recreation trips to coastal areas of the United States that included a stay of 2 or more nights. Respondents to the National Valuation Survey reported their activities for a period covering 6 to 8 months prior to the interview.

The Local Valuation Survey was also a combination mail and telephone survey implemented during 2012 and 2013. The Local Valuation Survey, however, differed from the National Valuation Survey primarily by:

- Focusing exclusively on recreation trips taken within the Gulf Coast region, defined as coastal areas from Texas to Georgia.
- Requesting data on trips regardless of length, including single-day trips.
- Gathering data on boating trips separately from shoreline trips.
- Requesting trip data for only a 2- to 4-month period prior to the interview (Lupi & Welsh 2015; Welsh 2015b).

The Trustees combined data from both surveys to estimate the shoreline recreation valuation model. The boating valuation model only used data from the Local Valuation Survey, since the majority of private boating trips to the Gulf Coast originate from adjacent states.

4.10.5.2 Valuation Model Structure

When an event, such as an oil spill, occurs that reduces the quality of recreation at certain sites, a valuation model can be used to evaluate the impact of that event on recreation trips. As described previously, some of these trips will be diverted to alternative destinations, some trips will be canceled, some will be diminished in quality due to the event, and some may be unaffected. The valuation model converts all of these impacts to dollar values using baseline information about individuals' willingness to travel farther—and thereby incur additional costs—to avoid lower quality sites.

Two valuation models were used: one for all shoreline activities, including fishing; and another for boating. These models provide a quantitative description of people's recreation behavior. For example, the shoreline model describes the total number of recreation trips from throughout the contiguous United States to 83 shoreline areas in Texas, Louisiana, Mississippi, Alabama, Florida, and Georgia's Atlantic coast. The boating model describes boating trips to 67 sites in the same geographic area, but only includes trips originating in those six states.

For a given individual, recreation choices depend on demographic characteristics, the cost of traveling to the available recreation sites, and the quality of the available recreation sites. Trustees obtained data on shoreline recreation trips and demographic characteristics from the Local and National Valuation Surveys. The cost of traveling to recreation sites was calculated based on a combination of out-of-pocket costs (e.g., gasoline, depreciation, airline tickets) and the value of time spent traveling. Information about the relative quality of available recreation sites was determined within the model based on the relative number of trips and distances people travel to available sites.

The Trustees used the valuation models to determine the lost value per lost user day due to the spill (Herriges 2015). In each model, the Trustees simulated an event that led to a decline in the quality of all affected recreation sites. The magnitude of this event was carefully calibrated so that the overall percentage reductions in trips to the North Gulf and Florida Peninsula sites matched the percentage decline in user days due to the spill measured through the overflights and Infield Surveys (i.e., the percentages in Table 4.10-2 and Table 4.10-3). The model was then used to estimate the lost value associated with this calibrated event. The estimated loss incorporates lost, substituted, and diminished-value trips due to the event. The total loss in value estimated by the model was then divided by the number of the lost user days, resulting in an estimate of lost value per lost user day. For the shoreline model, the value per lost user day was $36.25, representing an average of all activities included in the shoreline model and over the two different periods of loss. For the boating model, the average value per lost user day was $16.20.

4.10.5.3 Fixed Costs of Boating

The valuation model developed for recreational boating, like most travel cost models, estimates people's willingness to pay for the opportunity to take boating trips net of any costs incurred to maintain a boat. However, there are also substantial fixed costs involved in boat ownership and maintenance. Because an oil spill involves transient impacts to recreation, boaters would not be expected to sell their boats in response to the spill. It is, therefore, not appropriate to net out the fixed costs associated with boat ownership. To obtain a more complete estimate of boating losses, however, the cost of owning a boat must be added to the lost value estimated by the boating valuation model

(English & Lupi 2015). Therefore, the Trustees divided the total annual fixed costs by the total annual number of saltwater boating days in the Boating Survey to obtain a per-trip increment of value of $7.90. Adding this amount to the average value per lost user day from the valuation model of $16.20 resulted in a total value per lost user day of $24.10.

4.10.5.4 Damages in Texas

Due to uncertainty about the potential for recreation impacts in Texas, the Trustees did not collect infield data at Texas sites. Although subsequent evaluation of existing data indicated that recreational use losses likely did occur in Texas, those data were not sufficient to generate a primary estimate of lost recreational use damages. Instead, the Trustees used information from the Local and National Coastal Activity Surveys to roughly evaluate potential lost recreational use in Texas. Results indicate that approximately 876,865 recreation trips to coastal sites in Texas were canceled due to the spill. Using the average lost value per lost user day calculated from the valuation surveys, damages may be approximately $31.8 million. This is the Trustees' best estimate of damages in Texas (Welsh & Horsch 2015).

4.10.6 Estimate of Damages

The Trustees calculated lost recreational use damages for a given period, activity, and region by multiplying the number of lost user days estimated from the Infield Surveys (with adjustments to coverage) by the value per lost user day measured by the valuation surveys (McConnell & English 2015). This calculation is summarized in the following equation.

$$\text{Lost User Days} \times \text{Value per Lost User Day} = \text{Recreational Use Damages}$$

Since damages occurred over time, the Trustees made two additional adjustments to represent damages in present value terms. Damages were adjusted for inflation from 2013, the year of data collection for the valuation surveys, to the current year. Damages were also compounded to account for the period between the date of injury and the present. Losses accrue interest at the rate of 3 percent per year and inflation is accounted for using the Consumer Price Index.

The damages exhibit statistical uncertainty due to sampling. The Trustees calculated the precision of estimates for the primary Shoreline and Boating Studies, but not for the subsequent adjustments to coverage. Table 4.10-4 shows the results of the primary studies.

Table 4.10-4. Estimates of damages due to lost shoreline and boating user days.

Damages (2015)	Lower Limit	Point Estimate	Upper Limit
Primary Shoreline Study	$354.23 million	$519.81 million	$685.38 million
Primary Boating Study	$1.01 million	$4.04 million	$7.06 million

The point estimate is the Trustees' best estimate of damages from the spill. Uncertainty in this estimate is represented by the upper and lower limits of approximate 95 percent confidence intervals.[8]

Table 4.10-5 shows additional losses based on the adjustments for missing coverage. It was not possible to estimate confidence intervals for these additional amounts, so they are added as fixed components to the lower limit, upper limit, and point estimates of total damages from Table 4.10-4.

Table 4.10-5. Damages from adjustments to coverage of primary studies.

Category	Source	Damages (2015)
Early Data Collection	Lost user days in May 2010 for shoreline activities, inland fishing, and boating.	$66,810,561
Supplemental Shoreline Study	Lost user days for shoreline activity outside of regular sampling hours.	$51,191,963
Backyard Boating	Lost user days for boating launched from private residences.	$429,994
Night Fishing	Lost user days for fishing outside sample period.	$6,357,177
For-Hire Fishing	Lost user days as measured through the MRIP for-hire fishing survey.	$9,003,910
Federal Lands Outside of Sample Area	Lost user days as measured using National Seashore visitation data.	$952,371
Fixed Costs of Boating	Underestimate of value due to fixed costs incurred in boating: incremental addition to the value per lost boating trip.	$2,848,632
Damages in Texas	Lost user days estimated from self-reported canceled trips.	$31,790,272

Table 4.10-6 presents total damages, incorporating results of the primary studies, adjustments to coverage, and damages in Texas.

Table 4.10-6: Total lost recreational use damages. (Numbers may not sum to totals due to rounding)

Damages (2015)	Lower Limit	Point Estimate	Upper Limit
All damages	$527.6 million	$693.2 million	$858.9 million

In the assessment, the Trustees examined a variety of uncertainties for their potential impact on damages. Some potential adjustments increase damages, others decrease damages. Memoranda included in the Administrative Record summarize the analyses of these sensitivities (Tourangeau et al. 2015a; Von Haefen et al. 2015).

[8] An approximation of the 95 percent confidence interval for this estimate is derived by adding a point estimate for the Tier 2 subset of total recreational use damages to the upper and lower 95 percent confidence interval of the Tier 1 recreational use damages, recognizing that the statistical uncertainty of the Tier 2 estimates is unknown.

In conclusion, the DWH oil spill resulted in a loss of recreational value. The Trustees estimate that 16,857,116 user days were lost, and these trips would have occurred along the coasts of Texas, Louisiana, Alabama, Mississippi, and Florida. The effects of the spill impacted recreation in the Gulf of Mexico as late as November 2011. In total, the Trustees estimate that the public lost $693.2 million (with uncertainty ranging from $527.6 million to $858.9 million) worth of recreational value as a result of the spill.

4.10.7 Conclusions and Key Aspects of the Injury for Restoration Planning

Impacts from the DWH oil spill, including oiled shorelines and closing of areas to recreation, resulted in losses to the public's use of natural resources for outdoor recreation, such as boating, fishing, and going to the beach.

The Trustees considered all aspects of the lost recreational use injury assessment in restoration planning to offset the losses, including:

- Spill impacts for shoreline activities in the North Gulf lasted for many months, starting in May 2010 and continuing through November 2011.
- Recreational losses due to the spill affected sites in the states of Texas, Louisiana, Mississippi, Alabama, and Florida. Residents throughout the contiguous United States were included as part of the affected public.
- The Trustees conducted a number of studies to measure the lost recreational value to the public due to the spill. The Trustees estimated that 16,857,116 boating, fishing, and other shoreline activity user days were lost throughout the five affected states. Total recreational use damages due to the spill are estimated to be $693.2 million with uncertainty ranging from $527.6 million to $858.9 million.

As described in Chapter 5 (see Section 5.5.14), the Trustees have identified a portfolio of restoration approaches for these injuries. These approaches include increasing recreational opportunities, improving habitats used for recreation, and using education and outreach to promote engagement in restoration and stewardship of natural resources.

4.10.8 References

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4.11 Injury Assessment: Summary and Synthesis of Findings

What Is in This Section?

- **Introduction (Section 4.11.1):** What events led to development of the injury assessment conclusions presented in Section 4.11?
- **Exposure to Oil and Response Activities Resulted in Extensive Injuries to Multiple Habitats, Species, Ecological Functions, and Geographic Regions (Section 4.11.2):** After assessing injury from the *Deepwater Horizon* (DWH) incident to representative habitats, processes, communities, resources, and services of the northern Gulf Coast ecosystem, what did the Trustees find?
- **Use of Inference to Assess Natural Resource Injuries Not Directly Measured by Trustees (Section 4.11.3):** How did the Trustees assess injury to natural resources not studied?
- **The Scope of Adverse Effects from the *Deepwater Horizon* Incident Constitutes an Ecosystem-Level Injury (Section 4.11.4):** What findings led the Trustees to conclude that the effects of the DWH incident constitute an ecosystem-level injury?
- **Treatment of Unquantified Injuries (Section 4.11.5):** How did the Trustees' injury assessment and restoration plan account for injuries they could not quantify?
- **References (Section 4.11.6)**

4.11.1 Introduction

The April 20, 2010, explosion, subsequent fire, and sinking of the DWH mobile drilling unit triggered a massive release of oil and other substances from the BP Macondo well. For 87 days after the explosion, the well continuously released oil into the northern Gulf of Mexico, ultimately releasing 3.19 million barrels (134 million gallons) of oil into the Gulf of Mexico.

The scope of the oil spill was unprecedented; it was the largest offshore oil spill in U.S. history. As the oil rose from the well, it spread over the sea surface. Carried by wind, wave, and tidal action, oil reached shoreline areas where it polluted beaches, bays, estuaries, and marshes from eastern Texas to the Florida Panhandle.

Source: *NASA (2010).*

Figure 4.11-1. The DWH oil spill was the largest offshore oil spill in U.S. history, releasing more than 3 million barrels (134 million gallons) of oil into the northern Gulf of Mexico over an 87-day period. This satellite image, taken on May 24, 2010, shows the spreading surface slick of oil approximately 50 miles off the Louisiana coast.

The DWH Trustees—the U.S. Department of Commerce; the U.S. Department of the Interior; the U.S. Environmental Protection Agency; the U.S. Department of Agriculture;[1] and designated agencies representing each of the five Gulf states (Alabama, Florida, Louisiana, Mississippi, and Texas)—undertook a natural resource damage assessment, or NRDA, to evaluate the nature and extent of adverse effects of the DWH incident on natural resources and their services. This assessment forms the basis of the Trustees' programmatic restoration plan.

As a result of this extensive, multi-year NRDA, the Trustees concluded that the DWH oil spill and related oil spill response actions caused a wide array of injuries to natural resources and the services they provide throughout a large area of the northern Gulf of Mexico (Sections 4.2 to 4.10). These conclusions were based on the scientific findings of the studies performed by the Trustees as part of the NRDA and on data collected during the oil spill response, together with supplemental findings published by the scientific community.

This section of the Final PDARP/PEIS summarizes the Trustees' injury assessment conclusions, which provide the basis for the programmatic restoration plan presented in Chapter 5.

Key Points

- The DWH spill resulted in a surface slick ultimately covering approximately 43,300 square miles (112,115 square kilometers), an area larger than the state of Virginia.
- Oil was pushed toward the shorelines of the Gulf states by currents, winds, and wave action. At least 1,300 miles (2,100 kilometers) of shoreline were exposed to oil from the spill. The extent of shoreline oiling exceeded the distance by road from New Orleans to New York City.
- The oil released into the environment was found to be toxic to a wide range of organisms, including fish, invertebrates, plankton, birds, and mammals, causing a wide array of toxic effects including death, disease, reduced growth, impaired reproduction, and physiological impairments that reduce the fitness of organisms (their ability to survive and reproduce).
- Concentrations of oil found to cause toxicity were exceeded in surface waters, sediments, and marsh habitats in many locations in the northern Gulf of Mexico. The degree and extent of these exceedances of toxic concentrations varied by location and time. The extent and degree of such exceedances has declined substantially from 2010 to the present.
- Natural resources were exposed to oil and other contaminants released as a result of the DWH incident over a vast area. Exposure to oil and response activities resulted in extensive injuries to multiple habitats, species, ecological functions, and geographic regions.

[1]The Department of Defense (DOD) also is a Trustee for natural resources associated with DOD-managed land on the Gulf Coast, which is included in the ongoing NRDA.

- The DWH incident resulted in injuries to marsh habitats, including marsh plants and associated organisms; to shoreline beaches and sediments and organisms that live on and in the sand and sediment; to fish and invertebrates that live in the water; to a wide range of bird species; to floating *Sargassum* habitats offshore and submerged aquatic vegetation; to deep-sea and nearshore ocean-bottom habitats, including to rare, deep water corals; to endangered and threatened sea turtles that live in the Gulf of Mexico; and to dolphins and other marine mammals.
- The spill directly reduced the public's use of popular recreational activities such as boating, fishing, and beach going between May 2010 and November 2011. The injuries caused by the DWH incident affected such a broad array of linked resources and services over such a large area that they cannot be adequately described at the level of a single species, a single habitat type, a single set of services, or even a single region. Rather, the effects of the DWH incident constitute an ecosystem-level injury. Consequently, the Trustees' restoration plan employs an ecosystem-level approach to address ecosystem-level injuries.

4.11.2 Exposure to Oil and Response Activities Resulted in Extensive Injuries to Multiple Habitats, Species, Ecological Functions, and Geographic Regions

The scale of the DWH spill was unprecedented, both in terms of the area affected and the duration of the spill. Due to the enormous scope of this incident, evaluation of all potentially injured natural resources in all potentially oiled locations at all times remains cost-prohibitive and scientifically impractical. The Trustees, therefore, undertook an ecosystem approach to injury assessment that included the evaluation of representative (see Section 4.1.3.1) habitats, ecosystem processes and linkages, ecological communities, specific natural resources, and human services. A summary of the Trustees' findings for these representative habitats and resources follows.

Key finding: Natural resources were exposed to oil and other contaminants released from the DWH incident over a vast area. (Section 4.2)

As described in Chapters 2 and 4, the release into the Gulf of Mexico of 3.19 million barrels (134 million gallons) of oil and 1.84 million gallons (almost 7 million liters) of dispersant resulted in extensive exposure of natural resources. For 87 days, BP's Macondo well released an average of nearly 37,000 barrels (1.5 million gallons) of fresh oil each day into the ocean. This is essentially equivalent to a substantial oil spill occurring every day for nearly 3 months.

- Combining direct observations, remote sensing data, field sampling data, and other lines of evidence, the Trustees documented that oil flowed within deep ocean water currents hundreds of miles away from the blown-out well; and that it moved upwards and across an area of the ocean surface. This movement resulted in observable slicks that covered an area of approximately 43,300 square miles (112,115 square kilometers), an area greater than the state of Virginia, affecting water quality and exposing aquatic biota. Oil was deposited onto at least 400 and possibly more than 700 square miles (1,030 to >1,910 square kilometers) of the sea floor and washed up onto at least 1,313 miles (2,113 kilometers) of shoreline, a distance greater than the road mileage between New Orleans and New York City.

- The estimated average daily volume of contaminated water under surface oil slick was 15 trillion gallons—approximately 40 times the average daily discharge of the Mississippi River at New Orleans.

- Natural resources were exposed to oil and dispersants across a broad range of habitats, including the deep sea, about 5,000 vertical feet of water column, the sea surface, and nearshore habitats such as beach, marsh, mangrove, and submerged aquatic vegetation.

- A wide variety of biota—ranging from those at the base of the food web to upper-level predators such as fish, sea turtles, marine mammals, and birds—were exposed to oil throughout the northern Gulf of Mexico. Natural resources were exposed through various pathways, including direct exposure to oil and dispersant, as well as contact with water, air, and sediments containing the contaminants.

- Despite being subject to natural weathering processes over the past 5 years, oil persists in some northern Gulf habitats, where it continues to expose species of natural resource value to residual contaminants.

Key finding: **Water column resources,** including fish, invertebrates, and *Sargassum*, were injured as a consequence of exposure to oil floating on the ocean surface; to oil mixed into the upper water column by wind and wave action and the addition of chemical dispersants; to oil as it moved from the wellhead to the surface; and to oil mixed into the deep sea. (Section 4.4)

- This exposure to oil at or near the surface occurred in an area of high biological abundance and high productivity during a time of year (spring and summer) that corresponds with peaks in seasonal productivity in the northern Gulf of Mexico.

- The Trustees determined that developing fish larvae exposed to the surface slick suffered almost 100 percent mortality, and, during the time period oil was present, oil concentrations in three water column zones—1) nearshore and offshore sea surface and upper mixed layer of the water column; 2) rising cone of oil from the wellhead; and 3) deep sea—exceeded levels known to cause mortality and sublethal effects to fish. Sublethal toxic effects can reduce an organism's health, fitness, and ability to reproduce and survive. These toxic effects were not uniform over the entire spill area; rather, they varied by location and time.

- Using information on oil toxicity and environmental exposures, the Trustees quantified the direct kill and production foregone of larval fish and invertebrates exposed to oil in the water column. The Trustees estimate that 2 trillion to 5 trillion larval fish and 37 trillion to 68 trillion invertebrates were killed in the surface waters as a result of floating oil and mixing of that oil into the upper water column. With respect to the deep waters, the Trustees' assessment showed that exposure to DWH oil resulted in the death of 86 million to 26 billion fish larvae and between 10 million and 7 billion planktonic invertebrates. Of these totals, 0.4 billion to 1 billion larval fish and 2 to 6 trillion invertebrates were killed in estuarine surface waters. This translates into a loss of from millions to billions of fish that would have reached 1 year of age. Additionally, the larval fish that were killed but would not have survived to age 1 are a significant loss; they are an energy source for other components of the ecosystem. The Trustees estimated that the lost larvae of just nine of the more than one thousand known fish species in the Gulf of Mexico would have developed into thousands of tons of adult fish had the spill not occurred.

- Available information indicates that the injuries, although substantial during the time oil was present, have not resulted in any apparent system-wide population crashes to surveyed fish or water column invertebrate species. However, while the populations of directly affected species appear not to have suffered a lasting impact, the death of such large numbers of larval fish and invertebrates represents a substantial short duration loss to the water column food web (see Section 3.6.1).
- In addition to the lethal injuries quantified by the Trustees, injuries to shelf-reef fish and fish communities were observed at a number of locations and over a range of benthic habitats. Injuries included reductions in abundance and changes in community composition. Although these various injuries cannot be explicitly quantified at this time, the Trustees concluded that fish and fish communities suffered physiologically and demographically important injuries in hard-bottom habitats along portions of the continental shelf. Species-specific data for red snapper, a key recreational and commercial species and a focus of intensive fisheries management efforts, indicate growth reductions, shifts in diet, and increased prevalence of lesions.
- The Trustees determined that *Sargassum,* a floating brown algae that creates essential habitat for invertebrates, fish, birds, and sea turtles, was injured as a result of exposure to oil. Trustees quantified both the loss of *Sargassum* resulting from direct oiling and also the area of *Sargassum* foregone due to lost growth. Based on this analysis, the Trustees determined that up to 23 percent of the *Sargassum* in the northern Gulf of Mexico was lost due to direct exposure to DWH oil on the ocean surface. In addition, foregone *Sargassum* area from lost growth due to exposure to this oil was estimated to be as large as 3,600 square miles.

Key finding: **Benthic resources** were injured over a variety of habitats and depths from the deep sea to the coastline. (Section 4.5)

Benthic resources live on, in, and in association with the bottom of the ocean. A wide variety of benthic organisms were injured as a result of the DWH incident, including hard and soft corals, small invertebrates, crabs, and fish that rely on benthic habitats.

- The Trustees documented a footprint of over 770 square miles around the wellhead within which different types and levels of injury to benthic resources were observed. This is greater than 20 times the size of Manhattan or nearly two-thirds the size of Rhode Island.
- The most severe injuries were observed closest to the wellhead, in an inner zone representing an area of approximately 11 square miles, where there was coral mortality and reduction of

infauna diversity and abundance. These injuries were caused by a combination of smothering by drilling muds and other debris materials, as well as exposure to oil, dispersants, and oil-associated marine snow. Exposure was confirmed through measurement of oil constituents (polycyclic aromatic hydrocarbons—PAHs) in megafauna collected from the deep-sea floor. Further, injury was confirmed with laboratory toxicity tests conducted with a deep-sea benthic organism (an amphipod, *Leptocheirus*). These tests showed that oil constituent concentrations (PAHs) measured in sediment samples collected within the inner zone exceeded levels sufficient to cause mortality to deep-sea benthos.

- The second and third concentric zones (covering areas of 75 and 306 square miles, respectively) exhibited different severities of injury, ranging from coral mortality at scattered hardground sites, to reductions in the diversity of sediment-dwelling animals that were significant, though less dramatic, than the reductions observed in the innermost zone. The second concentric zone, for example, experienced coral mortality and reductions in diversity in sediment-dwelling biota , but less extensively than in the innermost zone. Within the third zone, injuries were more patchy, but still ecologically significant. For example, injuries to 600- to 1,000-year-old hardground corals manifested over time, but injuries to sediment-dwelling biota were less severe than observed in zone 2. While the ecosystem functions of these unique deep-ocean corals are not well understood, their vertical structure likely covers and protects mobile biota seeking refuge from predators, and provides habitat for species to live and breed—similar to key ecological functions provided by other fan-like coral species growing in shallower habitats.
- In the fourth zone, the chemical quality of the seafloor habitat was adversely affected by contamination over 490 square miles. Specifically, sediments at some locations in this zone had PAH concentrations that exceeded values sufficient to cause mortality to amphipods that live in the deep-sea benthos. Further, some resident species, such as red crabs, had tissues contaminated with DWH oil hydrocarbons, which represents a degradation of food quality for organisms that prey on red crabs. Hence, while the magnitude of impact within this outer fourth zone is difficult to quantify, due to the uneven deposition of oil and floc throughout the area, the Trustees determined that injury had also occurred within this fourth zone.
- Significant losses to resident corals and fish were documented within approximately 4 square miles of mesophotic reef habitat along the continental shelf edge. It can be reasonably inferred that ecological functions provided by this biologically rich and important habitat were impaired. Exposure and spill impacts may also have occurred in a larger area, approximately 3,600 square miles in size, that extends beyond and between the areas where the Trustees quantified injury.
- Injury to tall soft corals on the mesophotic reefs reduced the amount of habitat important to fish and other smaller invertebrates.
- Harmful effects on coral individuals, colonies, and communities can degrade overall ecosystem health and function. Effects caused by the DWH incident that may have broader ecosystem ramifications include degradation of coral colony size and surface area, reduction in colony numbers, shifts in species dominance, and reductions in the diversity of benthic infauna. While the ecosystem-level impacts of the individually described injuries have not been directly

documented, their occurrence can reasonably be scientifically inferred from the nature and extent of the confirmed injuries.

Key finding: A wide variety of **nearshore and shoreline resources** were injured over hundreds of miles of coastline of the northern Gulf of Mexico. (Section 4.6)

The Trustees found injuries to multiple shoreline habitats over hundreds of miles of oiled shoreline of the northern Gulf of Mexico, including to estuarine coastal wetland habitats such as mainland salt marsh, sand beaches, submerged aquatic vegetation, and oyster reef habitats, as well as to plants and animals that live in these habitats. Water column injuries to fish and invertebrates also occurred in nearshore waters.

Specific injuries documented by the Trustees included reduced plant cover and vegetative (aboveground) biomass, and harmful reductions in the abundance, survival, growth, reproduction, and fitness of a number of important marsh animals. The Trustees also documented substantial reductions in nearshore oyster abundance and percent cover of nearshore oyster habitat, and an increase in marsh edge erosion rates. In some areas, oil is still present and injuries are ongoing.

- Injury to the estuarine coastal wetlands shoreline was observed over hundreds of miles, with more severe and broader injuries documented along more heavily oiled shorelines. Coastal marsh and mangroves are habitats critical to the overall health of the northern Gulf of Mexico. They provide invaluable spawning, nursery, and feeding grounds for the many commercial and recreational fish and shellfish species that depend on the physical protection offered by these habitats to complete their life cycles. They also help to protect water quality by capturing suspended sediment and removing excess nutrients and pollutants brought in from upland environments. The marsh edge, where the most acute injuries occurred, serves as a highly productive and critical transition zone between the emergent marsh vegetation and open water for the movement of organisms and nutrients between intertidal and subtidal estuarine environments.

- Animals that use the marsh edge for refuge and forage were exposed to oil through contact with oiled plants, soil, bottom and suspended sediments, detritus on the marsh surface, and water. Exposure occurred as the marshes flooded with the tide, as well as through ingestion or contact with oil entrained in submerged sediments near the marsh edge. Toxicity testing conducted with marsh sediment containing DWH oil demonstrates that PAH concentrations in soil and sediment found in oiled marsh areas are toxic to many marsh species. Cleanup and oil removal activities at the edge of marshes smothered, crushed, or removed animals in oiled areas.

- Substantial decreases in secondary production occurred along heavily oiled marshes for representative marsh species, including marsh periwinkles, brown and white shrimp, Gulf killifish, and southern flounder.

- Injuries to both subtidal and nearshore oysters were documented, causing a loss of ecological services that these organisms provide. Oysters play a unique role in the coastal ecosystem. They serve not only as an exploitable resource, but also as habitat for other aquatic organisms such as shrimp, crabs, and finfish. They provide filtration services that improve water quality and clarity. Oyster reefs adjacent to marshes reduce marsh erosion; when these reefs were injured, erosion increased.

- Salinity control structures were opened as part of response actions intended to reduce the movement of oil into sensitive marsh and shoreline areas. Unlike the sediment diversions the state of Louisiana uses for coastal restoration, the structures opened in response to the DWH incident have been historically used for releasing river water into surrounding embayments to maintain estuarine conditions. As such, they are normally opened during specific times of the year, for limited durations, and with controlled flow rates to effect targeted impacts to salinity levels. In contrast, when used for response to the DWH incident, these structures were opened at or near maximum capacity for extended time periods to repel the approaching oil. The highly atypical flow of river water over a sustained period greatly reduced salinity levels in Louisiana coastal areas during spill response. These resulting salinity reductions caused collateral injuries to estuarine organisms such as oysters and brown shrimp.

- The Trustees concluded that reduced salinity dramatically reduced the abundance of subtidal oysters in coastal Louisiana, whereas nearshore oysters were primarily injured by exposure to oil and the impacts of response activities (potentially in combination with reduced salinity in some locations). Annual NRDA sampling of both oyster settlement and abundance has shown that the initial injuries severely impaired oyster reproduction in the years following the spill, limiting their recovery. With reduced numbers of juvenile and adult oysters in subtidal areas in 2010, fewer larvae were produced in 2011 and subsequent years. Reduced oyster cover in nearshore areas contributed to recruitment decline and limited recovery throughout the region. Diminished recruitment has continued at least into 2014 and is compromising the long-term sustainability of oyster reefs in some areas.

- Beaches and dunes are ecologically and recreationally important shoreline habitats that serve as important breeding, nesting, wintering, and foraging habitats for nearshore and dune-dwelling biota. Injuries from response activities on beaches and dunes included:
    - Direct mortality and persistent behavior modification of nocturnal animals, such as beach mice and ghost crab, and destruction of dune vegetation.
    - Reductions in abundance, species richness, and diversity of small beach-dwelling organisms, such as crabs, snails, and shrimp, which occurred due to physical crushing, desiccation, and smothering and to the removal of wrack (decomposing vegetation washed up on shore by surf) that is an important habitat and food source for many beach organisms.

- An estimated 1,100 to 3,600 federally listed threatened Gulf sturgeon were potentially exposed to DWH oil in nearshore areas, representing a large proportion (an estimated 27 to 100 percent) of the populations of six of the eight natal rivers systems (Pearl, Pascagoula, Escambia, Blackwater, Yellow, Choctawhatchee). The Trustees found evidence of genotoxicity and immunosuppression at the molecular, cellular, and organ levels in sturgeon resulting from oil exposure, although the degree and consequences of exposure could not be quantified.

- Submerged aquatic vegetation (SAV) provides highly productive coastal habitat, including food, and shelter for birds, fish, shellfish, invertebrates, and other aquatic species. SAV was injured across the northern Gulf of Mexico due to oiling and the physical effects of vessels responding to the DWH incident. The Trustees' assessment documented 9,429 square feet of vessel scars and blowholes in Florida seagrass beds, of which 5,404 square feet were within the boundaries of Gulf Islands National Seashore, Florida District.

- SAV coverage totaling 60 acres along the Lake Cataouatche shoreline in Jean Lafitte National Historical Park and Preserve was reduced by approximately 83 percent as a result of river water releases during oil spill response actions (Section 4.6.8.2.3).

- Oil from the DWH spill injured SAV in the Chandeleur Islands, Louisiana. From 2010 to 2012, seagrass spatial distribution decreased along the shallow shelf west of the Chandeleur Islands. A total of 112 acres of seagrass beds were identified as "persistent loss" (defined as loss for two consecutive mapping intervals), and 160 acres were classified as "delayed loss" (areas where seagrass was present in 2010 and 2011, but lost in 2012).

Key finding: Exposure to oil and response actions injured a large number of **bird species** occupying different habitats, from offshore to nearshore, and including coastal marshes. (Section 4.7)

The Trustees confirmed that many tens of thousands of birds were killed by oil exposure, including offshore sea birds, shorebirds, waterfowl, marsh birds, and colonial nesting birds. Many other birds were injured through sublethal effects of oil exposure, loss of habitat, or displacement by response actions, as well as through reproduction foregone due to loss of breeding adult birds. As described below, these injury estimates do not include mortality estimates for all birds that likely were exposed and died due to the DWH incident.

- For those bird species and habitats for which mortality was quantified, 51,600 to 84,500 birds died as a result of the spill. Mortality likely is closer to the upper end of this range, due to factors that could not be quantified, such as birds within the interior marsh that were not captured by the models, or mortality in colonies that was not recorded. Species with high mortality estimates included brown pelicans, laughing gulls, terns, skimmers, and northern gannets.

- Adult birds that died between May 2010 and April 2011 as a result of the oil spill were not available to produce or sustain young during the breeding season following their death, leading to lost reproduction amounting to 4,600 to 18,000 fledglings during that breeding season.

- The Trustees did not quantify mortality to other types of bird species, particularly marsh birds and colonial nesting birds, because of practical difficulties in field observation and sampling.

- Coastal marsh and small barrier island restoration, through the ecosystem approach, will benefit the bird species whose injuries could not be quantified.

Key finding: Four species of federally endangered or threatened **sea turtles** (Kemp's ridleys, green turtles, loggerheads, hawksbills) were injured by exposure to oil and as a consequence of response activities. The Trustees estimated that exposure to oil resulted in up to 160,000 deaths to small juvenile turtles living offshore. Thousands of larger sub-adult and adult turtles were killed in shallower waters closer to the shoreline. Tens of thousands of sea turtle hatchlings were displaced by response actions and lost from the Gulf of Mexico ecosystem. The number of sea turtles nesting along northern Gulf beaches declined after the spill. (Section 4.8)

- Sea turtles and their habitats were exposed to DWH oil in the open ocean, across the continental shelf, and into nearshore and coastal areas, including beaches. Sea turtles were exposed to oil when swimming through oil at or near the surface and in the water column; breathing oil droplets, oil vapors, and smoke; and ingesting oil-contaminated water and prey. Response activities and shoreline oiling also directly injured sea turtles, and disrupted or deterred sea turtle nesting in the Gulf.

- The Trustees' quantification of sea turtle injuries caused by the DWH incident showed that sea turtles from all life stages were lost from the northern Gulf ecosystem. In particular, hundreds of thousands of small juvenile turtles died from oil exposure and response activities. Overall, the Trustees estimated that response activities injured nearly 35,000 hatchling sea turtles (loggerheads, Kemp's ridleys, and green turtles), and that the DWH incident killed 55,000 and up to 160,000 small juvenile sea turtles (Kemp's ridleys, green turtles, loggerheads, hawksbills, and hardshelled sea turtles not identified to species) and 4,900 and up to 7,600 large juvenile and adult sea turtles (Kemp's ridleys, loggerheads, and hardshelled sea turtles not identified to species). In addition, the Trustees estimated tens of thousands of Kemp's ridley and loggerhead hatchlings as potential reproduction foregone since 2010 due to the loss of breeding-age turtles killed on the continental shelf by the DWH incident.

- As noted above, DWH oil caused significant losses of *Sargassum* habitat on which small, oceanic-stage juvenile turtles rely, further compounding impacts on sea turtles and their ability to recover.

Key finding: Coastal and oceanic **marine mammals**—dolphins and whales—were injured by exposure to oil from the DWH spill. Injuries included elevated mortality rates, reduced reproduction, and disease. Without active restoration, these populations will require decades to recover from these injuries. (Section 4.9)

- Tens of thousands of federally protected marine mammals were exposed to the DWH surface slick, where they inhaled, aspirated, ingested, and came into contact with oil components. The oil's physical and toxic effects damaged tissues and organs, leading to a constellation of adverse health effects, including reproductive failure, adrenal disease, lung disease, and poor body condition in bottlenose dolphins.

- Animals that succumbed to these adverse health effects contributed to the largest and longest marine mammal unusual mortality event (UME) on record in the northern Gulf of Mexico. The dead, stranded dolphins in this UME included near-term fetuses from failed pregnancies. More than 75 percent of pregnancies in Barataria Bay and Mississippi Sound were unsuccessful.

- Barataria Bay and Mississippi Sound bottlenose dolphins were some of the most severely injured populations, with a 51 percent and 62 percent maximum reduction in their population sizes, respectively. Northern and western coastal populations of bottlenose dolphins and all of the shelf and oceanic marine mammal populations that overlap with the DWH oil spill footprint also suffered injuries. Dolphins are long-lived animals and slow to reach reproductive maturity. Without active restoration, these populations will require decades to recover from the injuries caused by the DWH incident.

- Smaller percentages of the marine mammals that live in deeper oceanic waters were exposed to DWH oil. However, they still experienced increased mortality (as high as 17 percent), increased reproductive failure (as high as 22 percent), and a higher likelihood (as high as 18 percent) of other adverse health effects.

Key finding: The oil spill reduced human uses of shoreline and coastal resources, resulting in **lost use** valued at hundreds of millions of dollars. (Section 4.10)

The Gulf of Mexico is a popular destination for many types of recreation, including beach-going, boating, and fishing. The spill directly reduced recreational use of these coastal resources across the northern Gulf of Mexico.

- The U.S. public lost almost 17 million user days of boating, fishing, and beach-going experiences between May 2010 and November 2011 (Section 4.10.6). This number does not include losses to private or commercial enterprises or municipalities, which are not compensable under the NRDA regulations in the Oil Pollution Act (OPA).

- The Trustees estimated the public value of these lost uses to be $693 million (due to uncertainty, the actual value may range from $528 million to $859 million).

4.11.3 Use of Inference to Assess Natural Resource Injuries Not Directly Measured by Trustees

The injuries to natural resources and services documented by the assessment are likely not the only injuries that occurred:

- The vast scale of the DWH incident precluded studying all individual components of all affected ecosystems in all affected locations over the full time period of potential effects. For this reason, the Trustees' designed their injury assessment to evaluate *representative* locations, habitats, species, and injury types.

- As with any ecosystem, Gulf natural resources are inter-linked through fundamental ecological relationships (e.g., habitat-community-species interactions, predator-prey relationships, nutrient transfer and cycling, and organism migration and behavior). Therefore, resources not directly exposed to oil or other consequences of the incident could be exposed indirectly through their ecological links to exposed resources.

In their injury assessment, and as reflected in the key findings described in Sections 4.11.2 and 4.11.4, the Trustees considered not only directly observed or measured injuries, but also injuries that could not be directly studied. To assess injury to natural resources not studied, the Trustees used scientific inference to extend their conclusions beyond the resources and locations they did observe or sample directly. Scientific inference involves using data, observations, and knowledge to make reasonable conclusions about things that were not directly observed. For example, observations and data supporting a conclusion that sufficient amounts of oil can smother wetland plants may be used to infer that similar plants that are similarly oiled would also be smothered, even when this effect was not directly observed. Similarly, existing knowledge can support reasonable scientific inferences. For example, if certain species of organisms are known to depend on marsh plants, scientists can reasonably infer that eliminating those marsh plants would harm the dependent organisms.

This section describes four types of ecosystem inferences considered by the Trustees in developing conclusions from the injury assessment. Not all these inferences apply to every injury category.

4.11.3.1 Inference Based on Foodweb Relationships

Impacts to a specific resource can indirectly affect both predators and prey:

- “Bottom-up” trophic impacts can occur when an important food resource species is impacted. For example, brown shrimp were injured because the incident adversely affected their invertebrate prey. Marsh periwinkles, terrestrial insects, amphipods, and Gulf killifish all are important prey for larger fish and birds, thus injuries to these food sources could also injure their predators. Similarly, larval fish and invertebrates injured in the water column are an important source of prey for larger fish. Birds are highly responsive to variation in their prey. Prey reductions, when they occur, can have cascading effects on both larger species and older life stages. Animals in the wild live in a dynamic relationship with their environment and available resources, balancing energy expenditures and nutritional uptake in order to survive, remain healthy, and reproduce. Any impact that shifts that balance by diminishing food resources or requiring unusual expenditures of energy—whether to acquire prey, avoid predators, fight disease and infection, or successfully reproduce—is inherently harmful to the species. Such harm is "an adverse change in a natural resource or impairment of a natural resource service," constituting an injury as defined in OPA regulations (15 CFR 990.30).

- Alternatively, impacts to a species higher on the food chain (such as dolphins) can reduce predation pressure on their prey, resulting in potential changes to the prey’s community structure, as well as changes to dynamic relationships within a species and among multiple species. For example, injuries to tall soft corals not only reduce the structural complexity of mesophotic reef habitats that attracts other animals, but also affect populations of invertebrates that graze on polyps. Birds are also known to exert top-down effects on the number and distribution of prey species.

4.11.3.2 Inference Based on Cascading Ecological Effects

The northern Gulf of Mexico ecosystem is a network of diverse habitats and species and functions linked through important ecological processes. Consequently, injury to natural resources can cause cascading ecological effects, including changes in trophic structure (such as altering predator-prey dynamics as mentioned above), community structure (such as altering an area's species composition), and ecological functions (such as altering nutrient flow and organic production), as illustrated by the following examples.

- **Water column cascading impacts.** Ecological processes in the water column affect the flows of organic matter and nutrients that, in turn, influence ecological processes on the deep-sea floor or in shallow nearshore habitats. Many animals living on the sea floor or in nearshore habitats spend early parts of their lives drifting in the water column.
- **Shoreline and nearshore cascading impacts.** Injuries to shoreline and nearshore ecosystems also have cascading effects on offshore ecosystems, including changes in the sequestration of sediments and nutrients in coastal wetlands; reduced capacity for intertidal oysters to serve as a source of oyster larvae for regional subtidal reefs; and reduced capacity for supporting juveniles of offshore species that use these habitats as nurseries.
- **Impacts to ecosystems beyond the Gulf.** Many bird species move across and beyond the Gulf's ecosystems. In addition to their role in food webs, birds transfer nutrients between marine and terrestrial biomes, disperse seeds, and provide other ecological functions. The ramifications of bird injury resulting from the DWH incident are not necessarily limited to the ecosystems of the northern Gulf, as many birds migrate to other areas of North America, where impaired performance or reduction in numbers can have radiating effects on the ecosystems there. Reduced populations of sea turtles, too, can have significant effects on the ecosystems to which they migrate.

4.11.3.3 Inference Based on Reasonable Analogy from More- to Less-Studied Ecosystems

Ecosystem scientists routinely make inferences by analogy from more- to less-studied ecosystems. For example, as described in Section 4.5 (Benthic Resources), injuries to hardground corals caused by the DWH incident manifested over time in the form of broken coral branches and reduced colony size and health. The ecosystem functions of these unique hardground corals are less well understood in the deep ocean; however, scientists do know that the substantial vertical structure of other fan-like coral species in shallower habitats provides cover and protection to mobile marine life seeking places to live and breed and refuge from predators. It is reasonable to infer that fan-like corals in the deep sea would provide similar ecological services.

4.11.3.4 Inference to Unstudied Resources and Locations Based on Representative Studies

Environmental scientists routinely use representative sampling to make statistically based inferences from sampled locations to locations that could not be sampled. For example, the Trustees used information collected at representative coastal wetland vegetation sampling sites to estimate the extent of injury to other wetland habitats in a statistically unbiased manner.

4.11.4 The Scope of Adverse Effects from the *Deepwater Horizon* Incident Constitutes an Ecosystem-Level Injury

The injuries caused by the DWH incident affected such a broad array of linked resources and services over such a large area that they cannot be adequately described at the level of a single species, a single habitat type, a single set of services, or even a single region. Rather, the effects of the DWH incident constitute an ecosystem-level injury. As described below, an ecosystem-level injury can reasonably be scientifically inferred from the demonstrated injuries across all trophic levels and across all northern Gulf of Mexico habitats, and from impacts to ecological communities and ecosystem functions.

Key finding: Injuries occurred at **all trophic levels.**

Based on the NRDA injury studies, and additional non-NRDA studies published in the literature, the Trustees determined that the DWH incident injured virtually every trophic level in the northern Gulf ecosystem, from bacteria, to primary producers (plants), to secondary producers such as zooplankton, to top-level predators such as bottlenose dolphins. Within coastal marshes, for example, a wide array of organisms—from microbes to large animals and including primary producers, animal consumers, and top predators—were injured, in addition to the marsh plants themselves. Injured organisms included the very small meiofauna and microalgae inhabiting marsh soil; larger invertebrates such as amphipods, periwinkle snails, and fiddler crabs; and Gulf killifish. Important resource species, such as shrimp, flounder, and bottlenose dolphins, that live in adjacent waters and feed on affected invertebrates and small fish were also injured.

Key finding: Injuries occurred to **virtually all marine and estuarine habitats** that came in contact with oil, from the deep sea to the shoreline.

The DWH incident affected resources throughout virtually every known marine and estuarine habitat in its trajectory in the northern Gulf of Mexico, from the deep sea to the shoreline, although the injuries were not uniform in severity or location. Oil in the deep sea injured both **soft-bottom habitats** and rare deep-sea corals. Toward the shoreline, important mesophotic reef habitats and associated communities were injured, as were benthic fish in shallow reef communities. Injuries were also observed in fish that live along the continental shelf habitat, such as red snapper.

In other benthic areas, however, injuries did not occur. For example, the Trustees conducted dozens of sampling trips to shallow water coral reefs in the Florida Keys and Flower Gardens reef areas and documented no evidence of exposure to DWH oil, dispersants, or disruptive response activities (DWH Trustees 2012).

Water column resources were injured in the open ocean, in coastal waters, and in nearshore waters and estuaries over approximately 43,300 square miles where oil was present. The injuries to water column resources occurred at the ocean surface, to organisms living in the mixed layer beneath the ocean surface, and to the extensive floating *Sargassum* habitats that support a wide variety of organisms, such as invertebrates, fish, and juvenile sea turtles. Water column resources also were injured in deep, colder waters as a result of exposure to both the rising cone of oil from the wellhead and the deep water plume of oil that formed at a depth of over 3,600 to 4,600 feet below the ocean's surface.

The northern Gulf of Mexico supports a wide array of **nearshore and shoreline habitats**, including estuarine coastal wetlands, such as marsh and mangrove habitat, submerged aquatic vegetation, subtidal and intertidal oyster reefs, barrier islands, and sand beaches. Injuries occurred to each of these habitats. In addition, these nearshore and shoreline habitats and biological resources are linked to both **coastal and offshore habitats** and resources through ecological and physical relationships, such as foodweb dynamics, organism movements, nutrient and sediment transport and cycling, and other fundamental ecosystem processes. As a result of these interactions, injuries to nearshore resources can have cascading impacts throughout the ecosystem, and injuries to nearshore and shoreline resources influence the overall health and productivity of the Gulf ecosystem.

Because the DWH incident injured diverse habitats, injury was not confined to a single set of species or ecosystem functions. Rather, the incident likely impacted important linkages across all Gulf habitats and resources.

Key finding: Injuries occurred to **species, communities, and ecosystem functions.**

In addition to the direct injuries to specific species observed by the Trustees (for example, injury to bottlenose dolphins or brown shrimp), the Trustees determined that adverse effects had occurred to ecological communities and ecosystem functions.

The multiple species and trophic levels injured within salt marsh communities, discussed above, are one example of **community-level injury**. In another example, benthic communities in areas where deposited oil had contaminated deep-sea sediments experienced decreased species diversity, indicating effects on multiple invertebrate species. Similarly, multiple components of the mesophotic reef community were injured, from tall soft corals to bottom-dwelling fish.

Ecosystem functions were also injured. For example:

- Marsh plants contribute several important ecosystem functions and services. They produce biomass through photosynthesis and form the basis of wetland and estuarine food webs. They help stabilize shorelines by holding, retaining, and accumulating marsh sediments. They also contribute to coastal flood protection by reducing storm surge and waves, and they provide critical structural habitat (as refuge and forage) for a wide variety of organisms. Injuries to marsh vegetation resulted in losses of these important ecosystem functions and associated services.

- The Trustees documented accelerated erosion rates along heavily oiled marsh shorelines in Louisiana where injuries to vegetation and intertidal oysters were observed. This increased erosion exacerbates Louisiana's already critical coastal erosion problem.

- Other examples of ecosystem function injuries include impaired cycles of organic matter and nutrients from the water column to oil-contaminated bottom sediments; altered transfer of energy and nutrients from coastal to offshore ecosystems where estuarine-dependent fish and shrimp were injured; and water filtration and nutrient cycling where oysters were injured.

4.11.5 Treatment of Unquantified Injuries

The Trustees could not fully quantify all direct and indirect injuries resulting from the DWH incident due to the vast geographical and ecological scope of impacts. However, they did document evidence of a number of injuries that could not be explicitly quantified. As detailed in Chapter 5, the Trustees' damage assessment and restoration plan accounts for unquantified injuries in several ways:

- The plan addresses injuries that could be determined, but not actually quantified, by directing restoration at ecosystem components that are similar in location or type or connected to ecosystem components with quantified injuries. Also, many species will benefit from habitat-level restoration (e.g., restoration of shoreline marsh or improvements to water quality), regardless of whether their injury status is known. In this manner, restoration projects can benefit resources known to have been injured, as well as analogous or related resources for which injury information was not available or could not be quantified.
- In some cases where the Trustees have documented that injury has or likely has occurred, the potential long-term effects or consequences are less well quantified because of environmental complexities in ocean systems, and because future environmental conditions that may influence organisms (such as temperature and precipitation) are unknown. The Trustees have decided that waiting for a better understanding of recovery from injury is not the best way to compensate for these potential ongoing injuries. Rather, they have proposed to start work now to achieve offsetting environmental benefits. By emphasizing both resource- and habitat-based restoration, the Trustees can provide a wide range of beneficial ecosystem services, which will reduce the likelihood of further injuries.
- For many unquantified injuries, additional time and more study is not likely to substantially change the Trustees' understanding of the nature or extent of injuries. The inherent difficulties in studying many oceanic systems limit the degree to which some conclusions can be reached with numerical precision. Additional study is unlikely to result in information that will substantially alter the Trustees' current conclusions. Further, confounding factors can arise over time, making injury quantification even more difficult as time passes. Therefore, the Trustees have decided that the best way to address unquantified losses is to initiate restoration now, rather than delay in the hope that further study will enhance quantification.

Despite these uncertainties that are inherent in any NRDA, the information gathered and analyzed is sufficient to allow the Trustees to form reasonable scientific conclusions about the nature and scope of the injuries.

4.11.6 References

DWH Trustees (Deepwater Horizon Natural Resource Damage Assessment Trustees). (2012). *Natural resource damage assessment. April 2012. Status update for the Deepwater Horizon oil spill*. Retrieved from *http://www.doi.gov/deepwaterhorizon/upload/FINAL_NRDA_StatusUpdate_April2012-2.pdf*

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