OCS Study
BOEM 2021-048

Oil Spill Effects Literature Study of Spills of 500–20,000 Barrels of Crude Oil, Condensate, or Diesel







US Department of the Interior

Bureau of Ocean Energy Management

Anchorage, Alaska

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Jacqueline Michel, Editor

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DISCLAIMER

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REPORT AVAILABILITY

To download a PDF file of this report, go to the U.S. Department of the Interior, Bureau of Ocean Energy Management [Data and Information Systems webpage](http://www.boem.gov/Environmental-Studies-EnvData/) (http://www.boem.gov/Environmental-Studies-EnvData/), click on the link for the Environmental Studies Program Information System (ESPIS), and search on 2021-048. The report is also available at the National Technical Reports Library at https://ntrl.ntis.gov/NTRL/.

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ABOUT THE COVER

Photographs of the types of resources discussed in the report. All photographs are from public sources.

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List of Abbreviations and Acronyms

AWOIS	Automated Wrecks and Obstruction Information System
bbl	barrel
BEA	Bureau of Economic Analysis
BIOS	Baffin Island Oil Spill
BOEM	Bureau of Ocean Energy Management
CFR	Code of Federal Regulations
cm	centimeter
CO	carbon monoxide
CRFS	California Recreational Fisheries Survey
CYP1A	cytochrome p4501A
DINAA	Digital Index of North American Archaeology
DOI	Department of the Interior
ESA	Endangered Species Act
ESI	Environmental Sensitivity Index
g	gram
GDP	Gross Domestic Product
GOM	Gulf of Mexico
GSI	gonadosomatic index
ha	hectare
IEc	Industrial Economics, Inc.
kg	kilogram
km	kilometer
L	liter
m	meter
mg/kg	milligram per kilogram
mg/L	milligram per liter
mm	millimeter
MMPA	Marine Mammals Protection Act
MRIP	Marine Recreational Information Program
NADB	National Archaeological Database
NAGPRA	Native American Graves Protection and Repatriation Act
N:C	nematode:copepod
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NHPA	National Historic Preservation Act
NOAA	National Oceanic and Atmospheric Administration
NOEP	National Ocean Economics Program

NRDA	Natural Resource Damage Assessment
NRHP	National Register of Historic Places
OCS	Outer Continental Shelf
OECM	Offshore Environmental Cost Model
OMA	oil-mineral aggregate
OR&R	Office of Response and Restoration
P:A	polychaete:amphipod
PCDD/PCDF	polychlorinated dibenzodioxins and polychlorinated dibenzofurans
PAH	polycyclic aromatic hydrocarbon
PM	particulate matter
ppm	parts per million
RPI	Research Planning, Inc.
SHPO	State Historic Preservation Office
SAV	submerged aquatic vegetation
STTWG	sea turtle technical working group
SOA	sediment-oil agglomerate
tPAH	total polycyclic aromatic hydrocarbon
TPH	total petroleum hydrocarbons
TROPICS	Tropics Oil Pollution Investigations in Coastal Systems
TWRC	Tafira Wildlife Rehabilitation Center
µg/g	micrograms per gram
µg/L	micrograms per liter
µm	micron
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
UVF	ultraviolet fluorescence spectroscopy
VOCs	volatile organic compounds
WTP	willingness to pay

1 Introduction

1.1 Study Objectives

The Bureau of Ocean Energy Management (BOEM) uses reference information regarding the potential effects of sized oil spills on the physical, biological, social, and/or economic resources on the Outer Continental Shelf (OCS) to support environmental analyses under the National Environmental Policy Act (NEPA). The objective of this study is to synthesize documentation regarding impacts to physical, biological, social, and economic resources from spills of crude oil, condensate, or diesel (and diesel-like oils such as No. 2 fuel oil and home heating oil) ranging in size from 500 to 20,000 barrels (bbl) in volume. This focus on these three oils reflects the types of oil likely to be spilled during exploration, development, and production of oil and gas resources on the OCS. Crude oil and condensate are produced on the OCS; diesel is used as a fuel for support vessels and power generation on offshore platforms. The range of spill volumes (500 to 20,000 bbl) was chosen to include the median OCS spill sizes that BOEM uses for NEPA analyses. Therefore, these spills will be defined as "median-range spills" for the discussions in each chapter.

A study of spills >20,000 bbl of crude oil, condensate, or diesel was also completed for BOEM by the same study team (Michel 2021). Together, both reports cover a range of spills that could affect physical, biological, social, and economic resources that may be considered during BOEM's NEPA analyses.

1.2 Study Methods

This literature review followed a systematic and reproducible method for identifying, evaluating, and synthesizing the available body of work on oil spill effects from spills of 500 to 20,000 bbl of crude oil, condensate, or diesel. The focus was on studies of actual spills and not of laboratory studies or modeling exercises that estimated impacts. Field experiments were included if deemed to be representative of actual spill conditions or if they linked oil exposure to specific effects. Also, this review did include a limited number of studies that are not focused on individual spills but were still relevant to assessment of spill impacts, such as the value of foregone beach user days for a specific region. Impacts from response activities were included; however, the individual- and population-level biological impacts of dispersants were not specifically addressed because a review of such impacts was recently completed by the National Academies of Sciences, Engineering, and Medicine (NASEM 2020) report on "The Use of Dispersants in Marine Oil Spill Response."

The spills included in this review documented impacts to resources that BOEM addresses in their NEPA documents. The findings of this review are not necessarily representative of all spills because documentation of impacts is not typically available for spills without observed impacts to identified resources. However, because BOEM's objective in commissioning this review is to understand potential impacts in the event that a spill occurs, this review's focus on spills with documented impacts is appropriate. This review includes the best available information as of 2020 to support BOEM's NEPA process.

Spill location was also a key consideration in the selection of spills to be included in this review because spills that reached water in coastal and offshore areas were considered to best represent OCS-related spill scenarios for BOEM's NEPA analyses. Thus, spills that were completely contained on land or in wetlands were not included, unless they provided information that enhanced the understanding of impacts of oil and/or response activities. A good example is the use of in-situ burning in coastal wetlands, which almost always occurs due to pipeline spills in the wetland interior (Michel and Zengel 2021). International spills where dispersants were applied to the intertidal zone or in shallow nearshore waters were not included because this response method has not been approved or used in the U.S.

As noted above, this review largely focused on documented impacts of specific crude oil, condensate, or diesel spills. We stayed within this scope with two exceptions. First, we included some median-range spills of heavy fuel oils that provided the only spill-documented effects on a resource. For example, studies were available on impacts to cultural resources only for the heavy fuel spills from the M/V *Kuroshima* and the M/V *Selendang* Ayu, both in the Aleutian Islands, Alaska. Also, impacts to recreation were included after the heavy fuel oil spills from the T/B *Bouchard-120* in Buzzards Bay, Massachusetts and the M/V *Cosco Busan* in San Francisco Bay, California. Second, a limited number of studies were included that were not focused on individual spills but are still relevant to assessment of spill impacts such as laboratory or field experiments (e.g., Tropics Oil Pollution Investigations in Coastal Systems [TROPICS] in Panama and Baffin Island Oil Spill [BIOS] in the Canadian Arctic) that used realistic exposures to document how such exposure affects the resources.

The literature review and data collection task was completed by a thorough search of databases with online search capabilities such as Web of Science and Google Scholar. Existing reviews and their reference lists were a key focus because such reviews had already identified those studies that were most useful and summarized the impacts.

The following sources for spill-related data were searched:

- National Oceanic and Atmospheric Administration (NOAA) Office of Response and Restoration (OR&R) Response Link website, which has response-related data on all spills where NOAA provided support since the late 1970s (https://responselink.orr.noaa.gov/);
- NOAA Damage Assessment, Remediation, and Restoration Program website, which includes all spills for which NOAA was involved in the Natural Resource Damage Assessment (NRDA) (https://darrp.noaa.gov);
- Louisiana Oil Spill Management System generated by the Louisiana Oil Spill Coordinator's Office, which includes oil spill response records and NRDA information for the State of Louisiana (https://data.losco.org);
- ITOPF website on oil spill case studies (https://www.itopf.org/in-action/case-studies/)
- California Office of Spill Prevention and Response NRDA cases (https://wildlife.ca.gov/OSPR/NRDA);
- Alaska Department of Environmental Conservation spill response summaries (https://dec.alaska.gov/spar/ppr/spill-information/response/);
- U.S. Department of the Interior NDRA case document library; and (https://www.cerc.usgs.gov/orda_docs/CaseSearch.

These searches resulted in a list of spill names that were included in ongoing searches.

This traditional approach to a literature search was expanded because some of the oil spill effects studies are in the "gray" literature, meaning they have not been published in peer-reviewed journals or in conference proceedings. Because Research Planning, Inc. (RPI) has been under contract to the NOAA OR&R for oil and chemical emergency response for over 40 years and is part of the Industrial Economics, Inc. (IEc) team that has supported government NRDA cases for 30 years, our team has an extensive in-house library on oil spill studies dating back to the early 1970s. In addition, we were able to reach out to our network of researchers and organizations to obtain studies that were unpublished. This effort resulted in acquisition of reports that have not been previously available.

1.3 Spills Researched and Resources Impacted

As a result of the literature search, sixty-two spills that met the criteria for spill volumes and oil types and had published information on the impacts to the resources applicable to BOEM were identified and are listed in **Table 1-1**. The location, oil type, and volume ranges for these spills were mapped in **Figure 1-1**. The literature was used to identify the resources for which impacts from these sixty-two spills were documented, shown with an "X" in **Table 1-1**. If a study was conducted for a resource and impacts were not detected, that is indicated by "N" in the table. Appendix A includes information on name, location, date, oil type, and spill volume (bbl) for each of the spills in **Table 1-1**. In the final analysis, all sixty-two spills had sufficient information on impacts to at least one resource category to be included in the synthesis.

The resource categories included in this analysis and the authors and reviewers of each chapter are:

- Physical resources
    - Air quality: Jason Price and Jacob Ebersole (IEc)
    - Water quality: Gail Fricano and Michaela Murray (IEc)
- Pelagic communities: Gail Fricano and Niamh Micklewhite (IEc)
- Marine benthic communities: Lauren Szathmary and Scott Zengel (RPI)
- Coastal and estuarine habitats
    - Salt marshes and mangroves: Pam Latham and Scott Zengel (RPI)
    - Beaches and tidal flats: Jacqueline Michel and Scott Zengel (RPI)
    - Rocky shores: Gail Fricano and Sophie Swetz (IEc)
    - Kelp: Christine Boring and Jacqueline Michel (RPI)
    - Submerged aquatic vegetation: Hal Fravel and Scott Zengel (RPI)
- Fish and motile invertebrates: Jennifer Weaver and Tracy Collier (RPI)
- Marine and coastal birds: Christine Boring and Jacqueline Michel (RPI)
- Sea turtles: Hal Fravel and Tracy Collier (RPI)
- Marine mammals: Hal Fravel and Tracy Collier (RPI)
- Terrestrial habitats and wildlife: Hal Fravel and Scott Zengel (RPI)
- Commercial and recreational fisheries: Jason Price, Jacob Ebersole, and Ahana Raina (IEc)
- Employment and income: Jason Price and Jacob Lehr (IEc)
- Cultural resources: Adam Stack and Jason Price (IEc)
- Marine archaeological resources: Adam Stack and Jason Price (IEc)
- Vulnerable coastal communities: Jason Price and Jacob Lehr (IEc)
- Recreation and tourism: Jason Price, Jacob Ebersole, and Ahana Raina (IEc)

Figure 1-2 shows the number of spills with studies on impacts to these twenty resource categories included in this synthesis. Note that cultural resources and marine archaeological resources are combined in **Table 1-1** and **Figure 1-2** because they shared similar spill data. Impacts to birds were most often documented (twenty-nine spills), but often just counts of bird mortalities. Marine benthic communities were assessed for twenty-two spills. Impacts to fish and invertebrates were documented for twenty spills. There were data on impacts for nineteen spills each for marshes and mangroves and beaches and flats. **Figure 1-2** clearly shows the paucity of studies of oiling impacts to human-use and socioeconomic resources (other than recreation and tourism). In fact, two median-range spills of a heavy fuel oil were included because of this lack of spill-related studies on impacts to these resources. Other resource categories with very limited data on impacts include air quality, submerged aquatic vegetation, sea turtles, and terrestrial habitat and wildlife. For marine mammals, often the only information available was reports of oiled animals rather than quantification of the impacts in terms of population changes.

Table 1-1. Spills of 500–20,000 bbl of crude oil, condensate, or diesel-like oils that were included in the literature search and the resources that were reported to have been impacted

The impacts to resources in **bold** were carried forward in the synthesis; other studies lacked sufficient data. “X” = resource category was affected; “N” = study was conducted and no substantial impact found. Note that cultural and marine archaeological resource categories are combined.

Spill	Air Quality	Water Quality	Pelagic Communities	Marine Benthic Communities	Marshes/ Mangroves	Beaches/ Flats	Rocky Shores	Kelp	Submerged Aquatic Vegetation	Fish/Invertebrates	Birds	Sea Turtles	Marine Mammals	Terrestrial Habitats/ Wildlife	Commercial/ Recreational Fishing	Employment/ Income	Cultural/Marine Archaeological	Vulnerable Coastal Communities	Recreation/ Tourism
Agip Abruzzo	-	-	-	**X**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
American Trader	-	-	-	-	-	**X**	-	-	-	**X**	**X**	-	**X**	-	-	-	-	-	**X**
Anitra	-	-	-	-	-	-	-	-	-	-	**X**	-	-	-	-	-	-	-	-
Antonio Gramsci 1987	-	**X**	-	**X**	-	-	-	-	-	**X**	-	-	**X**	-	-	-	-	-	-
Apex Barges	-	-	-	**X**	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Apex Houston	-	-	-	-		-	-	-	-	-	**X**	-	-	-	-	-	-	-	-
ARCO Anchorage	-	**N**	**N**	**X**	-	**X**	-	**X**	-	**X**	**X**	-	-	-	-	-	-	-	-
Baffin Island Oil Spill	-	-	-	**X**	-	**X**	-	**X**	-	-	-	-	-	-	-	-	-	-	-
Bahia Paraiso	-	**X**	-	**X**	-	**X**	-	-	-	X	**X**	-	**X**	-	-	-	-	-	-
Barataria Bay	-	-	-	-	**X**		-	-	-	**X**	-	-	-	-	-	-	-	-	-
Bayou Perot	-	-	-	-	**X**	X	-	-	-	-	-	-	-	-	-	-	-	-	-
Bohai Sea	-	**X**	**X**	**X**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bouchard 65 1974	-	-	-	**X**	**X**	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bouchard 65 1977	-	-	-	**N**	-	-	-	-	-	**N**	-	-	-	-	-	-	-	-	-
*Bouchard-120***	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**X**
BP MP-80 Delta 20	-	-	-	-	-	-	-	-	-	-	**N**	-	-	-	-	-	-	-	-
Bunga Kelana	-	-	-	-		**X**	-	-	**X**	**X**	-	-	-	-	-	-	-	-	-
*Cosco Busan***	-	-	-	-	-	-	-	-	-	**X**	**X**	-	-	-	**X**	-	-	-	**X**
Eagle Creek	-	-	-	-	-	**X**	-	-	-	-		-	-	-	-	-	-	-	-
El Omar	-	-	-	-	**X**	**X**	**X**	-	-	-	**X**	-	-	-	-	-	-	-	-
El Segundo	-	-	-	-	-	-	-	-	-	-	**N**	-	-	-	-	-	-	-	-
Equinox blowout	-	-	-	**X**	**X**	-	-	-	-	**X**	**N**	-	-	-	-	-	-	-	-
Esso Bayway	-	-	-	-			-	-	-	**N**	-	-	-	-	-	-	-	-	-
Estrella Pampeana	-	**X**	**N**	**X**	**X**	**X**	-	-	-	**N**	-	-	-	-	-	-	-	-	-
Eurobulker	-	-	-	**X**			-	-	-	X	-	-	-	-	-	-	-	-	-
Exxon Bayway Refinery	-	-	-		**X**	-	-	-	-	-	**X**	-	-	**X**	-	-	-	-	-
Fidalgo Bay	-	-	-	-	**X**	-	-	-	-	-	**X**	-	-	-	-	-	-	-	-
Florida West Falmouth	-	-	-	**X**	**X**	-	-	-	-	**X**	**X**	-	-	-	-	-	-	-	-
Garbis	-	-	-	**X**	**X**	**X**	-	-	-	**X**	-	-	-	-	-	-	-	-	-

Spill	Air Quality	Water Quality	Pelagic Communities	Marine Benthic Communities	Marshes/ Mangroves	Beaches/ Flats	Rocky Shores	Kelp	Submerged Aquatic Vegetation	Fish/Invertebrates	Birds	Sea Turtles	Marine Mammals	Terrestrial Habitats/ Wildlife	Commercial/ Recreational Fishing	Employment/ Income	Cultural/Marine Archaeological	Vulnerable Coastal Communities	Recreation/ Tourism
Glacier Bay	-	-	-	-	-	-	-	-	-	-	-	-	-	-	X	-	-	-	-
Greenhill Blowout	-	-	-		X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Jessica	-	-	-	X	-	-	N	-	-	X	X	X	X	-	-	-	-	-	-
Julie N	-	-	-		X	-	-	-	-	X	X	-	-	-	-	-	-	-	-
*Kuroshima***	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	X	X	-
Lake Barre	-	-	-	-	X	-	-	-	-	-	X	-	-	-	-	-	-	-	-
Lamma Island	-	-	-	-	-	X	X	-	-	-	-	-	-	-	-	-	-	-	-
Laura d'Amato	-	-	-	-	-	X	-	-	-	-	N	-	-	-	-	-	-	-	-
Long Island Sound	-	X	-	-	-	X	-	-	-	X	N	-	-	-	-	-	-	-	-
McGrath Beach	-	-	-	-	-	X	-	-	-	-	X	-	-	X	-	-	-	-	-
Moreton Bay	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mosquito Bay	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nakika MP-69	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nella Dan	-	-	-	-	-	-	X	X	-	-	-	-	-	-	-	-	-	-	-
North Cape	-	X	X	X	X	X	-	-	-	X	X	-	-	-	X	-	-	-	-
Refugio Beach	-	X	-	-	-	X	X	X	X	X	X	-	X	-	-	-	-	-	X
Rose Atoll	-	-	-	X	-	-	-	-	-	X	N	N	-	-	-	-	-	-	-
RTC-380	-	-	-	-	-	X	-	-	-	X	N	-	-	-	-	-	-	-	-
Ryuyo Maru No. 2	-	-	-	-	-	-	-	-	-	X	-	-	X	-	-	-	-	-	-
Sao Paulo pipeline	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
*Selendang Ayu***	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	X	X	-
Selfir	-	-	X	X	-	X	-	-	-	X	-	-	-	-	-	-	-	-	-
Tebar V Refinery	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-
Tenyo Maru	-	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-
Terra Nova	-	-	-	-	-	-	-	-	-	-	X	-	-	-	-	-	-	-	-
Texaco March Pt Refinery	-	-	-	N	-	X	X	-	-	-	X	-	-	-	-	-	-	-	-
Torch Platform	-	-	-	-	-	X	X	-	-	-	X	-	X	-	-	-	-	-	-
TROPICS	-	-	-	X	X	-	-	-	X	-	N	-	-	-	-	-	-	-	-
Uniacke G-72	X	X	-	N	-	-	-	-	-	N	N	-	N	-	-	-	-	-	-
Universe Leader	-	-	-	-	-	-	X	-	-	-	-	-	-	X	-	-	-	-	-
UNOCAL	-	-	-	-	-	-	-	N	-	-	X	-	X	-	-	-	-	-	X
World Encouragement	-	-	-	-	X	-	-	-	-	-	-	-	-	-	-	-	-	-	-
World Prodigy	-	X	X	X	-	X	-	X	-	X	X	-	-	-	-	-	-	-	X

** Heavy fuel oil spills included in this synthesis because of the paucity of data on impacts to cultural resources, vulnerable coastal communities, or recreation.

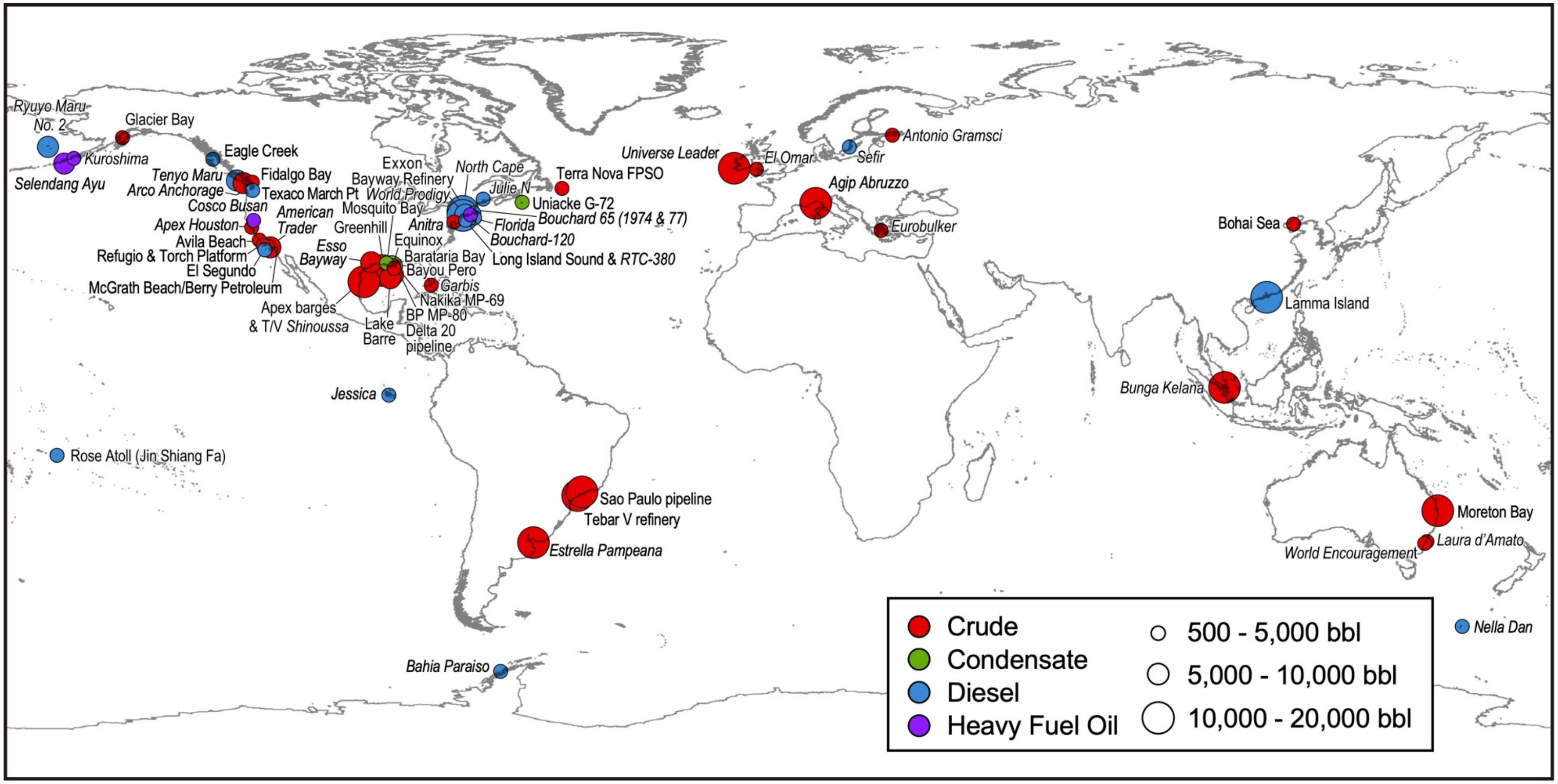

Figure 1-1. Location, oil type, and volume ranges for the sixty-two spills included in this synthesis
Four heavy fuel oil spills are included in this synthesis because of the paucity of data on impacts to cultural resources, vulnerable coastal communities, or recreation.

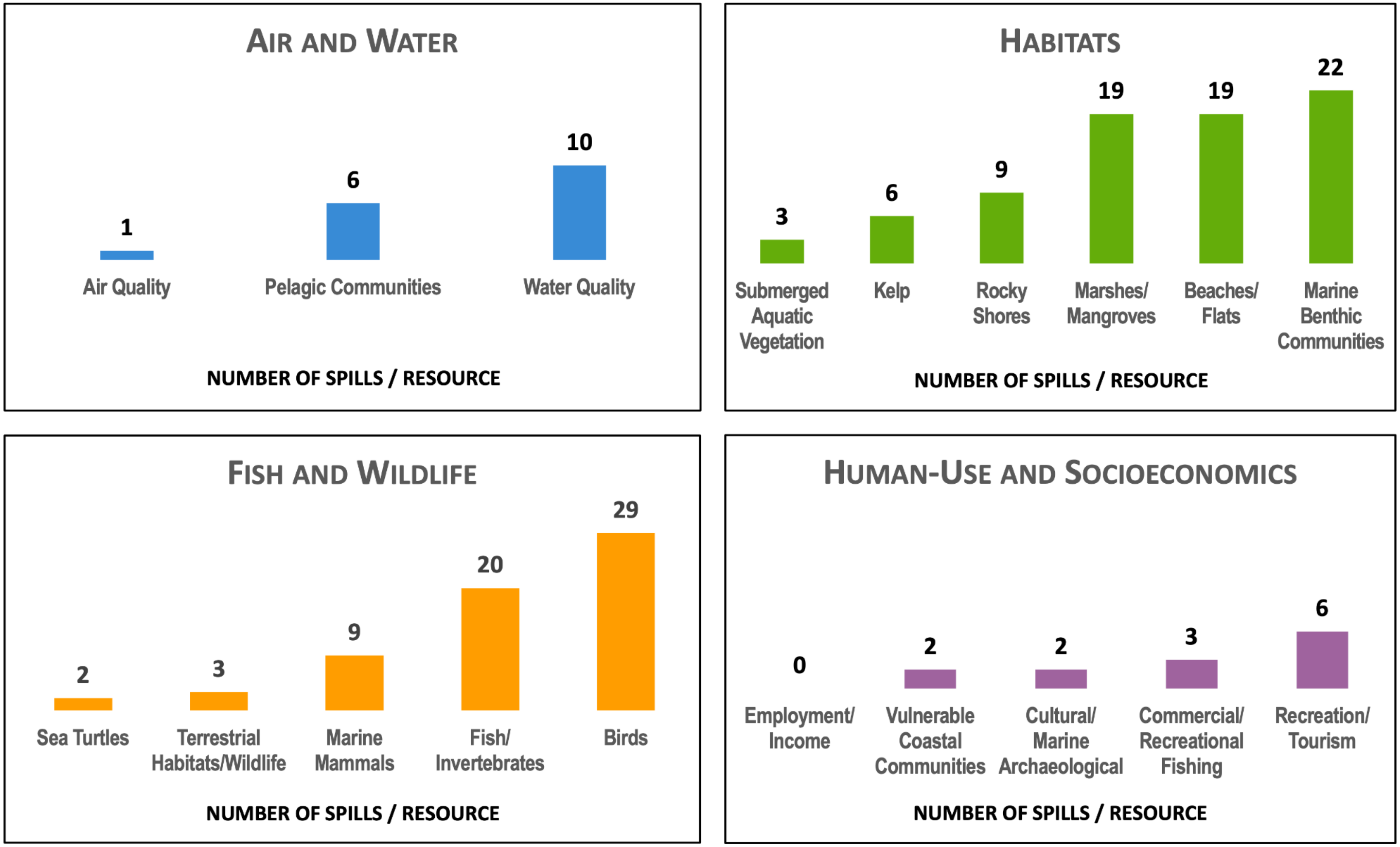

Figure 1-2. Number of spills with information on impacts to the twenty resource categories included in this synthesis
Note that cultural and marine archaeological resource categories are combined.

2 Oil Spill Effects by Resource Type

The degree and severity of impacts to resources from an oil spill depend on the spill location, size, oil type, timing, release depth, duration, meteorological and oceanographic conditions, various seasonal and environmental conditions, and the effectiveness of response activities. These factors can substantially affect weathering processes such as evaporation, emulsification, dispersion, dissolution, microbial degradation and oxidation, and transport of the spilled oil. Spills can have cascading effects on populations and ecosystems (BOEM 2016).

Direct exposure pathways include contact with skin, fur, feathers, and mucous membranes, ingesting contaminated food, grooming contaminated fur or feathers, and inhaling fumes, which can have short- and long-term health impacts (BOEM 2016; 2017). Direct exposure is primarily limited to oil spills that contaminate shorelines and to organisms frequenting shorelines or the sea surface where oil may spread. Spills may also alter habitats and contaminate or deplete available food resources in those environments.

Factors affecting the magnitude of impacts to resources from median-range spills include the following:

- **Spill location**. Spills that occur relatively close to shore are more likely to affect coastal resources and expose human populations to increased risk of adverse health effects and socioeconomic impacts than spills that occur farther from shore.
- **Spill volume**. All other factors being equal, larger spills will result in greater impacts than smaller spills.
- **Spill type**. Lighter crude oils, condensate, and diesel-like oils tend to be less persistent, whereas heavier crude oils are more persistent and thus can affect larger areas and have greater impacts. However, lighter oils can have greater acute toxicity and can therefore also cause substantial impacts when spilled nearshore or mixed into the water column during storms.
- **Oil Trajectory**. Depending on currents and wind, oil can be transported onshore, where impacts can be larger, or offshore, where impacts may be smaller. There are several spills included in this synthesis that did not have documented impacts because the oil never made landfall.
- **Atmospheric conditions at the time of the spill**. The direction and speed of winds at the time of a spill and its aftermath can influence the degree to which spill-related air pollution is transported from the spill site to other locations, including population centers on shore.
- **Season**. The season during which a spill occurs can affect its impact on air quality, because emissions released during the summer ozone season are likely to have a greater impact on ambient ozone concentrations than emissions released during other times of year. Season is also very important in determining the impacts to habitats and animals, because of seasonal migration patterns, life history (e.g., spawning and nesting periods), and times when animals tend to form concentrated aggregations–all factors that can increase the likelihood of impacts.
- **Type and effectiveness of spill response**. Specific spill response strategies have their own impacts. For example, emissions associated with in-situ burning of spilled oil differ from the emissions associated with response vessels deploying boom to contain spilled oil. Use of dispersants can increase exposure to water-column resources. In addition, the degree to which response measures are successful in recovering spilled oil may affect the magnitude of impacts.

In the following chapters, the impacts of spills 500–20,000 bbl of crude, condensate, or diesel are summarized for each resource category. Where appropriate, tables and graphs are used to describe the impacts and rates of recovery for each spill. Each chapter ends with a summary of impacts and identification of key information needs.

3 Air Quality

3.1 Resource Description

Air quality is defined according to the ambient concentration of individual pollutants in the air, such as tropospheric ozone (O_3) or fine particulate matter (PM) with a diameter equal to or less than 2.5 microns (µm) ($PM_{2.5}$), with lower concentrations indicating better air quality. Under the Clean Air Act, the U.S. Environmental Protection Agency (USEPA) sets National Ambient Air Quality Standards (NAAQS) for criteria to protect the public from adverse health and welfare impacts for the following pollutants:

- Nitrogen dioxide (NO_2)
- Carbon monoxide (CO)
- Sulfur dioxide (SO_2)
- O_3
- $PM_{2.5}$
- Particulate matter with a diameter equal to or less than 10 µm (PM_{10})
- Lead

For each of these pollutants, the Clean Air Act established two types of standards: a primary standard to protect public health and a secondary standard to protect public welfare, including protection against reduced visibility and harm to animals, crops, vegetation, and buildings.

In addition to establishing ambient air quality standards for criteria pollutants, the Clean Air Act requires USEPA to set emissions standards for 189 hazardous air pollutants (also referred to as HAPs or air toxics) based on the maximum achievable control technology for the affected sources. The Clean Air Act also requires USEPA to assess the public health risk remaining after the implementation of each of these emissions standards to ensure that the residual risk provides an ample margin of safety.

3.2 Impacts of Oil Spills on Air Quality

This chapter summarizes the effects of crude oil, condensate, or diesel spills of 500–20,000 bbl on air quality. Oil spills may result in air quality impacts due to emissions from spills themselves (e.g., emissions of volatile organic compounds [VOCs] from the evaporation of spilled oil), emissions from response vessels and other response equipment, and (when applicable) combustion-related emissions from the in-situ burning of spilled oil. In the case of a well blowout for wells containing both oil and natural gas, another potential source of emissions is combustion products from the flaring of recovered natural gas. The pollutants emitted from these sources may include, but are not limited to, VOCs, SO_2, oxides of nitrogen (NO_x), methane, $PM_{2.5}$, and PM_{10}. The release of air pollutants from these emissions sources may affect ambient concentrations of air pollutants both at the spill site and at downwind or other locations. Depending on the location of the spill and atmospheric conditions such as the speed and direction of prevailing winds, spill-related changes in ambient pollutant concentrations may occur both offshore as well as over land. Exposure to increased concentrations of air pollutants from oil can also lead to increased incidence of several adverse health effects.

Factors affecting the magnitude of a spill's air quality impacts include the following:

- **Spill location**. Spills that occur relatively close to shore are more likely to affect air quality in the onshore environment and expose human populations to increased risk of adverse health effects than spills that occur farther from shore.

- **Spill volume**. All other factors being equal, larger spills will result in greater air pollutant releases than lower-volume spills.
- **Atmospheric conditions at the time of the spill**. The direction and speed of winds at the time of a spill and its aftermath can influence the degree to which spill-related air pollution is transported from the spill site to other locations, including population centers onshore.
- **Season**. The season during which a spill occurs can affect its impact on air quality; for example, emissions released during the summer ozone season are likely to have a greater impact on ambient ozone concentrations than emissions released during other times of year.
- **Type and effectiveness of spill response**. Individual spill response strategies have their own emissions profile. For example, emissions associated with the in-situ burning of spilled oil differ from the emissions associated with response vessels deploying boom to contain spilled oil. In addition, the degree to which response measures are successful in recovering or destroying spilled oil may affect the magnitude of spill-related evaporative emissions.

The existing literature on the air quality impacts of median-range oil spills is limited to a single analysis of ambient methane readings following the blowout of the Uniacke G-72 gas well that released condensate 280 kilometers (km) off the coast of Halifax, Nova Scotia in 1984. Gill et al. (1985) indicated that the maximum methane-equivalent reading, obtained within several hundred meters (m) of the oil rig, was 1,155 parts per million (ppm). Values of 200 to 600 ppm were observed elsewhere over the oil slick. In addition, a strong hydrocarbon odor was present near the rig and, on one occasion, as far away as 10 km.

Other than the Uniacke G-72 incident, the literature included only studies of air quality impacts of larger spill events, such as the *Deepwater Horizon* spill, which are summarized in Michel (2021). Although the overall emissions from median-range spills and the associated changes in ambient pollutant concentrations are likely to be less substantial than for larger spills, the emissions factors for smaller spills (e.g., kilograms [kg] of emissions per kg of fuel burned during in-situ burns) may be similar to those for larger spills. In addition, the chemical composition of the emissions associated with median-range spills is, all else equal, likely to be similar to that for larger spills, as documented in Michel (2021).

3.3 Summary and Information Needs for Assessing Impacts to Air Quality

Oil spills may result in adverse air quality impacts from emissions of air pollutants directly from the spilled oil itself, emissions from vessels involved in response, emissions from in-situ burning, and emissions from the flaring of natural gas in the well (for spills involving well blowouts). Emissions from these sources may increase ambient concentrations of air pollutants at and downwind of the spill site, including coastal and onshore locations. Exposure to increased air pollutant concentrations can adversely affect the health of local human populations.

Although the available literature provides important insights into the magnitude of these effects, published studies have focused almost exclusively on spills >20,000 bbl. Thus, very limited information is available on the overall emissions associated with median-range spills or the corresponding changes in ambient pollutant concentrations at spill sites or at downwind locations. To the extent that studies focused on air quality impacts of median-range spills are conducted in the future, it would be important for them to capture air quality effects immediately after a spill (before surface oil is reduced), when such impacts are likely to be most severe. In addition, literature that emerges in this area would ideally capture the air quality implications of spills occurring in different locations in U.S. waters, as spills occurring in different geographic areas may have vastly different consequences for air quality based on the physical geography and meteorology of a given area.

4 Water Quality

4.1 Contaminants of Concern

This chapter discusses the behavior and persistence of oil in the water column following spills of 500–20,000 bbl of crude oil, condensate, or diesel and the subsequent impacts on water quality. Water quality impacts are typically assessed through the measurements of hydrocarbons such as polycyclic aromatic hydrocarbons (PAHs), total petroleum hydrocarbons (TPH), and, to a lesser degree, VOCs. This chapter focuses on the two most prevalent hydrocarbon measurements, PAH and TPH.

Crude oil and other petroleum products contain hundreds of different chemicals, making it impractical for researchers assessing water quality to measure each chemical individually. Accordingly, TPH and PAH are commonly measured. TPH represents the sum of both aliphatic and aromatic hydrocarbon compounds and provides a gross estimate of overall oil contamination (ATSDR 1999). PAHs are aromatic hydrocarbon compounds such as naphthalenes, fluorenes, anthracenes, and phenanthrenes (NRC 1983). PAHs are often the most frequent measure of water quality following an oil spill because they are thought to be the most toxic and carcinogenic components of oil (DWH NRDA Trustees 2016; ITOPF 2014). PAHs vary in solubility and volatility, with low molecular weight (2–3 ringed) PAHs being more soluble and volatile than higher molecular weight (4–5 ringed) PAHs. Thus, higher molecular weight PAHs are generally more persistent in the marine environment following an oil spill (DWH NRDA Trustees 2016). PAH concentrations below 1 microgram per liter (µg/L) are typically considered to result in little impact to the marine waters, and the USEPA has designated 1 µg/L as the threshold for chronic effects to aquatic life (OSAT 2010). Thus, researchers assessing water quality impacts after an oil spill often use 1 µg/L total PAH as the impact threshold for oil contamination (Horn and French-McCay 2014; Reich et al. 2016).

Fresh (recently spilled) oil also contains VOCs, which are mostly monoaromatic hydrocarbons such as benzene, toluene, ethylbenzene, and xylenes (BTEX) with high vapor pressure that evaporate and enter the atmosphere under normal atmospheric conditions (Petroleum Equipment Institute 2014). These chemicals have the potential to be toxic to living resources. After an oil spill, VOCs are usually only a concern for a relatively short duration because they readily evaporate from oil floating on the water surface (NOAA 2012). VOCs are therefore infrequently measured in water samples following an oil spill (see Chapter 3 – Air Quality).

4.2 Oil Behavior and Persistence in the Water Column

Figure 4-1 illustrates pathways that spilled oil may take following a surface spill. The various physical, chemical, and biological processes that change the properties of spilled oil are collectively known as weathering (Lee et al. 2015; Tarr et al. 2016). Oil composition and behavior undergo progressive changes over time because weathering processes have different onset times and magnitudes, which vary with temperature and type of oil (**Figure 4-2**; Lee et al. 2015). **Table 4-1** presents characteristics of diesel, condensate, and various weight crude oils that affect oil behavior and toxicity.

Immediately after a surface spill occurs, oil begins to spread laterally across the water surface, creating an oil slick. The speed of spreading largely depends on oil composition and ambient environmental temperature; colder temperatures and heavy crude oils result in more viscous slicks that are more resistant to spreading compared with warmer temperatures and lighter oils and are sometimes centimeters thick (Barron et al. 2020; ITOPF 2014).

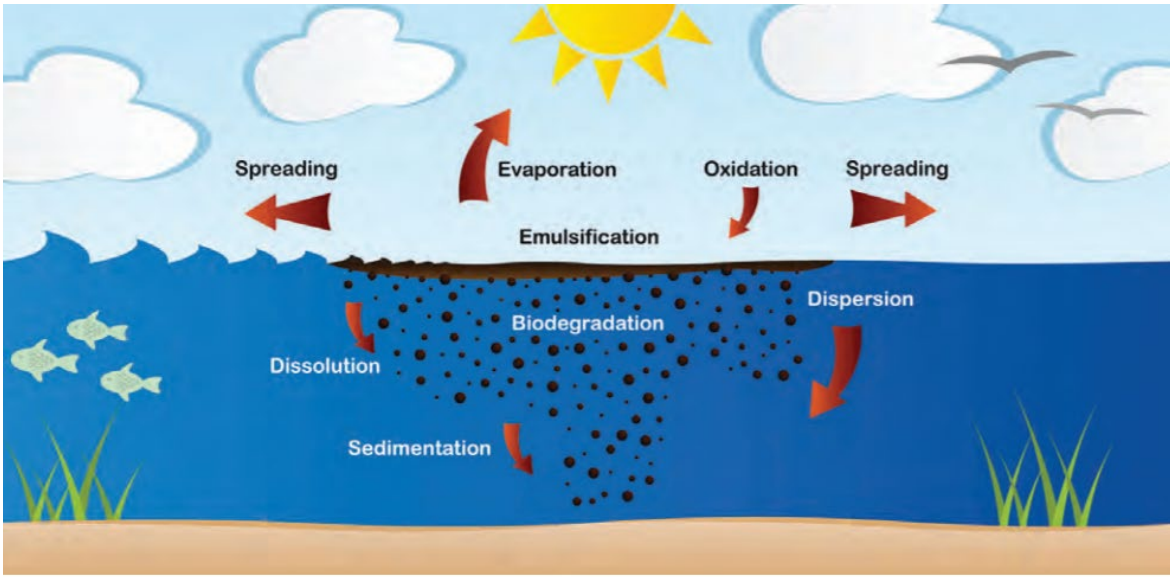

Figure 4-1. Oil pathways following a surface oil spill
Oil spreads laterally across the water surface and vertically throughout the water column due to dissolution and dispersion. Surface oil may evaporate, undergo photooxidation, and combine with water to form an emulsion. Biodegradation occurs in the water column, and some oil may settle to the seafloor via sedimentation. From ITOPF (2014).

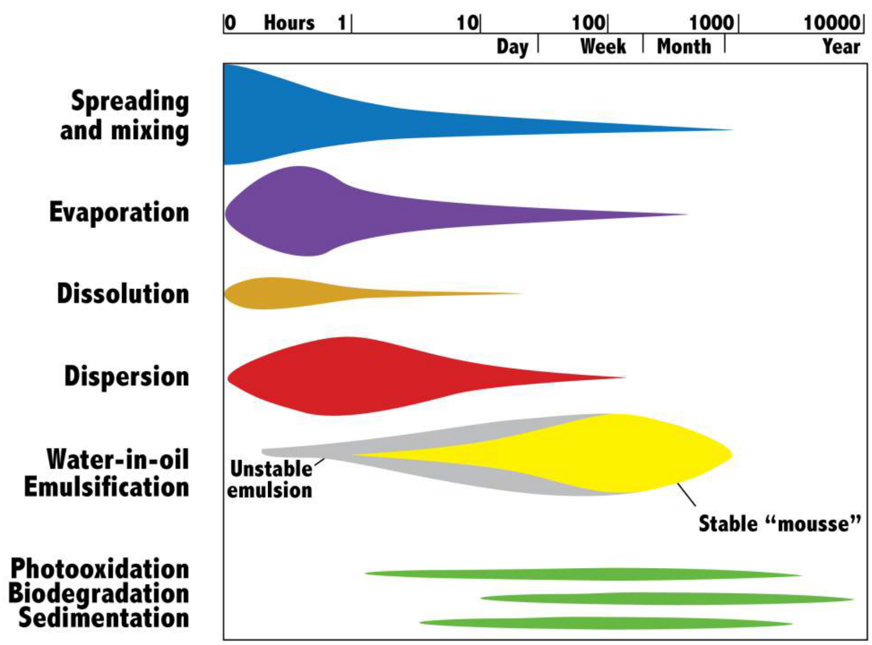

Figure 4-2. Time of onset and relative importance of weathering processes over time after an oil spill onto water
The onset and magnitude of effect varies with temperature and by type of oil. From Lee et al. (2015).

Table 4-1. Oil types included in this synthesis and their characteristics. Adapted from Michel and Rutherford (2013)

Diesel-like Products, Condensate, and Light Crude Oils • Specific gravity is 0.80–0.85; API gravity 35–45 • Moderately volatile and soluble • Refined products can evaporate to no residue • Crude oils and condensates can have residue after evaporation is complete • Low to moderate viscosity; spreads rapidly into thin slicks • Are more likely to affect animals in water and sediments because they are readily dispersed into the water column by winds and currents
Medium Crude Oils • Specific gravity of 0.85–0.95; API gravity 17.5–35 • Moderately volatile • For crude oils, up to one-third will evaporate in the first 24 hours • Moderate to high viscosity; will spread into thick slicks • Are more bioavailable than lighter oils (because they persist longer), so are more likely to affect animals in water and sediments
Heavy Crude Oils • Specific gravity of 0.95–1.00; API gravity of 10–17.5 • Very little product loss by evaporation or dissolution • Can be very viscous to semi-solid; may be heated during transport • Can form stable emulsions and become even more viscous • Tend to break into tar balls quickly • Low acute toxicity to water-column biota • Penetration into substrates will be limited at first, but can increase over time • Can cause long-term effects via smothering or coating, or as residues in the water column and sediments

In contrast, spills of lighter oils result in thin films that can spread more quickly (ITOPF 2014). Spreading is also dependent on other environmental factors such as winds, waves, currents, and ice, where it occurs.

The volatile components of the oil, which are usually the most acutely toxic, are quickly lost through evaporation. Surface spills of light crude oils, condensates, or diesel undergo greater rates of evaporation because those oils contain a higher proportion of hydrocarbons with lower molecular weights that are more volatile. Thus, spills of lighter crude oils, condensates, and light refined products generally have less persistent effects on water quality because most of the VOCs (up to 75% by weight) will evaporate (NRC 2003). In contrast, spills of heavier crude oil typically only lose about 5–10% of oil by weight to evaporation because they contain fewer hydrocarbons with lower molecular weights (Fingas 1995; NRC 2003). Due to

Key Terms

Weathering: Physical and chemical processes occurring in the environment that modify oil.

Evaporation: Loss of lighter-weight, volatile components of oil into the atmosphere.

Dissolution: Process by which compounds in oil that are at least slightly soluble dissolve when in contact with water.

Photooxidation: A chemical reaction between oil and oxygen in the presence of sunlight that results in new compounds with different properties that affect the oil fate and response.

Dispersion: Entrainment of oil droplets into the water column.

Emulsification: The incorporation of water droplets into oil which increases the viscosity and volume of the oil.

Biodegradation: Microorganisms consume hydrocarbons in oil, thus breaking it down into more elementary compounds.

their greater proportion of higher molecular weight compounds, heavy crude oils tend to persist in the environment for longer periods of time. Other factors that increase evaporative rates include rough seas, high wind speeds, and warm temperatures (ITOPF 2014).

In general, minor amounts of oil on or near the surface will undergo dissolution; however, because most water-soluble compounds are quickly lost through evaporation, dissolution is one of the least important weathering processes (ITOPF 2014). Oil on or near the surface is also subject to photooxidation in the presence of sunlight. The extensive and rapid changes to the physical and chemical properties of oil by sunlight may influence oil fate, transport, and the selection of response tools (Ward and Overton 2020).

Oil type and environmental conditions will determine the extent of natural dispersion. Typically, some oil will break up into droplets of varying size and disperse into the water column, usually becoming entrained in the top few meters. The smaller oil droplets remain suspended in the water column, while the larger droplets can rise back up to the surface to reform a slick or spread out into a thin film (ITOPF 2014). Rates of dispersion are usually more rapid for lighter oils than heavier oils but are conditional upon the amount of dissipative energy created by winds, tides, and currents (ITOPF 2014). Depending on environmental conditions and oil type, water may incorporate into the oil and form an emulsion (i.e., mousse). Emulsions increase the oil's viscosity and increase the volume of oil present in the water by up to a factor of five, which can delay other weathering processes such as evaporation (ITOPF 2014; NOAA 2016b). Generally speaking, emulsification is the main cause of the persistence of light and medium crude oil spills on the water surface (ITOPF 2014).

Microorganisms that are naturally present in seawater use the hydrocarbons in oil as energy and effectively degrade oil into water-soluble compounds (ITOPF 2014). This process of microbial degradation (also referred to as biodegradation) is influenced by oil composition and droplet size, nutrient availability, and water temperature. Though rates of microbial degradation are higher in warmer waters, the process occurs in cold Arctic waters as well (McFarlin et al. 2014; McFarlin et al. 2018). The use of dispersants (discussed below) has also been shown to increase rates of microbial degradation; because biodegradation occurs at the oil/water interface, the smaller oil particles created by the use of dispersants create more surface area for microbes to attach to, thereby increasing rates of biodegradation (Driskell and Payne 2018).

Though most crude oils, condensates, or diesel-like oils have low specific gravities and remain buoyant in water, some oil may sink to the seafloor as marine snow. Marine snow is composed of aggregated particles >5 millimeters (mm) that consist of a variety of smaller organic and inorganic particles, including bacteria, phytoplankton, microzooplankton, feces, feeding structures, detritus, and biominerals (Alldredge and Silver 1988). It typically forms in surface waters and is present throughout the world oceans. The sinking of marine snow is one of the primary processes through which surface-derived materials reach the deep sea and the seafloor (Alldredge and Silver 1988; Daly et al. 2016). Marine oil snow is formed when oil interacts with and is incorporated into the marine snow particles. Three formation mechanisms of marine oil snow have been proposed: 1) interaction of oil with mucus strands derived from microbial communities harbored in the oil, forming webs that eventually coagulate into marine oil snow; 2) collision of particles of oil components, bacteria, and natural suspended matter, directly forming aggregates known as "flocs"; and 3) coagulation of aggregates of phytoplankton (naturally forming in the absence of oil) that incorporates oil droplets when they are present (Passow et al. 2012). **Figure 4-3** shows a schematic of marine oil snow formation.

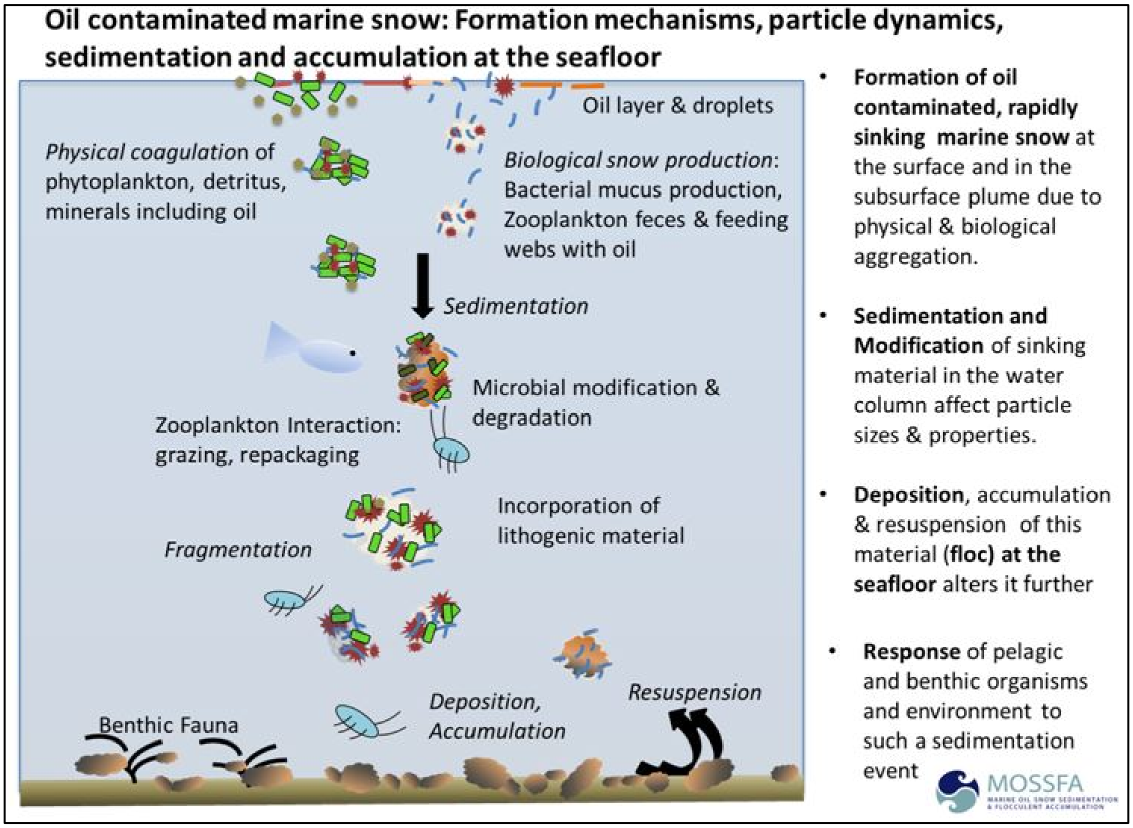

Figure 4-3. Schematic of the processes that form marine oil snow
From Deep-C Consortium (2013).

4.3 Dispersants

Dispersants are chemical mixtures of surfactants and solvents used to reduce the surface tension between oil and water, thus accelerating the formation of smaller oil droplets that more readily disperse into the water column (ITOPF 2014; NRC 2005; NRC 2020). The smaller oil droplets formed by dispersant action remain suspended in the water column and are subject to weathering processes such as natural dispersion, dissolution, and to a large extent, microbial degradation (ITOPF 2014; NRC 2005). **Figure 4-4** illustrates the process of microbial degradation following the use of dispersants at the surface.

Application of dispersants to surface oil can be from vessels or aircraft. Dispersants can be applied to subsurface oil using subsea injection (NASEM 2020). The effectiveness of dispersants is limited by environmental conditions. Some degree of wave energy is necessary to prevent smaller oil droplets from resurfacing and reforming slicks, but severe wave conditions will prevent direct contact between the dispersant and oil (ITOPF 2014; NRC 2005). Efficacy also depends on oil type; dispersant effectiveness decreases as oil viscosity increases (ITOPF 2014; NRC 2005).

Concerns over the widespread use of dispersants during the *Deepwater Horizon* oil spill led to additional research on the toxicity of oil, dispersed oil, and dispersants. Results have been ambiguous, due in part to inconsistent study methods. However, laboratory experiments have found that dispersant components degrade rapidly, within hours to days. Field studies have found that concentrations of dispersants generally decrease to less than 1 ppm within minutes to hours (NASEM 2020). Detailed information on dispersants, including their fate and transport, biological effects, and use in oil spill response, is provided in the NASEM 2020 report on "The Use of Dispersants in Marine Oil Spill Response."

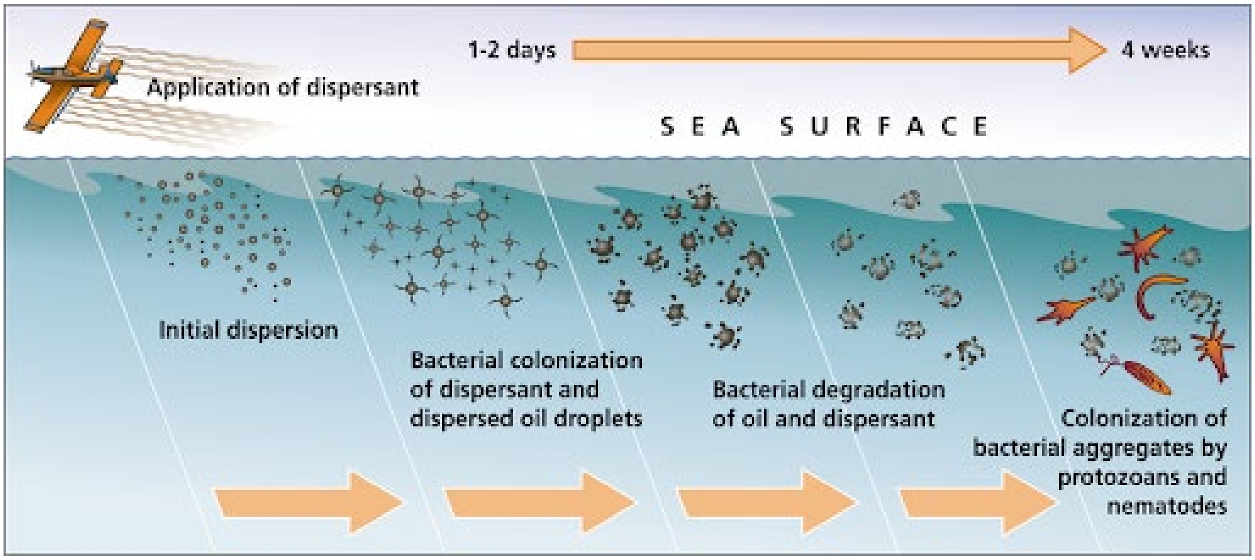

Figure 4-4. Aerial dispersant application and microbial degradation process
Initial dispersion breaks up oil into smaller droplets whereby bacteria can begin to colonize and degrade the oil and dispersants. Within 4 weeks of dispersant application, bacterial aggregates, protozoans, and nematodes have substantially degraded the dispersed oil. From Schmidt (2010).

4.4 Water Quality Assessment Methods

The trajectory and fate of spilled oil and subsequent impacts to water quality can be assessed using remote sensing, field sampling, and modeling. Each method has its own unique challenges and benefits. A basic familiarity with these methods is useful in understanding oil spill study results.

4.4.1 Remote Sensing

Oil spill response often includes remote sensing techniques to map the location and extent of the surface oil. Remote sensing can be active or passive, with the latter being most common in spill response (Fingas and Brown 2018). Passive remote sensing includes direct visualization of the water surface, photographs from drones, airplanes, or helicopters, and satellite imagery. Darkness, cloud cover, and sun glitter can affect passive remote sensing (**Figure 4-5**). Cameras with specific filters, polarizing lenses, or night-vision have been developed to reduce these impediments (Fingas and Brown 2018). Cameras or satellites that use infrared technology can characterize oil thickness. Thicker layers of oil appear "hot" compared to the surrounding water, intermediate thickness layers appear "cool," and thinner layers or sheens of oil are undistinguished (Fingas and Brown 2018). Though seaweed, turbidity, and oceanic fronts may interfere with infrared sensing, the technology generally produces viable results and is readily available during a response, making it a popular choice in remote sensing technology (Fingas and Brown 2018).

Methods of active remote sensing include laser fluorosensors and radar imagery. Laser fluorosensors use the distinct fluorescent intensities and spectral properties of oil to differentiate oil classes. Radar and satellite radar systems (e.g., Radarsat-2) can monitor large areas (**Figure 4-6**) and operate in dark, cloudy, or foggy conditions. However, these systems may be susceptible to false detection of oil slicks, and image resolution may be impacted by high wind speeds (Fingas and Brown 2018). There is also a delay in image acquisition, which can impede use by oil spill responders (Abt Associates 2018).

Figure 4-5. Photographs of an oil spill with and without confounding glare
Top depicts oil with glare. Bottom is at a different angle with a multispectral camera. From O. Garcia, WaterMapping.

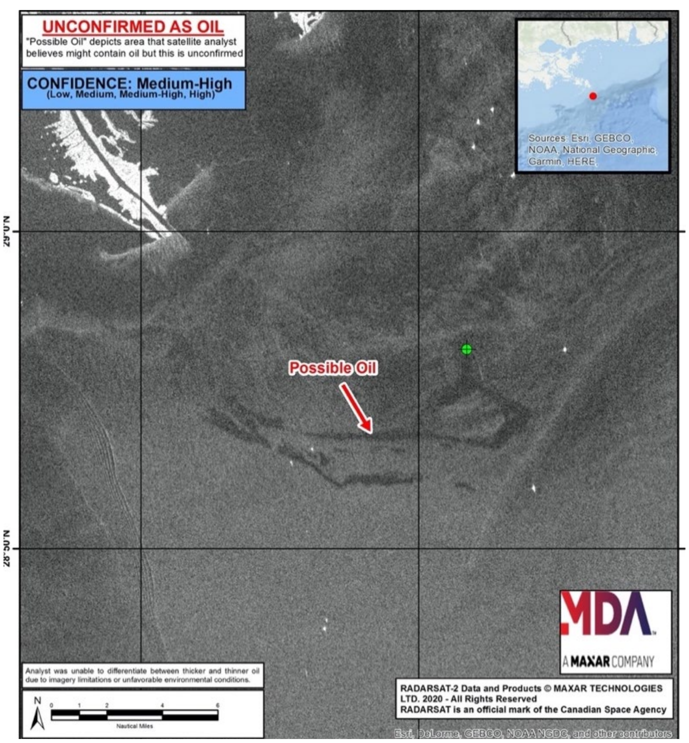

Figure 4-6. Radarsat-2 imagery of an oil spill off the Mississippi River delta on 5 March 2020
Analysis by the NOAA Satellite and Information Service.

4.4.2 Field Sampling

Oil slick thickness can be measured in-situ using several methods such as dip plates, sorbent pads, and sample tubes deployed with a high-definition camera. In general, plates, pads, jars, or tubes are deployed at the water surface to discern the volume of oil, which can be used in conjunction with surface area measurements to determine oil slick thickness (Abt Associates 2018).

To determine PAH and TPH concentrations in the water column, water samples can be collected using techniques such as Go-Flo or Niskin bottles and transported to a laboratory for analysis. Go-Flo bottles are lowered through the water column in a closed position and are opened at the desired depth to prevent contamination of the sample by oil films on the sea surface (ITOPF 2014). Niskin bottles are placed in the water in an open position. During the *Deepwater Horizon* spill, fresh water from hoses was used to create an oil-free opening that the rosette of Niskin bottles could be lowered into to avoid surface oil contamination. Collection of water samples can be challenging due to a variety of environmental conditions such as wave action and wind speeds, both of which influence how research vessels and oil slicks move across the water.

Water samples can be analyzed for contaminants using techniques such as solvent extraction, ultraviolet fluorescence spectroscopy (UVF) (**Figure 4-7**), gas chromatography-flame ionization detection (GC-FID), gas chromatography-mass spectrometry (GC-MS), or comprehensive two-dimensional gas chromatography (ITOPF 2014; Yim et al. 2012). Fluorescence measurements can be limited by the sensitivity, selectivity, and accuracy of fluorometric readings in detecting and quantifying oil. Sometimes colored dissolved organic matter can be mistaken for oil (Abt Associates 2018).

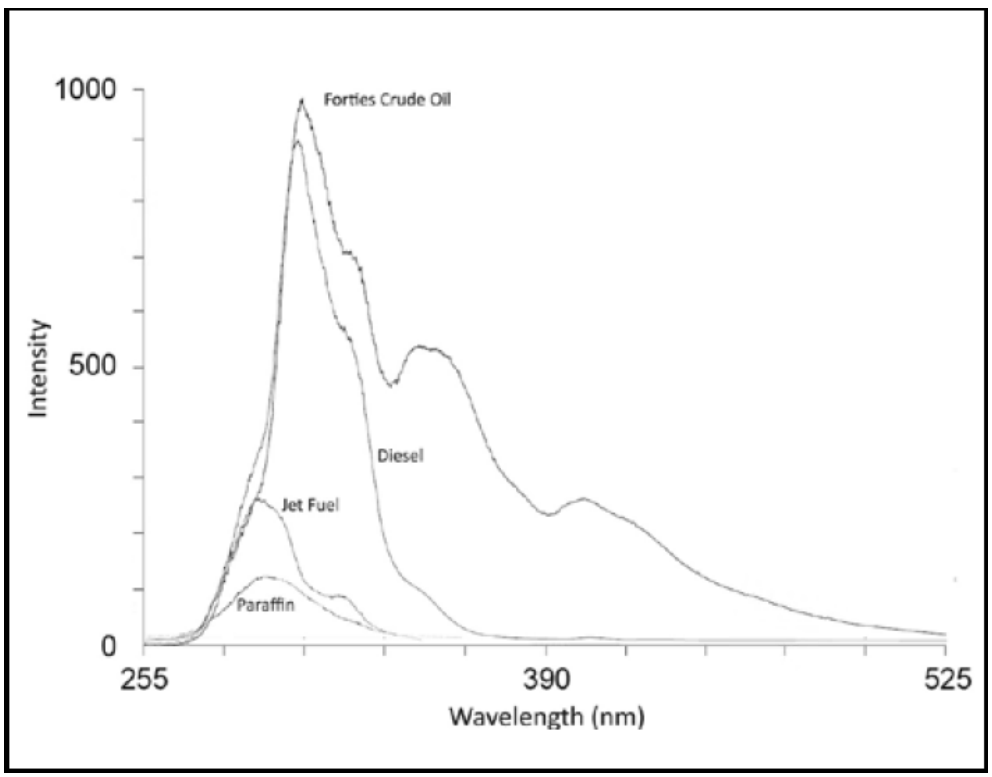

Figure 4-7. UVF emission spectra for four different types of oil
The four different types of oil (forties crude oil, diesel, jet fuel, and paraffin) have varying peak intensities depending on the wavelength. From ITOPF (2014).

4.4.3 Oil Spill Trajectory Modeling

Several models have been developed to predict the trajectory and fate of spilled oil in coastal and marine environments. These models, while varied in approach, consider parameters such as the type and quantity of oil spilled, the rate of release, and environmental data such as winds, currents, tides, and temperatures. They do not precisely predict the changes that oil undergoes in the environment, but they help researchers determine whether oil is likely to dissipate naturally from the water surface or whether it will reach shorelines or sensitive areas (ITOPF 2018). The accuracy and availability of data on the type and volume of oil spilled and the spatial detail for environmental parameters can limit the accuracy of oil spill modeling. Models generally become more accurate as the oil spill incident develops and site-specific data are collected.

Examples of oil spill models include the General NOAA Operational Modeling Environment (GNOME) Suite, which is a set of modeling tools use to predict the fate and transport of marine pollutants (Zelenke et al. 2012), and the Spill Impact Model Application Package (SIMAP), an integrated system that provides detailed predictions of oil trajectory, fate, and biological effects such as impacts to shellfish and wildlife (French-McCay 2004). French-McCay et al. (2018) developed models for oil fate and extent from subsea releases. The Chemical/Oil Spill Impact Module (COSIM) simulates three-dimensional oil fate and transport processes and aids researchers in examining concentrations of specific contaminants in water and transport to sediments on the seafloor (Kubitz et al. 2011).

4.5 Spill-specific Oil Behavior, Persistence, and Impacts

Though the water column is inevitably exposed after oil spills, few studies have documented the extent of TPH or PAH exposure over time and space. Based on a thorough review of the literature regarding water quality impacts from oil spills and response, eight oil spills of 500–20,000 bbl of crude oil, one spill of condensate, and one spill of diesel were identified that included water quality impact data (**Table 4-2**). Each of the ten spills is discussed in more detail in the following sections. In particular, this section highlights, to the extent possible, the degree of oiling (i.e., type and volume), behavior of the oil, range of TPH or PAH concentrations over time and space, and duration of water quality impacts. **Figure 4-8** summarizes the recovery time of water quality for each of the oil spills listed in **Table 4-2** and discussed in this section. The spills in **Table 4-2** are listed in increasing order of impacts and recovery.

Table 4-2. Studies with documented impacts to and recovery of water quality from spills of crude oil, condensate, or diesel-like oil (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Documented Range of Oil Impacts Over Time and Space	Recovery (months)
1985 T/V *ARCO Anchorage*, Port Angeles, WA[a]	5,690 bbl Alaska North Slope crude	Within 2 days post spill, water samples taken in the area showed no detectable petroleum hydrocarbons.	0
1989 T/V *World Prodigy*, Narraganset Bay, RI[b]	6,500 bbl home heating oil	Within 24 hours post spill, TPH concentrations ranged from 100–430 µg/L at 0.08 m below the surface and were as high as 500 µg/L at 5 m depths. By 2 days post spill, TPH concentrations ranging from 200–1,300 µg/L were observed at depths of 0.08 m. Hydrocarbons were not detected 6 days post spill.	<0.3
1984 Uniacke G-72 blowout, Sable Island, Nova Scotia[c]	1,500 bbl gas condensate	During and soon after the blowout, hydrocarbons were detected in the water column as far as 10 km from the wellhead were at depths of 0.5–21 m. Higher TPH concentrations were observed in	<1

Oil Spill	Oil Volume and Type	Documented Range of Oil Impacts Over Time and Space	Recovery (months)
		the vicinity of the wellhead at levels 15–50x higher than background concentrations. The maximum TPH concentration was 150 µg/L, though most samples contained values less than 100 µg/L. TPH levels returned to background levels 8–10 days after the spill was controlled approximately 3 weeks after the spill began.	
1972 Long Island Sound, CT[d]	1,905 bbl No. 2 fuel oil	By 6 days post spill, chromatograms of water samples at 2 of the 3 sampling stations presented no distinguishable evidence of oil contamination. Some residual contamination was observed at the mid-bay sampling station likely due to oil leaching from the sediment.	2
2015 Refugio Beach, Santa Barbara, CA[e]	500 bbl Monterey crude	In the first 2 weeks post spill, TPH and total PAH (tPAH) concentrations ranged from 40–697,000 µg/L and below detection to 73.2 µg/L, respectively, at sites adjacent to the release point. Overall, concentrations decreased with distance from the release site and were not observed after 2 months post spill.	2
1989 T/V *Bahia Paraiso*, Arthur Harbor, Antarctica[f]	3,760 bbl diesel fuel arctic	During the 2 months post spill, tPAH concentrations were 50–100 µg/L. tPAH concentrations were not detected 2–2.5 months post spill.	<2.5
1987 T/V *Antonio Gramsci*, Gulf of Finland[g]	4,200 bbl crude	In the first month post spill, average TPH values in surface waters ranged from 0.7–2.6 µg/L. Three months after the spill, average TPH concentrations peaked with a range of 0.6–17.0 µg/L, but quickly returned to background (range=0.5–1.5 µg/L) about a month later (4 months post spill).	4
1999 T/V *Estrella Pampeana*, Ria de la Plata coast, Argentina[h]	1,570 bbl Patagonia crude	Sites that were most impacted by the spill had TPH levels as high as 39 µg/L 8 months post spill. TPH concentrations at those sites did not return to baseline levels (1 µg/L) until 2.5 years after the spill.	30
2011 Bohai Sea, China[i]	3,200 bbl crude oil and drilling mud	Average TPH concentrations 5 years after the spill were 28.3 µg/L at the surface (0–0.3 m), 27.7 µg/L at mid-depths (14.0–14.3 m), and 29.5 µg/L at the bottom (27.5–28.0 m). Concentrations were highest in the vicinity of the platform where levels increased from the surface (36.1 µg/L) to the bottom (45.3 µg/L).	<60
1996 T/B *North Cape*, South Kingstown, RI[j]	19,700 bbl home heating oil	Samples collected within 12 km of the grounding site at depths of 2–25 m had TPH and tPAH levels as high as 3,940 and 115 µg/L, respectively, and average concentrations were 483 (TPH) and 21.9 (tPAH) µg/L.	Unknown

[a]Chamberlain et al. (1987); [b]Pilson (1990); [c]Carter et al. (1985); Gill et al. (1985); [d]EPA (1973); [e]Donohoe et al. (2019); [f]Kennicutt et al. (1991a); [g]Hirvi (1990); [h]Moreno et al. (2004); [i]Wang et al. (2020); [j]Reddy and Quinn (1999)

The duration of water quality impacts is defined as the length of time between the start of the oil spill and the last observance of TPH or PAH concentrations above background levels[1]. Given that various environments have differing background levels of contamination, this approach provides a characterization of impacts relative to the geographical setting in which the spill occurred. Assessing the duration of impacts is dictated by the temporal scope of sampling conducted by researchers. As sampling methods and frequencies vary by spill, comparisons of the duration of impacts across spills are challenging.

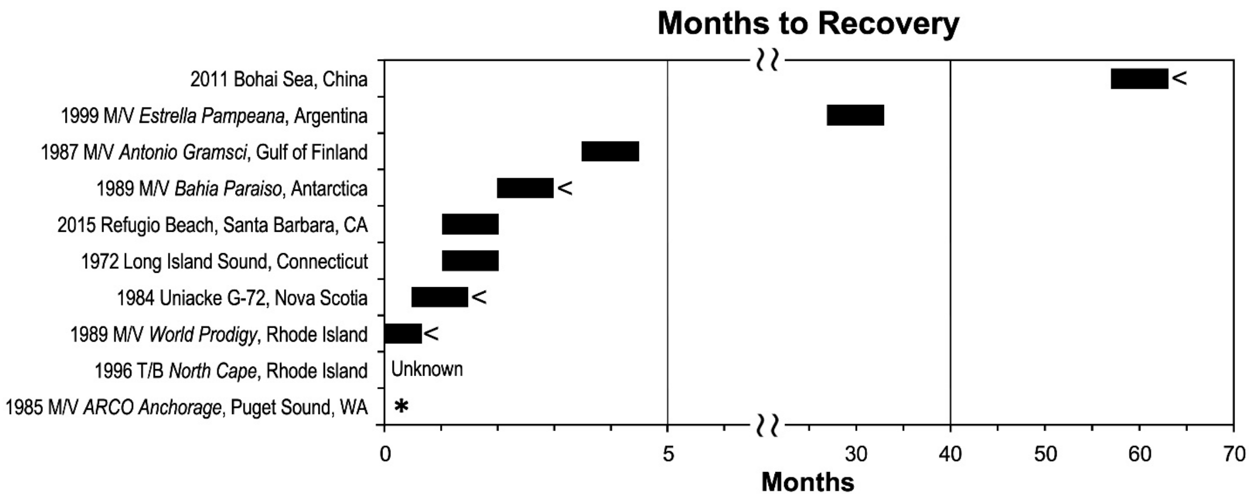

Figure 4-8. Recovery of water quality following the oil spills listed in Table 4-2
Asterisk indicates impacts were not detected. < symbol indicates that recovery occurred prior to the date of the most recent study.

The definition of water quality impacts used in this chapter does not address biological impacts. As noted previously, some researchers have used PAH concentrations exceeding 1 µg/L as a universal threshold for identifying oil impacts on biota (Horn and French-McCay 2014; Reich et al. 2016). However, this chapter does not use this or any other biological threshold to define water quality impacts, as biological effects cannot be easily predicted based on a single threshold. Further, many of the studies summarized here present concentrations of TPH rather than PAH.

4.6 Water Quality Case Studies

4.6.1 T/V *ARCO Anchorage*

The *ARCO Anchorage* tanker ran aground in Port Angeles Harbor, Washington, resulting in the release of 5,690 bbl of Alaska North Slope crude oil. Strong winds caused the oil to quickly spread across the water surface and reach nearby beaches. Response efforts were extensive, and within 5 hours of the spill, 4,500 feet of containment boom had been deployed (Chamberlain et al. 1987). Petroleum hydrocarbons were not detected in the water column, even at 1 m directly beneath a slick (Chamberlain et al. 1987). The rapid spread of the oil to shorelines and the extent of response efforts may have contributed to the indistinguishable impacts on water quality.

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[1] Background levels refer to preexisting contamination in the environment due to natural oil seeps, previous oil discharges, marine and land transportation, non-point source runoff, and industrial activities.

4.6.2 T/V *World Prodigy*

The *World Prodigy* tanker ran aground on Brenton Reef off the coast of Rhode Island resulting in a surface spill of 6,500 bbl of home heating oil in Narragansett Bay. Environmental conditions during the days following the spill limited the severity and duration of adverse impacts to water quality. Light winds transported oil slicks across the water surface, and warm weather increased rates of evaporation such that by 4 days after the spill, an estimated 99% of the surface oil had disappeared (Pilson 1990). Water quality samples taken during the 2 weeks following the spill demonstrated that hydrocarbon concentrations were greatest near the spill area. Within 24 hours of the spill, TPH levels ranged from 100–430 µg/L at 0.08 m below the surface and were as high as 500 µg/L at 5 m depth (Pilson 1990). Thus, there is evidence oil was distributed throughout the water column, though the temperature gradient in the water during the summer months following the spill likely diminished vertical mixing (Pilson 1990). Two days after the spill, TPH concentrations ranging from 200–1,300 µg/L were observed at depths of 0.08 m (Pilson 1990). Hydrocarbon concentrations were no longer detected in open-water samples within 6 days after the spill.

4.6.3 Uniacke G-72

There was a loss of well control at the Uniacke G-72 well, drilled to a depth of 5,142 m off the coast of Nova Scotia, resulting in a 10-day spill of 1,500 bbl of condensate at the water surface. A large portion of the oil (approximately 70%) evaporated due to the large percentage of light hydrocarbons. Winds and rough seas played a large role in moving the oil slick across the water surface, which may have contributed to greater levels of mixing throughout the water column (Carter et al. 1985; Gill et al. 1985). During and soon after the blowout, hydrocarbons (measured as condensate-equivalent hydrocarbons) were detected as far as 10 km from the wellhead at depths ranging from 0.5–21 m; however, the highest concentrations were observed in the vicinity of the wellhead at levels 15–50 times higher than background concentrations (Gill et al. 1985). The maximum observed hydrocarbon concentration was 150 µg/L (Carter et al. 1985; Gill et al. 1985), and most samples had concentrations <100 µg/L (Gill et al. 1985). Hydrocarbon levels returned to background levels 8–10 days after the spill was controlled (approximately 3 weeks after the spill began).

4.6.4 Long Island Sound

The *F.L. Hayes* tanker grounded on Bartlett Reef in Long Island Sound, resulting in a spill of 1,905 bbl of No. 2 fuel oil into the marine environment. Three days after the spill, strong winds and high seas accelerated natural dispersion of the oil (EPA 1973). By 6 days after the spill, there was no visible evidence of oil in any open water samples, and chromatograms of water samples at two of the three sampling stations presented no distinguishable evidence of oil contamination (EPA 1973). Thus, the environmental conditions effectively dissipated most of the oil contamination. Some residual contamination was observed at the mid-bay sampling station for approximately 2 months after the spill, likely due to oil leaching from the sediment and the currents transporting the oil to that particular area (EPA 1973).

4.6.5 Refugio Beach

An underground pipeline near Refugio State Beach in Santa Barbara County, California, released nearly 500 bbl of Monterey crude oil that overflowed into the nearshore environment of the Pacific Ocean. In the first 2 weeks following the spill, TPH concentrations ranged from 40–697,000 µg/L at sites adjacent to the release point where the oil entered the ocean and decreased with distance; approximately 15 km and greater from the release point, TPH concentrations ranged from 40–5,100 µg/L (Donohoe et al. 2019). Similarly, PAH concentrations were highest at sampling locations near the release point during the 2 weeks following the spill. tPAH concentrations ranged from below detection to 73.2 µg/L at a site adjacent to the release point, and concentrations were as low as below detection to 5.3 µg/L as a site approximately 20 km south (Donohoe et al. 2019). Water quality samples were not taken consistently until 5 days after the spill, so it is possible that maximum concentrations were higher than those recorded (Donohoe et al. 2019). Elevated levels of TPH and tPAH were observed for approximately 2 months following the spill (Donohoe et al. 2019).

4.6.6 T/V *Bahia Paraiso*

The *Bahia Paraiso* tanker ran aground in Arthur Harbor, Antarctica, resulting in a surface release of 3,760 bbl of diesel fuel arctic over several weeks. Impacts to water quality resulting from the *Bahia Paraiso* spill were observed throughout the water column within a few kilometers of the spill, but only in samples directly under visible oil slicks. Hydrocarbon composition in these samples was similar to that of the slick, indicating that oil droplets were present and hydrocarbons had not yet been dissolved (Kennicutt et al. 1991a). Overall, tPAH concentrations were 50–100 µg/L in the first few months following the start of the spill. Weathering mechanisms including evaporation, dilution, and transport from the area reduced the extent of impacts over time such that tPAH concentration were not detected 2–2.5 months after the spill (Kennicutt et al. 1991a).

4.6.7 T/V *Antonio Gramsci*

The grounding and release of 4,200 bbl of crude oil from the *Antonio Gramsci* in the Gulf of Finland occurred under heavy ice conditions, which influenced the fate of the spilled oil. Impacts of the spill persisted for several months and were more pronounced during the spring (3 months after the spill) due to warm weather melting the sea ice and releasing the spilled oil. Laboratory analyses showed that roughly 20% of the spilled oil was recovered by response efforts and about 30% was lost due to natural processes (mostly evaporation) (Hirvi 1990). While the other half of the spilled oil remained in the marine environment, analyses of water column samples showed little evidence of adverse impacts to water quality. In the first month following the spill, average TPH values in surface waters (1 m depth) ranged from 0.7–2.1 µg/L, which is typical of the Gulf of Finland (Hirvi 1990). During and after ice melts, films, emulsions, and tar balls were observed on the water surface for about a month (3-4 months after the spill) (Hirvi 1990). At this point, average TPH concentrations reached their highest with a range of 0.6–17.0 µg/L but quickly returned to background levels about a month later (4 months after the spill) (range=0.5–1.5 µg/L) (Hirvi 1990).

4.6.8 Bohai Sea

A series of oil spills at the Penglai 19-3 oil field in the Bohai Sea of China resulted in the release of 723 bbl of oil and 2,620 bbl of mineral oil-based drilling muds over several days. Following the spill, TPH concentrations in the water column within a 2.2 km radius of the platforms were up to 40.5 times higher than background levels (Wang et al. 2020). The highly dense and viscous nature of the spilled oil limited spreading and evaporation, and weak water currents in the Bohai Bay limited natural rates of oil dilution and dispersion. Additionally, residues of the spilled oil and drilling mud were likely trapped within the top 1.5–2.5 centimeters (cm) of sediments and could have reentered the water column under certain environmental conditions (Wang et al. 2020). These factors led to observed water quality impacts 5 years after the spill, though TPH concentrations (measured by UV-vis spectrophotometry at 225 nanometers) at that time were within range of China's seawater quality standards (50 µg/L). Average TPH concentrations 5 years after the spill were 28.3 µg/L at the water surface (0–0.3 m), 27.7 µg/L at mid-depth (14.0–14.3 m), and 29.5 µg/L at the bottom of the water column (27.5–28.0 m) (Wang et al. 2020). Concentrations were highest in the vicinity of the platform, with levels increasing slightly from the surface (36.1 µg/L) to the bottom (45.3 µg/L) (Wang et al. 2020). Overall, TPH concentrations observed 5 years after the Bohai Sea spills were comparable to levels found in most coastal waters of China.

4.6.9 T/V *Estrella Pampeana*

The collision of the *Estrella Pampeana* resulted in the release of 1,570 bbl of Hydra crude oil and corresponding adverse impacts to water quality off the Rio de la Plata coast in Argentina. Estimates suggest that nearly half of the spilled oil was lost to evaporation before it reached the coast, but persistent adverse impacts to water quality were observed. Sampling conducted 4–36 months after the spill found that hydrocarbon concentrations generally decreased over time until background levels (1 µg/L) were reached, likely due to natural weathering processes (Moreno et al. 2004). Total hydrocarbon levels were as high as 39 µg/L at 0.15 m depths 8 months after the spill. Hydrocarbon concentrations at these sites did not return to background levels until 2.5 years (30 months) after the spill (Moreno et al. 2004).

4.6.10 T/B *North Cape*

The grounding of the *North Cape* barge along the southern coast of Rhode Island resulted in the spill of 19,700 bbl of home heating oil into the surf zone. Immediately after the grounding, high winds and rough seas effectively dispersed the oil throughout the water column, increasing the severity of impacts. Sampling conducted within 12 km of the grounding site at depths of 2–25 m demonstrated relatively high maximum TPH and tPAH concentrations of 3,940 and 115 µg/L, respectively (Reddy and Quinn 1999). One week after the spill, TPH values ranged from 150–820 µg/L (Reddy and Quinn 1999). Overall, average TPH and tPAH concentrations were 483 and 21.9 µg/L, respectively (Reddy and Quinn 1999).

4.7 Summary and Information Needs for Assessing Impacts to Water Quality

Decades of research have facilitated an understanding of the general behavior of oil in the water column. Research discussed in this literature synthesis described how surface releases of oil lead to lateral spreading of oil, the formation of near-surface emulsions, and the weathering of oil by evaporation, photooxidation, and microbial degradation. Environmental conditions influence the fate and persistence of oil. For instance, weak water currents in Bohai Bay limited rates of oil dissolution and dispersion following the Penglai 19-3 spill; after the grounding of the *North Cape* barge, strong winds and seas effectively dispersed the spilled oil throughout the water column (Reddy and Quinn 1999; Wang et al. 2020). Light winds and warm waters in the days after the *World Prodigy* spill of home heating oil increased the rate of weathering such that the duration of adverse impacts to water quality was less than a week (Pilson 1990). In contrast, heavy sea ice conditions in the Gulf of Finland following the *Antonio Gramsci* spill delayed the onset of adverse impacts by several months (Hirvi 1990).

Though the general behavior of oil is fairly universal, and knowledge of the influence of environmental conditions is apparent, adverse impacts to water quality as measured by TPH and PAH concentrations are unique to each spill. As demonstrated in this chapter, the literature on the topic is limited. Immediately following any of the oil spills described in this chapter, TPH concentrations in the surrounding waters could be as high as 697,000 µg/L, as shown following the Refugio spill. Strong gale-force winds contributed to relatively high TPH concentrations of 3,940 µg/L following the *North Cape* spill, which released oil directly into the surf zone, but levels as low as 100 µg/L have also been observed when the oil release occurred during warm weather and light wind conditions, such as after the *World Prodigy* spill. PAH concentrations also vary by spill; tPAH concentrations in the months following the chronic release of a light fuel oil from the *Bahia Paraiso* ranged from 50–100 µg/L, and samples taken adjacent to the release point of the Refugio spill ranged from below detection to 73.2 µg/L. In summary, the literature synthesized in this chapter shows that water quality impacts following median-range oil spills are variable in nature.

There are numerous areas of uncertainty that hinder complete understanding of the impacts that median-range oil spills have on water quality. Oil fate and transport and water quality following an oil spill are often understood through a combination of in-situ field measurements, remote sensing, and modeling. In-situ measurement protocols are not standard across spills, which hinders comparison of results. A common set of sampling protocols, to include instrumentation, sampling frequency over space and time, and analytes, should be established, which can be modified to meet the unique characteristics of each spill. Relying solely on in-situ measurements to develop a detailed understanding of oil in and on water over space and time may not be logistically feasible. Modeling can be paired with in-situ measurements to inform a more detailed understanding of fate and transport and to determine the true extent of oil at concentrations high enough to cause toxicity to biota (Berenshtein et al. 2020). Buoy deployments to measure hydrodynamics and temperature may help reduce uncertainty associated with fate and transport modeling and thus exposure of biota. Remote sensing is a well-established tool for detecting oil on water. The interpretation of remote sensing data would be strengthened by additional bulk oil sampling and quantification of the oil:water ratios to help determine the quantity of oil on the surface after a spill (Abt Associates 2018). In addition, studies should be performed to establish a relationship between satellite detections of oil and areas that are toxic to biota (Berenshtein et al. 2020).

5 Pelagic Plankton Communities

5.1 Habitat Description, Communities, and Ecological Functions and Services

This chapter focuses on the planktonic component of the pelagic community. Plankton are passive swimmers that are carried by oceanic and tidal currents through the water column. Nekton, or active swimmers, are discussed in their respective chapters (fish and invertebrates, sea turtles, and marine mammals). The pelagic zone refers to the offshore water column. **Figure 5-1** illustrates the three dominant zones within the pelagic zone: epipelagic, mesopelagic, and bathypelagic. Plankton are a taxonomically diverse group, exhibiting morphologies that are a function of size and trophic order. Though small, plankton are vital to the marine food web and overall health of marine ecosystems. More specifically, plankton provide essential ecosystem services to pelagic habitats, such as nutrient transfer and cycling, trophic support for fish, invertebrates, and marine mammals, and fisheries production. Due to their size, ability to rapidly reproduce, and exposure to environmental surroundings, plankton can be highly sensitive to seawater temperature, salinity, pH, and nutrient concentrations. Small changes to these environmental factors may influence plankton assemblages.

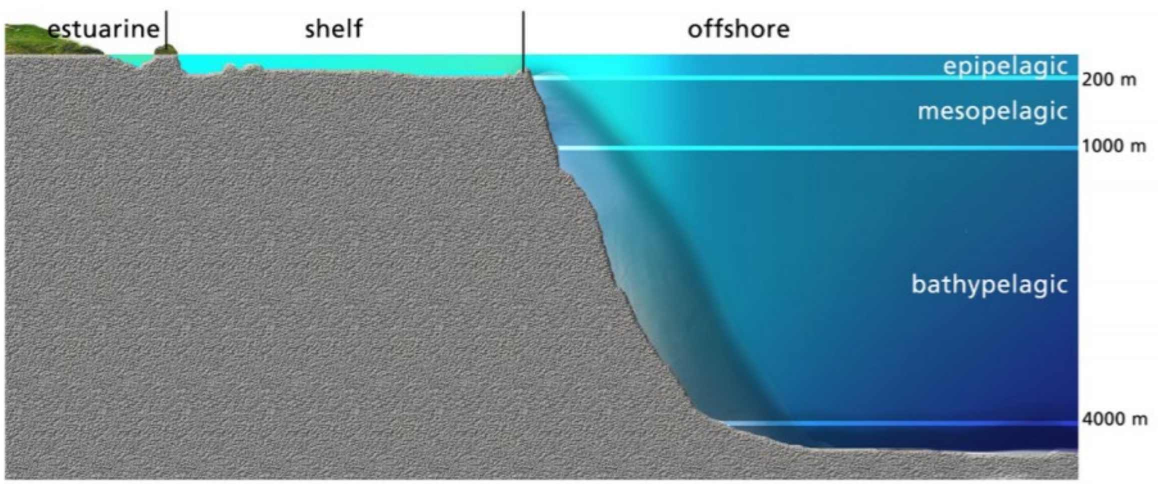

Figure 5-1. Pelagic zones of the water column
The pelagic zone is the offshore water column and is divided into three depth zones: epipelagic, mesopelagic, and bathypelagic. From DWH NRDA Trustees (2016).

The two most studied types of plankton are phytoplankton and zooplankton. As autotrophic members of the plant kingdom, phytoplankton utilize the process of photosynthesis to produce energy. Photosynthetic organisms require carbon dioxide, water, and sunlight to convert light energy into usable chemical energy (glucose) with oxygen as a byproduct. These single-celled organisms live primarily in the epipelagic zone of the water column, where photosynthetic pigments (e.g., chlorophyll-a) can access sunlight. Phyto-plankton community composition can include diatoms, dinoflagellates, cyanobacteria, and chrysophytes.

Unlike phytoplankton, zooplankton are heterotrophic and must rely on other organisms, such as phytoplankton and smaller zooplankton, for energy. Zooplankton encompass a wide range of animal species, including ichthyoplankton (fish eggs and larvae), single-celled protozoans, crustaceans, mollusks, and coelenterates, among others. Found throughout the pelagic zone, some zooplankton migrate vertically to feed in the epipelagic zone. Size classifications used to differentiate zooplankton are: microzooplankton (20–200 µm), mesozooplankton (200 µm–2 cm), macrozooplankton (2–20 cm), and megazooplankton (20–200 cm). Like phytoplankton, environmental conditions and seasonal variations may alter zooplankton species composition and relative abundance. Their sensitivity to environmental change and ability to rapidly reproduce make them indicators of environmental change.

Phytoplankton and zooplankton support the marine food web by supplying carbon and nutrients to higher trophic organisms. Although simplified, **Figure 5-2** illustrates the importance of plankton communities to biogeochemical processes in the marine ecosystem. As primary producers, phytoplankton contribute heavily to global primary production, transferring large amounts of organic carbon compounds to higher, heterotrophic organisms, such as grazing zooplankton (Falkowski et al. 1998; Field et al. 1998).

Zooplankton are vital in transferring and cycling nutrients horizontally (between nearshore and offshore areas) and vertically (between epipelagic, mesopelagic, and bathypelagic zones) in pelagic habitats. As secondary producers, zooplankton act as both predator and prey in the food web and contribute to active (e.g., vertical movement) and passive (e.g., sinking of fecal matter) transfer of energy from the upper portions of the water column to the deep waters. Zooplankton are an important food source for pelagic fish and crustacean species as well as birds and marine mammals.

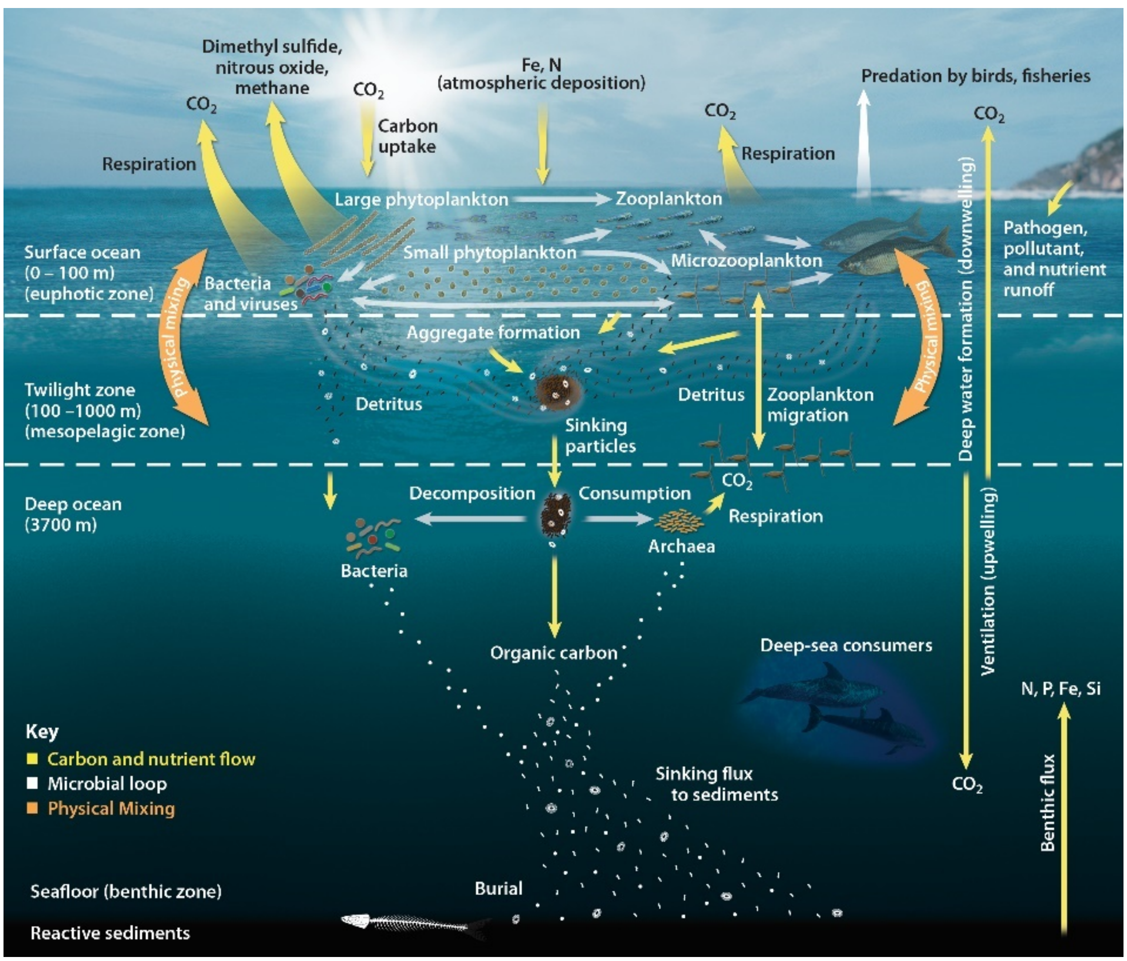

Figure 5-2. Simplified biogeochemical diagram of the marine food web
Plankton play a vital role in the marine food web, primarily contributing to vertical and horizontal nutrient transfer and cycling. From U.S. Department of Energy Office of Biological and Environmental Research (2008).

5.2 Exposure Pathways in Pelagic Plankton Communities

Because plankton are ubiquitous in the water column, they inevitably become exposed to oil during surface and subsurface spills. Environmental factors and oil spill characteristics influence the degree of exposure within the plankton community. For a detailed discussion on oil behavior and persistence in the water column, see Chapter 4 – Water Quality.

Plankton can contribute to the vertical transport of oil in the water column. Studies of several spills found that zooplankton can become contaminated with oil internally and externally (Almeda et al. 2014a; Almeda et al. 2014b; Lee et al. 2012). They can then transport oil through the water column by vertical migrations. Plankton can also be incorporated into marine snow, thereby transporting oil to the seafloor (Yilmaz and Isinibilir 2018). The contribution of marine snow can be substantial; during the *Deepwater Horizon* oil spill, 15–30% of the oil was transported from the marine surface to the seafloor as marine snow (Chanton et al. 2015; Passow and Ziervogel 2016; Romero et al. 2017; Yan et al. 2016).

In a study by Mitra et al. (2012), zooplankton passively absorbed or ingested oil, which contributed to bioaccumulation of PAHs within the zooplankton community. Other studies found that zooplankton ingested oil in the presence of other food sources (Almeda et al. 2014a; Ortmann et al. 2012). Zooplankton uptake of oil is one mechanism by which oil may become incorporated into the marine food web, transferring oil-derived carbon to higher trophic organisms (Graham et al. 2010).

Figure 5-3 Figure 5-3 illustrates the impacts of spills on plankton in the marine ecological environment (Tang et al. 2019). Oil may decompose into smaller compounds through weathering processes or undergo biodegradation. These mechanisms can increase nutrient availability within the water column and support conditions for phytoplankton blooms.

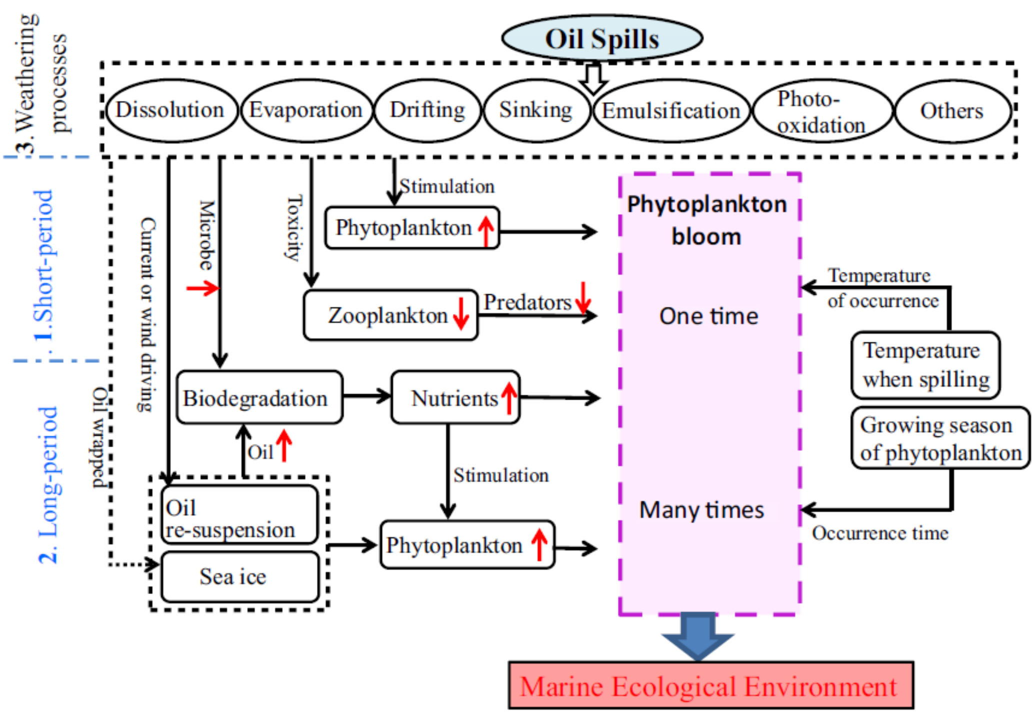

Figure 5-3. Conceptual model of short- and long-term impacts to plankton from oil spills
Black arrows indicate relationships, while dashed black lines suggest potential relationships. Red arrows indicate increased (up arrows) and decreased (down arrows) effects. From Tang et al. (2019).

5.3 Impacts of Oil Exposure and Treatment to Pelagic Plankton Communities

This section reports on the extent of impacts and recovery time for pelagic plankton communities exposed during oil spills. Few field studies have examined the impacts of oil spills 500–20,000 bbl on plankton. Data were identified from six spills of crude oil or diesel. No condensate studies documenting impacts to plankton were identified. Common metrics of pelagic community impacts and recovery are biomass, relative abundance, and species or taxonomic composition, as well as primary production and chlorophyll-a for phytoplankton. In this section, changes to these metrics are considered to be impacts, whether the metrics show an increase or decrease. The detection of oil in the plankton community alone is not considered to be an impact. The definition of recovery varies by study and by the species or taxonomic groups and metrics evaluated, but in general is considered to be a return to baseline conditions. This section summarizes and synthesizes the limited available information and acknowledges that recovery in this context does not necessarily equate to full ecological or ecosystem recovery. For pelagic communities, few studies define specific timelines of recovery, as nearly all studies that found impacts were not continued until baseline conditions were achieved.

Field studies of plankton were conducted in association with six oil spills: *Sefir*, Bohai Sea, *North Cape, World Prodigy*, *Estrella Pampeana*, and *ARCO Anchorage* (**Table 5-1** and **Figure 5-4**). Impacts were detected for the majority of these spills, including internal and external zooplankton contamination and increased phytoplankton biomass. Impacts were not observed in studies following the *ARCO Anchorage* and *Estrella Pampeana* spills (Kittle Jr. et al. 1987; Moreno et al. 2004).

Adverse impacts to zooplankton were observed in the immediate aftermath of the *Sefir* spill of a No. 1 fuel oil (Linden et al. 1983). Field samples were collected 3, 5, and 12 days after the start of the spill (Linden et al. 1983). Linden et al. (1983) evaluated the presence of oil droplets in zooplankton and found that zooplankton were contaminated internally and externally (specifically at feeding appendages) on each sampling date). The degree of contamination varied with distance from spill site, with 10–50% of plankton dead or dying due to exposure to this light refined oil (Linden et al. 1983).

Short-term phytoplankton impacts were minor following the *World Prodigy* spill (Pilson 1990). Impacts to zooplankton and lobster larvae were ephemeral, with reduced abundance 3 days after oiling, but a return to baseline conditions 5 days after the spill (Pilson 1990). For the *North Cape* spill, computer models were utilized to estimate injury to plankton (NOAA 1999). Impacts to phytoplankton and zooplankton were predicted to be minor (categorized as negligible relative to total production in Block Island Sound), with full recovery occurring 3–6 months following the spill (NOAA 1999). For the *North Cape* NRDA, impacts to total fish populations were determined to recover without intervention, with natural recovery to baseline in 1–2 years (NOAA 1999). Phytoplankton blooms were observed following the Bohai Sea spill at 20 days, 12 months, and 14 months following the spill, possibly due to reduced grazing pressure from zooplankton (Zhou et al. 2014).

Plankton impacts were not detected in two of the six oil spills studied. Plankton survival was not affected by the *Estrella Pampeana* spill (Moreno et al. 2004). Plankton communities were evaluated in canals and creeks, which are typically ecologically important breeding grounds, compared with areas that exhibited no oiling (Moreno et al. 2004). The community assemblages in both oiled and unoiled areas were dominated by diatoms and rotifers (Moreno et al. 2004). Similarly, no adverse effects to ichthyoplankton (including herring, smelt, and sand lance) and other zooplankton were observed after the *ARCO Anchorage* spill between Port Angeles and Dungeness Bay (Kittle Jr. et al. 1987). Abundance of ichthyoplankton was high, suggesting no harmful impacts to plankton communities (Kittle Jr. et al. 1987).

Table 5-1. Field studies with documented or estimated impacts to and recovery of plankton communities from crude oil or diesel spills (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Oil Cleanup	Plankton Groups	Documented Effect/Impacts	Recovery (months)
1985 T/V *ARCO Anchorage*, Puget Sound, WA[a]	5,690 bbl Alaska North Slope crude	Yes, agitation	Total plankton community, including ichthyoplankton	Impacts were not observed several months following the spill.	0
1999 T/V *Estrella Pampeana*, Rio de la Plata, Argentina[b]	15,700 bbl Patagonia crude	Yes	Diatoms and rotifers	Impacts were not observed over a period of 2 years after the spill.	0
1989 T/V *World Prodigy*, Narragansett Bay, RI[c]	6,500 bbl home heating oil	Yes	Total community, including ichthyoplankton	Minor short-term phytoplankton impacts were observed. Zooplankton and lobster larvae showed reduced abundance 3 days after oiling but were recovered after 5 days.	0.2
1981 T/V *Sefir,* Baltic Sea, Sweden[d]	2,800 bbl No. 1 fuel oil and diesel, leaked from sunken vessel for 6 weeks	Yes	Copepods and ichthyoplankton (crustacean and fish larvae)	Zooplankton were internally and externally contaminated with oil droplets on day 3, 5, and 12 of the study. Degree of contamination varied with distance from spill site, but throughout the study, 10–50% of plankton were dead or dying due to oil exposure.	>0.4
1996 T/B *North Cape*, South Kingstown, RI[e]	19,700 bbl home heating oil	No	Total community, including ichthyoplankton	Minor impacts were modeled for zooplankton and phytoplankton, while an evaluation of impacts to fish, including ichthyoplankton, predicted recovery in 1–2 years.	Plankton (non-ichthyo-plankton): 3 to 6 Ichthyo-plankton:12 to 24
2011 Bohai Sea, China[f]	3,200 bbl of crude and drilling muds	Yes	Total phytoplankton community	Phytoplankton blooms were detected 20 days, 12 months, and 14 months after the spill.	>14

[a]Kittle Jr. et al. (1987); [b]Moreno et al. (2004); [c]Linden et al. (1983); [d]Pilson (1990); [e]NOAA (1999); [f] Zhou et al. (2014)

Overall, plankton communities have typically demonstrated short-term impacts after oiling, ranging from days to months. Most of the studies found either minor impacts or impacts were not observed to plankton. Zooplankton were found to be contaminated with oil internally and externally, demonstrating a pathway for contamination of their predators (Linden et al. 1983).

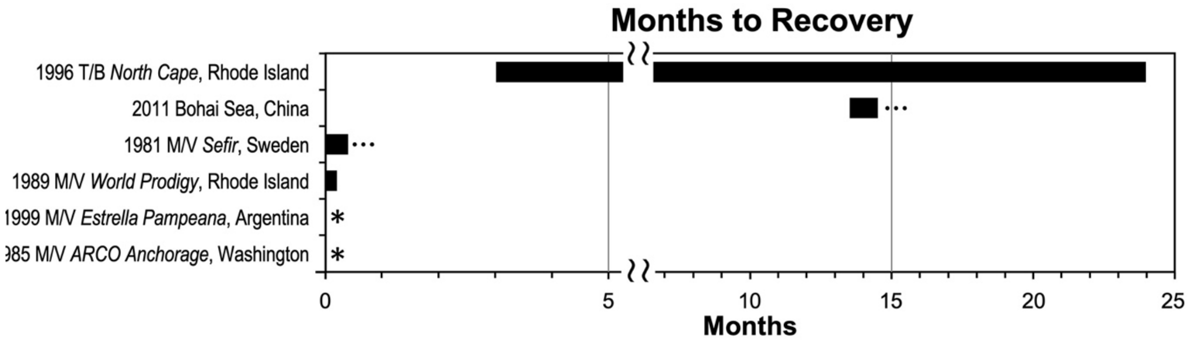

Figure 5-4. Recovery of phytoplankton and zooplankton communities following the oil spills listed in Table 5-1
Asterisk indicates impacts were not detected. Dotted lines indicate incomplete recovery at the time of the most recent study.

5.4 Summary and Information Needs for Assessing Impacts on Pelagic Plankton Communities

Impacts on plankton were detected following four of the six spills of crude oil or diesel studied, including increased phytoplankton biomass and internal and external zooplankton contamination. Impacts were not observed among two spills of crude oil (*ARCO Anchorage,* and *Estrella Pampeana* spills). The differing results may be due to variations in community assemblages, oil spill characteristics (e.g., type of oil, environmental conditions), or study design.

Few field studies have evaluated the effects of oil spills on plankton communities. It is recommended that additional field studies be conducted following oil spills. Those studies should employ consistent methodology to enable comparison of results across studies and a better understanding of causes for the varied response across spills (e.g., water quality data should be collected in tandem). To assess impacts more accurately to and recovery of plankton communities following an oil spill, more year-round, baseline monitoring of plankton biomass and community assemblage should be conducted to serve as a comparison to post-spill data. In addition, studies should be continued until baseline conditions are achieved to define the time to recovery.

6 Marine Benthic Communities

6.1 Habitat Description, Communities, and Ecological Functions and Services

This chapter includes marine hardbottom and subtidal soft-sediment communities. Substrate type, depth, and light penetration play prominent roles in determining the types of communities present in different benthic habitats. Hardbottom communities discussed in this chapter include shallow coral reefs, subtidal rocky reefs, and oyster reefs. Subtidal soft-sediment communities are also discussed. No studies of the impacts of median-range oil spills of crude oil, condensate, or diesel on mesophotic coral reefs, deep-sea coral communities, or rhodolith beds were found in a systematic review of the literature. Kelp, submerged aquatic vegetation (seagrass), and intertidal communities (beaches, tidal flats, rocky shorelines, marshes) are discussed in separate chapters.

6.1.1 Hardbottom Communities Description and Ecological Functions and Services

Coral reefs are found in shallow, tropical waters with high light penetration, low turbidity, low nutrients, and open-ocean salinity. Reef-building corals are colonial marine invertebrates that house symbiotic photosynthetic algae in their tissues that provide nutrition. These corals deposit calcium carbonate exoskeletons that form the reef structure. This three-dimensional structure supports the highest taxonomic diversity of any marine ecosystem. Common fauna include sessile invertebrates (hard and soft corals, anemones, sponges, bryozoans, ascidians, bivalves, and polychaetes, among others), mobile invertebrates (polychaetes, gastropods, crustaceans, and echinoderms, among others), and macroalgae. The four OCS regions of the U.S. (Gulf of Mexico, Atlantic, Pacific, and Alaska), shallow coral reefs occur as barrier reefs, platform reefs, fringing reefs, and patch reefs. Prominent coral reef areas within U.S. OCS jurisdiction include the Florida Keys Reef Tract and the Flower Garden Banks (offshore of Texas). Seven species of coral in Florida and the Caribbean are listed as threatened or endangered under the Endangered Species Act (ESA): staghorn (*Acropora cervicornis*), elkhorn (*A. palmata*), pillar (*Dendrogyra cylindrus*), lobed star (*Orbicella annularis*), mountainous star (*O. faveolata*), boulder star (*O. franksi*), and rough cactus (*Mycetophyllia ferox*).

Coral reef zones (**Figure 6-1**) are defined by abiotic factors such as depth and wave exposure that influence the biotic communities inhabiting each zone. The reef flat is the most shoreward zone and is a shallow, sheltered environment with relatively low coral diversity. The reef crest is seaward of the reef flat and is the shallowest zone with the highest wave exposure; it may be exposed at low tide. Reef crests are commonly dominated by coralline algae and low-profile corals. The fore reef is seaward of the reef crest and is deeper and less exposed to wave energy. It has the highest coral cover and diversity of any zone. In addition to containing high biodiversity, coral reefs provide many ecosystem services that make them economically valuable, including fish production for commercial and recreational fisheries, coastal protection, and tourism and recreation.

Subtidal rocky reefs include hard substrate benthic areas below the low tide mark, including sloping bedrock, rock walls, and boulder fields across all latitudes, but excludes carbonate coral reefs. Prominent areas of subtidal rocky reefs within the U.S. OCS jurisdiction include the waters off the west coast and the northeast coast (e.g., the Gulf of Maine). Community composition on subtidal rocky reefs is greatly influenced by the slope of the rock bottom, with horizontal and gently sloping areas in temperate zones dominated by macroalgae and vertical rock walls dominated by epifaunal invertebrates. This difference is in part due to the higher light levels on horizontal surfaces. Other factors that contribute to differences in

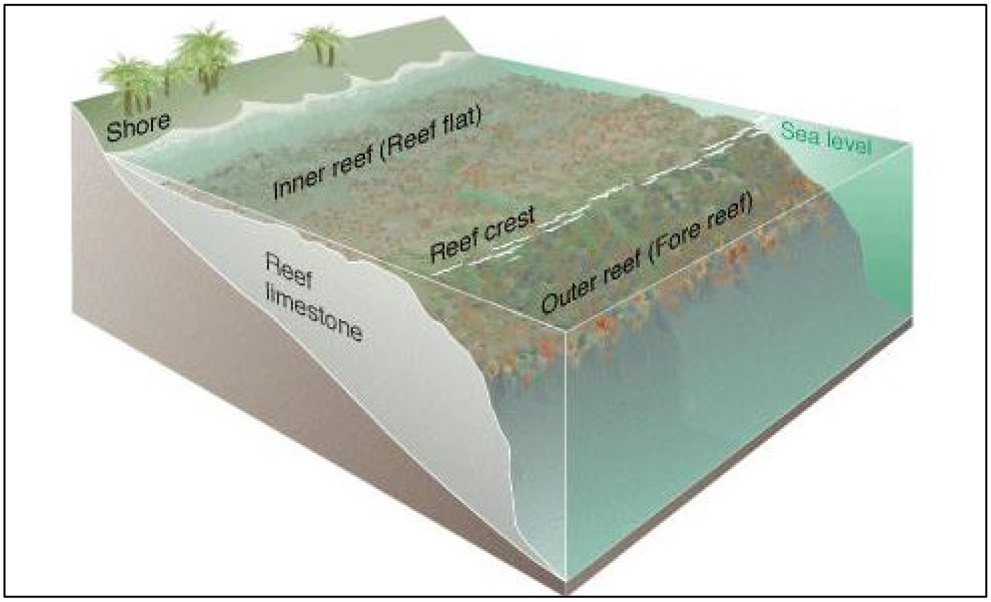

Figure 6-1. Coral reef zones (reef flat, reef crest, and fore reef) as illustrated on a fringing reef
From Field et al. (2002).

community structure by substrate angle include increased sedimentation and higher rates of physical and biological disturbance on horizontal substrates than on vertical walls. Dominant taxa on subtidal rocky reefs include kelp, red algae, sponges, sea anemones, soft corals, mussels, sea stars, brachiopods, ascidians, and sea urchins. Kelp is absent from the tropics and erect macroalgae are much less abundant, so horizontal surfaces in the tropics are often dominated by hard and soft corals.

Encrusting invertebrates are more common on vertical surfaces (Bertness et al. 2001). Ecosystem services provided by subtidal rocky reefs include: providing habitat for high diversities of fish and invertebrates, providing spawning locations for many fish species, supporting key foraging areas for marine mammals and diving birds, and supporting high ecosystem productivity.

Oyster reefs are biogenic habitats formed from the accumulation and fusing of live and dead oyster shells as oysters grow. They are found in both intertidal and subtidal zones in temperate and sub-tropical estuaries worldwide. Notable areas of oyster reefs within the U.S. OCS jurisdiction include estuaries along the Gulf (e.g., Galveston Bay, Apalachicola Bay, etc.) and east (e.g., Chesapeake Bay, South Carolina, Georgia, etc.) coasts. Oyster reefs are functionally extinct throughout much of the U.S. west coast, but restoration efforts, particularly in the Pacific Northwest, are underway. On the east and Gulf coasts, the eastern oyster (*Crassostrea virginica*) is the dominant species; whereas, on the west coast, the Olympia oyster (*Ostrea lurida*) is the native species. In many places on the west coast, the Olympia oyster has been displaced by the Pacific oyster (*Crassostrea gigas*). Oyster reefs provide a hard-substrate habitat in areas that otherwise consist of soft sediments. Oyster reefs augment the biodiversity of the regions where they occur, as they provide substrate for settlement of epibiotic invertebrates and provide habitat for fishes and mobile invertebrates. They also serve as nurseries for commercially important species. As suspension-feeding bivalves, oysters filter estuarine waters. This decreases particulate matter and increases light penetration, which in turn can enhance the growth of seagrass beds. Their filtration activities also play an important role in nutrient cycling. Further, oyster reefs provide storm protection, stabilize coastal shorelines, and reduce erosion of coastal areas by decreasing wave energy and trapping sediment.

6.1.2 Subtidal Soft-Sediment Communities Description and Ecological Functions and Services

Marine sediments cover over 80% percent of the ocean floor and form a three-dimensional, fluid habitat. Subtidal soft-sediment communities are largely characterized by macrofauna (generally defined as invertebrates retained on a 0.5 mm screen) and meiofauna (generally defined as invertebrates that pass through a 0.5 mm screen but are retained on a 44 µm screen). Macrofauna are dominated by invertebrate species such as polychaetes, echinoderms, crustaceans, and mollusks, which may be suspension-feeders, deposit-feeders, or predators. They may be epifauna (animals living on the substrate) or infauna (animals living in the substrate). Meiofauna are generally considered to be infauna and include organisms such as nematodes, crustaceans (notably copepods), and polychaetes. Many macrofauna species are part of the meiofauna in their juvenile stages, which are referred to as "temporary meiofauna." In contrast, "permanent meiofauna" are species that stay in the meiofauna size range as adults as well. Organisms that inhabit soft sediments impact the condition of their surroundings, either through bioturbation (e.g., burrowing, digging), sediment stabilization (e.g., creating structures, producing fecal pellets that bind sediments), or irrigating sediments (e.g., creating burrows that allow water to flow through sediments). Ecosystem services provided by subtidal soft-sediment communities include: supporting fisheries of crabs, shrimps, and bivalves; organic matter and nutrient cycling; turnover and oxygenation of sediments; pollutant detoxification; food web support; and trophic linkage of benthic and pelagic systems.

6.2 Oil Behavior and Persistence in Marine Benthic Communities

Most oil spills of crude oil, condensate, or diesel, float. However, spilled oil can reach marine benthic communities by several mechanisms. Gustitus and Clement (2017) presented a conceptual model of oil-sediment interactions in nearshore environments (**Figure 6-2**). Microscopic oil-mineral aggregates (OMAs) form when oil that is physically dispersed in the water column as oil droplets interacts with very fine (<10 µm) suspended particles (Gustitus and Clement 2017). OMAs may remain suspended in the water column or may settle to the seafloor, depending on the ratio of oil to mineral in the aggregate. Because OMAs are more buoyant than sediments alone, they remain in suspension longer and may travel greater distances (Stoffyn-Egli and Lee 2002). The formation of OMAs is not considered to be a pathway of oil sedimentation for surface oil spills in offshore areas because suspended inorganic particle concentration is relatively low in the open ocean (Daly et al. 2016). Macroscopic sediment-oil agglomerates (SOAs) are formed when viscous oil interacts with coarse sediments as a result of wave action. Once the oil picks up enough sediment for its density to be greater than that of sea water, the SOA sinks and settles on the seafloor. Sunken oil may aggregate on the seafloor in nearshore troughs, inside estuaries, or other lower-energy settings, forming larger agglomerates called sediment-oil mats (Gustitus and Clement 2017).

Shallow benthic communities can also be exposed via oil in the water column. This pathway becomes important in cases with rough seas (or a turbulent release) and a lighter, more soluble oil. Turbulent conditions cause oil to naturally mix into the water column, either in solution or as physically suspended droplets.

In the deeper, offshore zone, contaminated marine snow is an important oil exposure pathway to the benthos. See the discussion in Section 4.2 and **Figure 4-3** that shows a schematic of marine oil snow formation. In most cases, marine oil snow has a combined density greater than seawater, and the sinking particles can be a major mechanism of the transport of oil to the seafloor (Vonk et al. 2015).

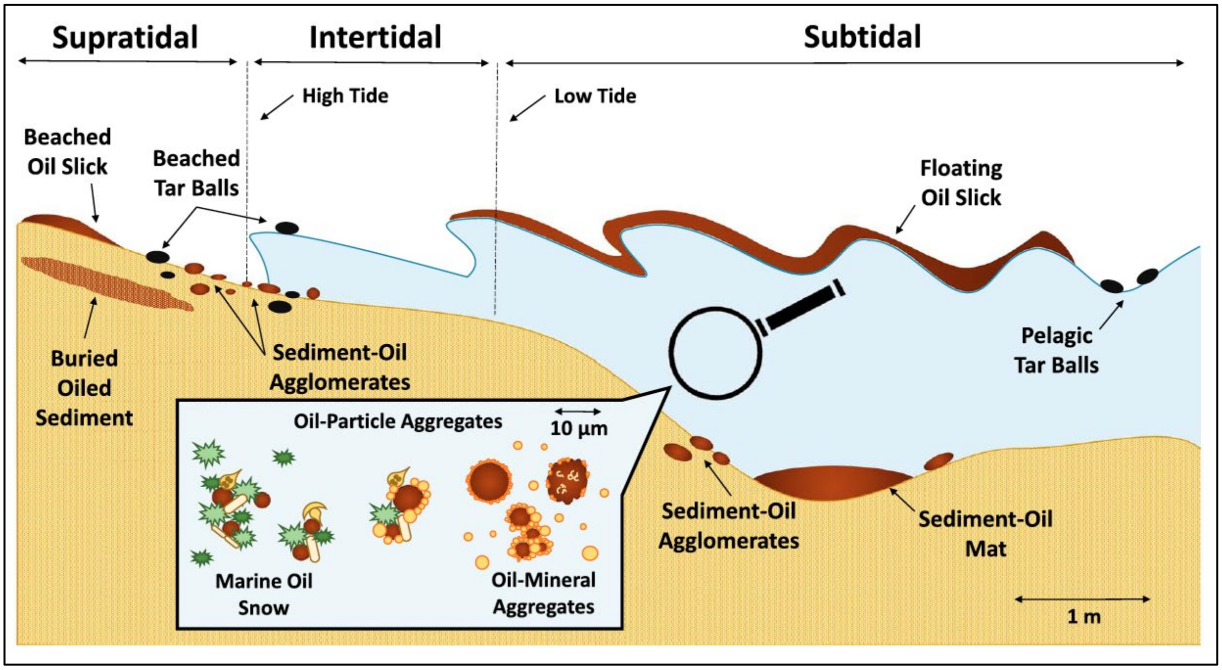

Figure 6-2. Conceptual model of the size and location of various oil residues in the nearshore environment
From Gustitus and Clement (2017).

Persistence of oil in benthic habitats is driven by environmental conditions and level of contamination, and oil can persist in sediments for months to years. The rates of chemical and physical oil decomposition are driven by temperature, with warmer temperatures accelerating the rates of these processes (Soto et al. 2014; Tansel et al. 2011). Rates of microbial degradation of oil are influenced by temperature and oxygenation: higher temperatures and levels of oxygenation in the substrate lead to higher degradation rates (Leahy and Colwell 1990) and lower levels of oxygenation in anaerobic sediments lead to longer oil persistence and chronic exposure to the benthos. Study of the degradation rate of low molecular weight PAH in subtidal sediments following the *North Cape* spill found that low molecular weight PAH should persist longer in organic-rich sediments than in sediments with low total organic carbon (Hinga 2003). PAH concentrations in sediments at two different sites peaked 6 and 33 days following the *North Cape* spill, then fell to stable low levels by 33 and 189 days, respectively. The site with the maximum concentration and longer persistence had contamination up to an order of magnitude greater than the other site (Ho et al. 1999). This finding agrees with other oil spill studies that have found that hydrocarbon persistence is greater in more heavily contaminated sediments (e.g., Bagby et al. 2014).

Relative to large oil spills (>20,000 bbl), median-range spills (500–20,000 bbl) have shorter PAH persistence in subtidal sediments. Sampling of subtidal sediments showed that PAH remained elevated for weeks or months after many median-range spills, including the *ARCO Anchorage* (Lindstedt-Siva et al. 1987; Miller 1989), *North Cape* (Ho et al. 1999), and *Bahia Paraiso* (Kennicutt et al. 1991b). Similar studies after large spills reveal persistence in subtidal sediments for years (e.g., Guzmán et al. 1994; Reuscher et al. 2017; Soto et al. 2014).

In addition to level of contamination, oil persistence can also be influenced by bioturbation in the benthic community. These effects are dependent on the initial distribution of oil. When sediments have an oil-contaminated subsurface layer, bioturbating infauna can bury the oil deeper, prolonging oil persistence.

Conversely, bioturbation can enhance physical removal and microbial degradation in subsurface oil layers and oil deposited on surface sediments (Timmermann et al. 2011).

6.3 Impacts of Oil Exposure and Response Actions on Hardbottom Communities

6.3.1 Impacts of Oil Exposure and Response Actions on Coral Reefs

Table 6-1 and **Figure 6-3** summarize the results from studies on the impacts to coral reefs from the *Garbis* and Rose Atoll oil spills and the TROPICS field experiment. The *Garbis* spill consisted of crude oil emulsion, and the Rose Atoll spill consisted of diesel and lube oil. The TROPICS experiment released crude oil. No studies were found that describe the impacts of condensate spills of 500–20,000 bbl on coral reefs.

Table 6-1. Studies with documented or estimated impacts to and recovery of coral reefs from spills of crude oil or diesel (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Studied Species	Documented Effect/Impacts	Recovery (years)
1975 *Garbis*, Florida Keys[a]	1,500 – 3,000 bbl crude oil emulsion	Corals	Impacts to coral reef communities were not observed immediately following spill, or at 2 weeks, 3.5 months, or 6 months post spill.	No impact measured
1993 Rose Atoll (*Jin Shiang Fa*), American Samoa[b]	2,380 bbl diesel and lube oil	Algae, corals, echinoderms (sea urchins and sea cucumbers), giant clams	Algae: Massive die-off of dominant crustose coralline algae on reef flat and reef slope 2–3 weeks post spill. Filamentous cyanobacteria became established in the weeks after the spill. Only minor recovery of crustose coralline algae and cyanobacteria and opportunistic algal species were dominant 2 and 3 years post spill, Corals: Injury and mortality were moderate to high up to 1 km from wreck site 6 months post spill. Echinoderms: Boring urchins were absent from areas near the wreck and spill site 2 weeks, 2 years, and 3 years after the spill. Unclear if variation in sea cucumber density was due to the spill or natural variation. Giant clams: 70% of giant clams died near the wreck site, presumably as a result of the spill, 2 weeks post spill; whereas only 1% appeared to have died because of the spill at a further site. Effects of spill on giant clam distribution and abundance around the atoll were still observed at 12–18 months. Two years post spill, giant clam recruits were common at most transects at the wreck site, suggesting recovery was underway.	Algae: >3 Corals: >0.5 Sea urchins: >3 Giant clams: >2
1984 TROPICS experiment, Bocas del Toro, Panama[c]	4.5 bbl dispersed and 6 bbl non-dispersed Prudhoe Bay crude	Reef corals, animals, and plants	Slight decrease in coral cover at non-dispersed oil site at 12 months. No substantial differences in coral growth rates between oil and control treatments. No measurable differences in cover of corals, plants, or between treatment and control sites at 10 years or 18 years. Increased coral cover at 25 years at all sites, likely unrelated to oil spill treatments. No measurable differences in cover of coral, plants, soft corals, or other organisms between sites at 32 years.	<10

Source: [a]Chan (1977); [b]USFWS (1997); [c]Ballou et al. (1987); DeMicco et al. (2011); Dodge et al. (1995); Renegar et al. (2017); Ward et al. (2003)

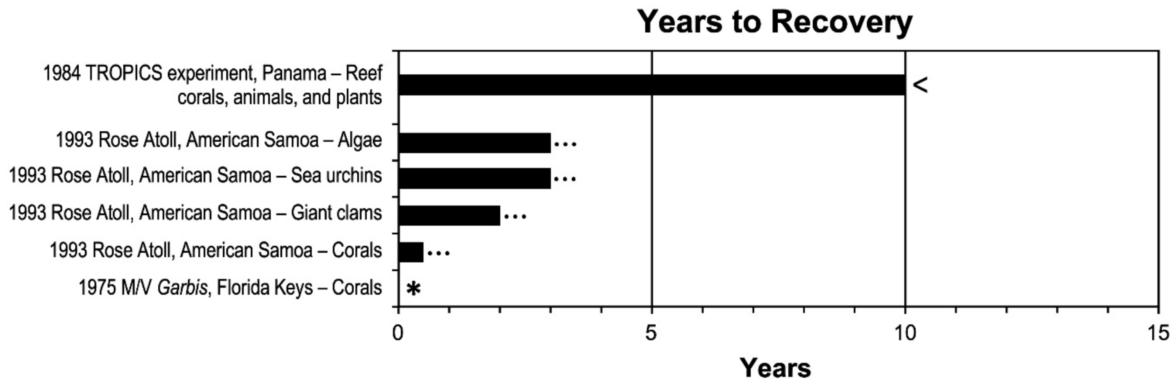

Figure 6-3. Recovery of coral reefs for the oil spills listed in Table 6-1
Asterisk indicates impacts were not detected. Dotted lines indicate incomplete recovery at the time of the most recent study. < symbol indicates recovery was complete at some point before the time shown.

In October 1993, the longliner *Jin Shiang Fa* ran aground at Rose Atoll in the South Pacific spilling 2,380 bbl diesel and lube oil. Heavy wave action forced oil into the water column and the reef structure, and oil was trapped in the reef matrix for at least 2–3 weeks. Oil persisted in reef sediments for at least 22 months. Crustose coralline algae began to die off on the reef flat and reef slope 2–3 weeks after the spill, and little recovery was observed over the next 3 years. The coral reef at Rose Atoll is primarily built by crustose coralline algae rather than hermatypic (reef-building) corals, and these algae provide the structure for the reef. Thus, the lack of recovery observed in crustose coralline algae portends persistent impacts for the entire ecosystem. Studies suggested that the corrosion of iron from the wreck contributed to the bloom of opportunistic algae that may inhibit the recovery of crustose coralline algae. Six months after the spill, a qualitative survey revealed that coral injury and mortality on the reef flat, reef slope, and in the lagoon were moderate to high up to 1 km from the wreck site. Coral injury was attributed to physical injuries from the ship grounding and associated debris, toxicity from the oil, and anoxia caused by decomposition of oil and dead reef organisms. Boring sea urchins were abundant on the outer reef flat at Rose Atoll, but no boring urchins were found from 60 m south to 90 m north of the wreck site 2 weeks after the grounding. Density of boring urchins increased with increased distance from the wreck site. Boring urchin mortality was attributed to both scour from the grounding and toxicity of the spilled oil. No boring sea urchin recovery was observed 3 years later. Many recently dead or stressed giant clams were observed 2 weeks post spill around the wreck site but not at sites distant from the wreck. Clam recruits were observed at the wreck site 2 years later, but no recruits were found along the transect that ran through the wreck debris. Recruitment to the site suggests that recovery was underway for all areas except those directly impacted by debris (USFWS 1997). All studies of impacts and recovery at Rose Atoll were impeded by a lack of baseline ecological data before the incident and by the lack of suitable control sites due to the small size of the reef. Thus, the conclusions above were drawn from data collected at varying distances from the wreck, rather than at impacted and control sites.

No damage to coral reef communities was observed following the *Garbis* spill, though physiological effects on corals were not measured. Calm seas at the time of the spill likely reduced physical contact and deposition of oil on corals and other reef organisms (Chan 1977).

In the TROPICS experiment, researchers exposed coral reefs, seagrass beds, and mangrove forests to oil or dispersed oil that was released over several hours and contained at the experimental sites within oil spill containment boom for 2 days. A control site with no oil release was used for comparison. Only results of the oil and no oil treatments are discussed here. Effects of the oil treatment on the coral reef were minor or not detectable. A slight decrease in coral cover was observed for up to 1 year at the oil treatment site, but there was no statistically significant difference in coral growth rates between the oil and control treatments during this time (Ballou et al. 1987).

The variable results of field studies of the impacts of median-size oil spills on coral reef communities reveal the unique nature of each oil spill and the limitations of attributing ecological changes to spills in the presence of other variables. Physical and environmental differences between spill times and locations, pre-spill baseline ecological conditions, presence of other pre-existing stressors, and type and behavior of the spilled oil make comparisons between spills on coral reefs difficult (Turner and Renegar 2017).

In instances where corals are impacted by oil spills, effects can be both lethal and sublethal. Decreases in coral cover, diversity, and abundance, and increased coral mortality have been observed in field studies. Sublethal effects included bleaching, tissue swelling, tissue loss, mucus production, and bacterial infections (Turner and Renegar 2017).

In the absence of field studies on the impacts of oil exposure on coral reproductive life history stages, laboratory studies with realistic exposures can inform understanding of these impacts. Overall, coral gametes and larvae are more sensitive to contaminants than adult corals; exposing corals to oil at any point in the reproduction process can impact reproductive output and success (Turner and Renegar 2017). Goodbody-Gringley et al. (2013) found that larvae of a brooding species (*Porites astreoides*) had decreased settlement and survival after exposure to 0.62 ppm crude oil. Larvae of a broadcast–spawning species (*Montastraea faveolata*, now named *Orbicella faveolata*) had decreased settlement and survival after exposure to 0.65, 1.34, and 1.5 ppm crude oil. The negative effects of oil exposure on larval settlement continued after direct exposure ended. Hartmann et al. (2015) found that settlement of *Orbicella faveolata* and *Agaricia humilis* larvae decreased by 85% and 40%, respectively, after exposure to oil–contaminated seawater ended. *Acropora tenuis* larvae had inhibited metamorphosis when exposed to condensate concentrations similar to those found in seawater following large spills (Negri et al. 2016). Turner and Renegar (2017) provided a review of additional oil toxicity studies on corals.

The reproductive strategy of the coral (i.e., brooders compared to broadcast spawners) influences the level of impact of oil on reproductive success. The gametes of broadcast spawners typically rise to the water surface, where they have a higher probability of encountering oil slicks. Conversely, brooding species do not disperse their larvae into the water column, thereby reducing the probability that they encounter oil. The timing of a spill also influences the level of impacts. For instance, if a spill occurs during coral spawning season, the recruitment for that entire year could be affected (Shigenaka 2001).

The environmental conditions common to tropical coral reefs increase the toxicity of oil to reef organisms. High levels of ultraviolet radiation, often occurring on coral reefs, increase the toxicity of oil components on corals by an average of 7.2-fold (Nordborg et al. 2020). This phototoxicity also reduces coral settlement success (Nordborg et al. 2018) and developmental success (survival and metamorphosis) of coral larvae (Overmans et al. 2018). High temperature and ocean acidification can also increase oil toxicity to corals, by an average of 3.0- and 1.3-fold, respectively. Both conditions are expected to increase in intensity and frequency with climate change, possibly increasing the toxicity of oil to coral reefs over time (Nordborg et al. 2020).

In addition to the impacts of the spilled oil, impacts to coral reefs may occur through response actions. Physical damage to corals may occur due to response vessel groundings, anchors, booms, dragging lines, and direct contact by response workers. Response vessel operators must proceed carefully to avoid grounding and prop wash on the reef. Using floating lines that do not drag over the reef is the preferred technique during salvage operations to minimize physical impacts; anchoring of boom must be done carefully not to damage corals and to avoid grounding and entanglement on the reef. Also, skimming and placement of certain types of boom should only be performed in deeper waters (>3 m) to avoid directly impacting corals. When spills occur on or near intertidal coral reefs, response workers should not walk on the reef at low tide (NOAA 2010b).

There are few case studies of median-size oil spills on coral reefs, and, in some cases, oil exposure may have been limited. Physical factors at the time of the spill, notably wave action over the reef, can play a pivotal role in whether oil reaches reef organisms. When coral reefs are exposed to spills, the effects can be severe and last for years. Furthermore, coral reefs are experiencing severe effects of multiple stressors worldwide, including bleaching from ocean warming, ocean acidification, overfishing, coral diseases, and runoff and sedimentation from terrestrial sources. Additional stresses due to oil exposure and response impacts may compound their on-going decline.

6.3.2 Impacts of Oil Exposure and Response Actions on Subtidal Rocky Reefs

One spill included in this analysis had field-based studies of spill impacts to subtidal rocky reefs; in this case the rocky reefs were dominated by invertebrates. Studies of kelp and associated communities are discussed in Chapter 10 – Kelp. The 2001 grounding of the oil tanker *Jessica* off the Galapagos Islands, Ecuador released 2,800 bbl diesel and 2,160 bbl bunker fuel. Variations in percent cover, abundance, and species richness of animals and plants in the shallow subtidal (<2 m depth) one month after the spill were within the range of natural variation previously observed at the study sites; therefore, no impacts were attributed to oiling (Edgar et al. 2003a; Lougheed et al. 2002). Physical characteristics, including the presence of exposed shorelines in the area, moderate wave action, high sunlight, and warm temperatures, were hypothesized to have minimized impacts (Edgar et al. 2003a; Lougheed et al. 2002). At 4–11 m depth, impacts to subtidal macroinvertebrate and plant communities were attributed to both physical damage to the reef from the grounding of the wreck and effects of oil on the biota. Four months post spill, increases in cover of opportunistic algae and hydroids and decreases in densities of sea urchins were observed. These impacts occurred within 100 m of the wreck site (Marshall and Edgar 2003).

6.3.3 Impacts of Oil Exposure and Response Actions on Oyster Reefs

The only median-range oil spill with documented impacts to oyster reefs is the *Apex* barge spill, where 16,700 bbl of partially refined crude oil were released. PAH concentrations in oysters at two sites sharply increased 6 days after the spill relative to pre-spill values but were not different from pre-spill values by 37 days post spill. However, analysis of alkylated aromatic hydrocarbons indicated that oysters were still contaminated with *Apex* oil 37 days after the spill at one site and 110 days after the spill at another site. Presence of alkylated compounds that may have originated from the *Apex* barge spill were still present more than 1 year after the spill, indicating the possible penetration of oil into sediments that could be released by storms or other disturbances (Wade et al. 1993). All available data consisted of exposure information; studies of impacts to oyster survival, condition, growth, or reproduction were not conducted.

6.4 Impacts of Oil Exposure and Response Actions on Subtidal Soft-Sediment Communities

Though many field studies have examined the impacts of median-range oil spills on subtidal soft-sediment communities, relatively few have quantified or estimated time to recovery. Studies of impacts on these communities focus on the responses of meiofauna and macrofauna invertebrates, although one spill (Texaco March Point Refinery) and one experiment (BIOS) had studies of impacts on macroalgae. From a thorough review of the literature on impacts from oil spills, seven oil spills and one experiment that mimicked an oil spill were identified as having data from field-based studies of crude oil spills of 500–20,000 bbl. Eight spills have data from diesel, home heating oil, or No. 2 fuel oil spills, and one spill has data from a median-range condensate spill (**Table 6-2** and **Figure 6-4**). Of the seventeen spills discussed in this section, four spills had data indicating oil exposure and/or duration of exposure but did not have studies that quantified organismal or ecological impacts of exposure.

Table 6-2. Studies with documented or estimated impacts to and recovery of subtidal soft-sediment communities from spills of crude oil or diesel-like oils (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Studied Species	Documented Effect/Impacts	Recovery (years)
1971 Texaco March Point Refinery, Guemes Island, WA[a]	4,700 bbl diesel oil	Subtidal fauna and flora	None – no observable mortality 3 days post spill.	No impact measured
1977 T/B *Bouchard 65*, Buzzards Bay, MA[b]	2,000 bbl No. 2 fuel oil	Macrofauna	None during the 5 months after the spill.	No impact measured
1984 Uniacke G-72, off Sable Island, Nova Scotia[c]	1,500 bbl gas condensate	Crabs	None – no contamination 10 days after blowout.	No impact measured
1991 T/V *Agip Abruzzo*, Ligurian Sea, Italy[d]	13,500 bbl Iranian Light crude oil and an unknown amount of IFO 380	Meiofauna	Immediate reduced densities of nematodes, turbellarians, and foraminiferans (forams); recovery by 2 weeks post spill. Impacts were not observed on copepod populations, but most were epibenthic and could escape oil contamination.	0.04
1981 T/V *Sefir*, Baltic Sea[e]	2,800 bbl No. 1 fuel oil	Bivalves (mussels and clams)	Increases in aliphatic hydrocarbons to about 50 times background level 1.5 months post spill.	>0.1
1989 T/V *World Prodigy*, Narragansett Bay, RI[f]	6,900 bbl home heating oil	Crustacean macrofauna (amphipods, isopods, ostracods, crabs >0.5 mm)	Mortality of amphipods and crabs observed at one site 6 days post spill. Decreases of dominant crustacean species of 69–95% at impacted sites during first 5 weeks post spill.	>0.1
1999 T/V *Estrella Pampeana*, Ria de la Plata, Argentina[g]	15,700 bbl crude oil	Clams	Coastal sites: Hydrocarbon concentrations in clams 6 months post spill were 4–9 times higher than baseline levels. Offshore sites: No hydrocarbon enrichment.	Coastal sites: >0.5 Offshore sites: Not impacted
2011 Bohai Sea, China[h]	3,200 bbl crude oil and drilling mud	Meiofauna (Foraminifera), Macrofauna (>0.5 mm)	Species composition of forams was impacted by the spill, with large forams more sensitive to oil pollution than small forams, though small forams also had high mortality. Abnormally formed forams were more common at polluted sites. Macrofauna abundance decreased from the time of the spill to 3.25 years post spill, and had increased at 5 years post spill. Biomass remained lower than pre-spill levels at 5 years post spill. Diversity was comparable for pre- and post-spill comparisons.	Forams: >0.5 Macro-fauna: >5

Oil Spill	Oil Volume and Type	Studied Species	Documented Effect/Impacts	Recovery (years)
2000 T/V *Eurobulker*, Aegean Sea, Greece[i]	4,850 bbl crude oil	Clams, other macrofauna	Offshore sites: Initial decreases of some sensitive taxa (such as echinoderms and some crustaceans), increases in opportunistic species. Recovery by 4 months post spill. Coastal sites: Delayed impacts starting 6 months post spill, decrease in species richness and community diversity through 8 months post spill.	Offshore sites: 0.3 Coastal sites: >0.7
1987 T/V *Antonio Gramsci*, Baltic Sea[j]	4,200 bbl crude oil	Macrofauna (>0.5 mm)	Contaminated amphipods were found in samples from 1987 and 1988. Contaminated isopod, gastropod, and bivalve samples were found at 1 year post spill. Amphipod and ostracod densities decreased at 1 year post spill at deep (15–23 m) sites; no changes at shallow (4–5 m) sites.	>1
1989 T/V *Bahia Paraiso*, Arthur Harbor, Antarctica[k]	3,760 bbl diesel fuel arctic	Clams	High PAH concentrations in clams near the spill. Little or no PAH contamination of subtidal sediments 2 years post spill.	<2
1981 Baffin Island Oil Spill (experiment), Canadian Arctic[l]	280 bbl sweet medium crude oil (Venezuelan Lagomedi) experi-mental field oiling: 1) oil only and 2) oil + dispersant	Infauna (clams, polychaetes), epifauna (echinoderms, crustaceans), macroalgae	Infauna: No short-term effects of untreated oil. No large-scale mortality, no changes in community structure. Effects on physical condition of infauna 2 years post spill. Epifauna: Possibly delayed effects of oil on reproduction of amphipods. Minor impacts only. Macroalgae: Impacts were not observed. Bivalve physiology: Invasive neoplasias (probably cancer) in clams from oil only treatment 1 year post spill. Biochemical analyses revealed no severe stress in clams.	Infauna: >2 Epifauna: >2 Macroalgae: Not impacted Bivalve physiology: >1
1996 T/B *North Cape*, South Kingstown, RI[m]	19,700 bbl, home heating oil	Lobsters, surf clams, crabs, amphipods, other invertebrates	Estimates of numbers of animals killed in offshore region of impact area: Lobsters: 9 million Surf clams: 19.4 million Worms and amphipods: 4.9 billion Rock and hermit crabs: 7.6 million Mussels 20.2 million Additional mortality of sea stars, other crabs, sea urchins, snails, razor clams, and sea cucumbers observed. Lobster abundances 1–3 months post spill were measurably different in impacted and control sites.	0.4–5 (estimated)

Oil Spill	Oil Volume and Type	Studied Species	Documented Effect/Impacts	Recovery (years)
1969 T/B *Florida*, West Falmouth, MA[n]	4,385 bbl No. 2 fuel oil	Macrofauna	Immediate mortality of ampeliscid amphipods, crabs, other crustaceans, bivalves, and gastropods. Oysters, clams, and scallops contaminated with hydrocarbons 8 months post spill. Number of offshore benthic species lower at impacted sites than at control sites 4 and 5 years post spill; population densities similar at impacted and control sites. Numbers of opportunistic species and individuals at offshore sites decreased from years 4 to 5. Densities and species composition fluctuate widely at impacted sites but are stable at control sites, driven by population explosion and subsequent crash of opportunistic species, including polychaetes.	Nearshore: 2.5 to >5 Offshore: 1
1974 T/B *Bouchard 65*, Buzzards Bay, MA[o]	600 bbl No. 2 fuel oil	Macrofauna (bivalves, gastropods, crabs)	At 1 week post spill, mortality of benthic macrofauna down to depths of 2.5 m; Impacts were not observed on benthic macrofauna at depths of 5–8 m.	0–2.5 m depth: Not estimated 5–8 m depth: Not impacted
1985 T/V *ARCO Anchorage*, Port Angeles, WA[p]	5,690 bbl Alaska North Slope crude oil	Subtidal invertebrates (including nematodes, polychaetes, bivalves, crustaceans)	Nematodes numerically dominated initial samples (8 months post spill); subsequent sampling through 2 years post spill showed increased numbers and biomass of polychaetes, bivalves, and crustaceans. No oil contamination in bivalves.	Not estimated
1998 Equinox well blowout, Lake Grande Ecaille, LA[q]	1,535 bbl South Louisiana crude oil	Injury to benthic organisms estimated from impacted sediments	Approximately 21 acres of subtidal sediments were adversely affected by the deposition of oily sand discharged during the incident, with the most severe impacts occurring over 1.6 acres immediately surrounding the wellhead. The deposited sand was removed by vacuuming it off the sediment surface by divers who finished this task approximately ten weeks following the beginning of the incident. Injury to benthic organisms from the oily sand is estimated to have resulted in the loss of 6.1 service-acre years of benthic services.	Not estimated

Source: [a]Woodin et al. (1972); [b]Schrier (1978); [c]Carter et al. (1985); [d]Danovaro et al. (1995); [e]Linden et al. (1983); [f]Pilson (1990); Widbom and Oviatt (1994); [g]Colombo et al. (2005); [h]Lei et al. (2015); Wang et al. (2020); [i]Zenetos et al. (2004); [j]Hirvi (1990); [k]Kennicutt and Sweet (1992); Kennicutt et al. (1991a); Kennicutt et al. (1991b); [l] Cross and Thomson (1987); Cross et al. (1987a); Cross et al. (1987b); Neff et al. (1987); [m]Michel et al. (1997); NOAA (1999); [n]Blumer et al. (1970a); Michael et al. (1975); Sanders (1978); Sanders et al. (1980); [o]Hampson and Moul (1978); [p]Kittle Jr. et al. (1987); Mancini et al. (1989); [q]LOSC et al. (2005)

The response and recovery of subtidal soft-sediment communities impacted by an oil spill occurs in four main phases: (1) a period of considerable mortality in sensitive species due to the toxic effects of the oil; (2) a period of low species richness and abundance; (3) a period of organic enrichment in which opportunistic taxa increase in abundance; and (4) a period in which opportunists decrease in abundance concurrent with re-colonization of oil-sensitive species that suffered mortality initially. During the final step of this process, the community begins to return to pre-spill or undisturbed conditions (Dauvin 1998; 2000; Glémarec and Hussenot 1982; Gomez Gesteira and Dauvin 2005).

Sensitive species whose populations are reduced or eliminated during the initial stages of a spill generally include copepods (in the meiofauna) and amphipods (in the macrofauna). Taxa with species that are more tolerant of oil exposure generally include nematodes (meiofauna) and some types of polychaetes (macrofauna). These latter taxa may opportunistically increase in abundance following oil spills due to high tolerance and their ability to rapidly colonize substrates (Gomez Gesteira et al. 2003). The ratios of tolerant to sensitive taxa (nematode:copepod [N:C] and polychaete:amphipod [P:A]) are used as indicators of oil exposure and degree of effects in sedimentary environments; an increase in this ratio is

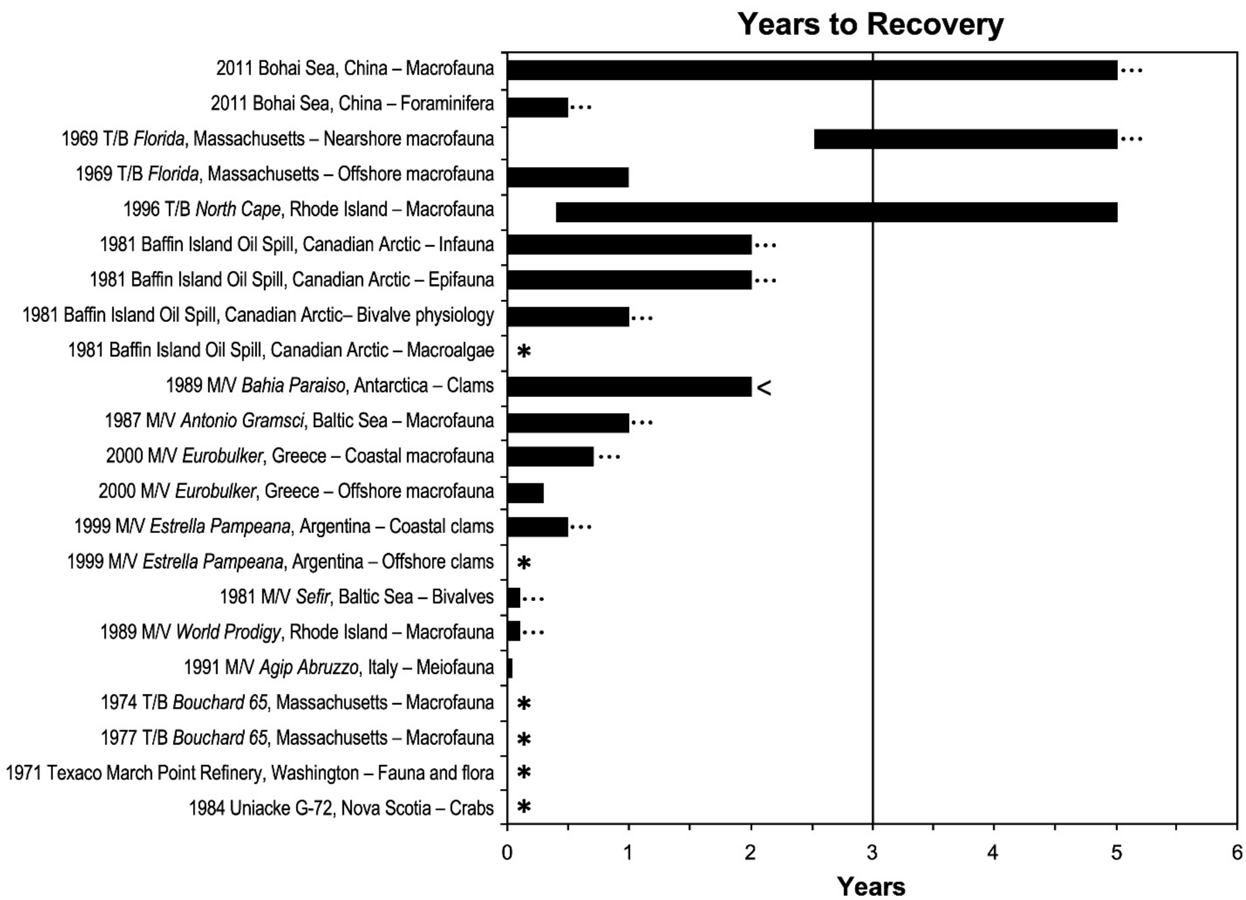

Figure 6-4. Recovery of subtidal soft-sediment communities for the oil spills listed in Table 6-2
Asterisk indicates impacts were not detected. Dotted lines indicate incomplete recovery at the time of the most recent study. < symbol indicates recovery was complete at some point before the time shown.

generally expected in communities that have been impacted by oil exposure (Baguley et al. 2015; Demopoulos et al. 2016; Gesteira and Dauvin 2000; Nikitik and Robinson 2003). These indices need to be carefully interpreted due to differences in sensitivity and tolerance among species in each taxonomic group and specific environmental and oiling conditions that can be unique to each spill.

Impacts to the meiofauna were investigated following the Bohai Sea and *Agip Abruzzo* spills. Six months after the Bohai Sea spill, low species richness and abundance of benthic foraminifera were found at sites impacted by the spill. Additionally, there was much greater incidence of formation abnormalities among foraminifera at impacted sites relative to control sites. Large foraminifera were more sensitive to oil than small foraminifera, and community structure was related to oil pollution (Lei et al. 2015). Meiofaunal responses to the *Agip Abruzzo* spill were counter to conventional understanding of oil pollution responses in this group of organisms. In this case, decreases in densities of nematodes, along with density decreases in turbellarians and foraminifera, were observed; whereas copepod densities were unaffected by the spill.

The copepod community at the study sites was dominated by epibenthic forms that could escape hydrocarbon pollution. Because of these results, Danovaro et al. (1995) suggested that N:C is strongly affected by sediment type, organism seasonality, and environmental conditions and is not an adequate tool for oil pollution monitoring. The meiofaunal responses to and recovery from the *Agip Abruzzo* spill were very rapid, with community effects occurring immediately after the spill, and recovery of community structure was complete after only 2 weeks (Danovaro et al. 1995).

Impacts to amphipod populations were observed following the *Antonio Gramsci*, *World Prodigy*, and *Florida* spills. Amphipod densities decreased after each of these spills, though amphipod impacts were only seen at deep (15–23 m) sites after the *Antonio Gramsci* spill (Hirvi 1990). Immediate mortality of ampeliscid amphipods occurred after the *Florida* spill, with more severe impacts at nearshore sites than offshore sites. The decline of amphipods and other sensitive taxa made resources available to other, opportunistic taxa, such as polychaetes. Polychaetes monopolized the substrate immediately after the *Florida* spill for the first 7–11 months, then their populations crashed. The macrofaunal community then underwent successional changes in community composition over the next several years as it recovered. Recovery of macrofaunal density, species richness, and diversity were complete after roughly a year at offshore sites; whereas, recovery at nearshore sites was only beginning after 2.5–5 years (Sanders et al. 1980). In addition to the *Florida* spill, increases in polychaetes were also observed after the *ARCO Anchorage* spill. Other studies examined changes to the macrofaunal community in general. Changes in macrofaunal community structure, abundance, biomass, diversity and/or mortality were observed after the Bohai Sea, 1974 *Bouchard 65*, *Eurobulker*, and *North Cape* spills (**Table 6-2**). During the *North Cape* spill, release of oil directly into the surf zone during high winds and heavy wave activity resulted in the dispersion of oil throughout the water column and into bottom sediments. These conditions are thought to have contributed to the observed mass mortalities of benthic macrofauna (NOAA 1999).

Not all median-range oil spills had measurable impacts to subtidal soft-sediment communities. In some cases, impacts were not detected, or community variation could not entirely be attributed to the spill. Studies following the Texaco March Point Refinery, 1977 *Bouchard 65*, and Uniacke G-72 spills impacts were not detected, and studies following other spills revealed impacts at some locations but not others. The 1974 *Bouchard 65* spill resulted in mortality to macrofauna down to 2.5 m depth, but impacts were not observed at 5–8 m depths. The *Estrella Pampeana* spill resulted in high hydrocarbon concentrations in clams from nearshore sites, but there was no hydrocarbon enrichment in clams from offshore sites. Environmental conditions at the time of spill often influence whether impacts occur to subtidal soft-sediment communities. Absence of impacts of the 1977 *Bouchard 65* spill was likely due to ice preventing oil from reaching shore and the slow release of oil trapped in ice. Additionally, low metabolic rates of organisms during cold winter temperatures may have reduced uptake of hydrocarbons (Schrier 1978). During the *Estrella Pampeana* spill, strong winds pushed oil toward shore and prevented oil from reaching and impacting offshore communities (Colombo et al. 2005). In the Bohai Sea spill, the presence of other pollution at the platform unrelated to the spill was hypothesized to have impacts on the subtidal soft-sediment community, and changes could not solely be attributed to the spill (Wang et al. 2020).

6.5 Summary and Information Needs for Assessing Impacts to Marine Benthic Communities

There is a wide range of communities in the benthic environment, and the level of understanding of impacts and recovery from median-range oil spills varies depending on community type. Spill impacts and recovery are generally better understood for coral reefs and subtidal soft-sediment communities, with evidence of impacts and recovery from several spills, than for subtidal rocky reefs and oyster reefs. No studies were identified that documented the impacts to mesophotic and deep marine benthic communities from median-size spills.

Impacts and recovery are also variable among case studies within individual community types. After some spills, impacts to coral reefs and subtidal soft-sediment communities were pronounced; after others, impacts were minimal or not detectable. This variability can be driven by the following factors: extent and degree of oiling of the benthos; persistence of spilled oil in sediments and on benthos; oil type and toxicity; species-specific responses to oil exposure, including the degree of sensitivity or tolerance to oil pollutants and organic enrichment; species life history traits, including growth rates, longevity, feeding strategies, reproductive strategies, and recruitment; and physical environment (e.g., sea state) at the time of the spill. Time to recovery can range from weeks (e.g., in some lightly impacted subtidal soft-sediment communities) to years (e.g., in heavily impacted coral reefs). In general, habitats with heavy oiling recover more slowly.

Studies of impacts on benthic communities are often limited by the difficulty in attributing ecological changes to a spill. Natural variation in species populations and/or presence of other stressors unrelated to an oil spill often complicate interpretation of results and preclude the determination that impacts are related to oil exposure. Collection of baseline (pre-spill) data and inclusion of control sites in impact studies, where possible, would help to address this issue.

Recovery is often difficult to quantify because studies of marine benthic communities rarely continue until full ecological recovery has been reached. In many of the cases discussed in this chapter, studies were only conducted during the acute phases of spills (weeks or months following initial oil exposure), so longer-term impacts are unknown. Furthermore, many studies only surveyed the benthic communities once after a spill, so it is not possible to track changes through time or to determine time to recovery. To better understand recovery following median-range oil spills, monitoring of impacted communities should continue until baseline conditions are reached or an alternate stable state has become established.

7 Salt Marshes and Mangroves

7.1 Habitat Description, Communities, and Ecological Functions and Services

Coastal salt marshes are periodically to continuously inundated by salt water from tides and are characterized by emergent herbaceous vegetation (e.g., grasses, sedges, and rushes) tolerant of both salinity and saturated soil conditions. Mangroves are also inundated by tidal waters and adapted to salinity and flooded soils, but are characterized by woody trees and shrubs rather than herbaceous plants. The structure and form of salt marsh and mangrove communities are a function of wave exposure, tidal range, salinity, soil type, plant species, and faunal communities.

Salt marshes, mangroves, and adjacent habitats provide a broad range of ecosystem services including: nursery grounds for commercially and recreationally valued species; bird nesting and foraging habitat; habitat for invertebrates such as crabs, snails, bivalves, and marine worms; habitat and trophic support for fish; sediment/soil storage and transport; pollutant capture and filtering; nutrient mineralization and recycling; fisheries production; moderation of coastal inundation and erosion; carbon sequestration; and human recreation (Beland et al. 2017). Monetary values of these ecosystem services are an estimated $8,236/hectare [ha]/year in reduced hurricane damages alone for salt marshes and mangroves and range from $16,178/ha (marshes) to $2,468/ha (mangroves) for fisheries maintenance (Barbier et al. 2011).

Salt marshes and mangroves occur along low-energy coastlines where they are sheltered from high wave energy that characterizes other coastal habitats such as beaches. Salt marshes are distributed throughout the coastal areas of the U.S. on east, Gulf, and west coasts, from New England to Texas and from California to Alaska's Arctic coast (**Figure 7-1**). Mangroves in the U.S. are limited to warmer coastal tropical and subtropical zones along the Atlantic coast of Florida, the Gulf coast of the U.S., Puerto Rico, the U.S. Virgin Islands, Hawaii, and the U.S. Pacific Trust Territories (**Figure 7-2**). Nearly all (97%) of U.S. salt marshes occur on the Atlantic and Gulf coasts. Of these, 58% are along the Gulf of Mexico coast (Michel and Rutherford 2013). Georgia and South Carolina account for 33% of the salt marshes on the U.S. east coast (Wiegert and Freeman 1990). For example, the Mississippi River delta marshes make up 40% of the coastal marshes of the U.S. (Mitsch and Gosselink 2015). The Pacific coast (excluding Alaska) has only 3% of the nation's salt marshes, mostly (75%) in California (Michel and Rutherford 2013). Alaska includes approximately 139,617 ha of salt marsh, which is <1% of salt marshes in the U.S.

Each of these coastal regions is also characterized by differences in wave energy, tidal frequency and amplitude, and local topography; regional differences in salt marsh vegetation are also apparent. For example, along the northeast Atlantic coast, salt marshes form on peat overlying marine sediments with little upland sediment input. These marshes are dominated by *Spartina alterniflora* (smooth or saltmarsh cordgrass), *Juncus roemerianus* (black needlerush), *Spartina patens* (salt meadow cordgrass), *Distichlis spicata* (salt grass), *Salicornia* spp. (glasswort), and *Juncus gerardii* (salt meadow rush). Along the mid-Atlantic and south-Atlantic coasts, extensive marshes are formed in estuaries and behind barrier islands. *S. alterniflora* remains dominant, but *J. roemerianus*, *Batis maritima* (saltwort), *Monanthochloe littoralis* (shore grass), *Sesuvium portulacastrum* (sea purslane), *Spartina spartinae* (gulf cordgrass), and *Typha domingensis* (southern cattail) become increasingly common associates (USGS 2015). From the mid-Atlantic to the southern U.S. and along the Gulf of Mexico, freshwater flows from rivers become more important to the structure and function of the salt marshes.

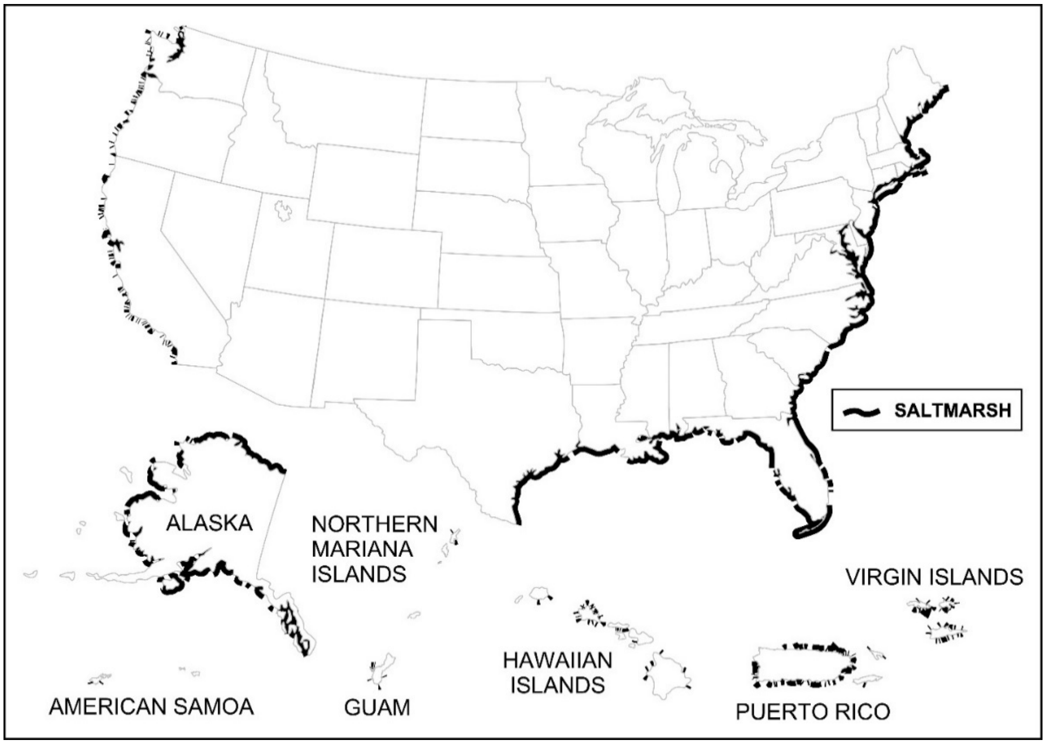

Figure 7-1. Distribution of salt marshes in the U.S. and U.S. Territories
Modified after Saintilan et al. (2009).

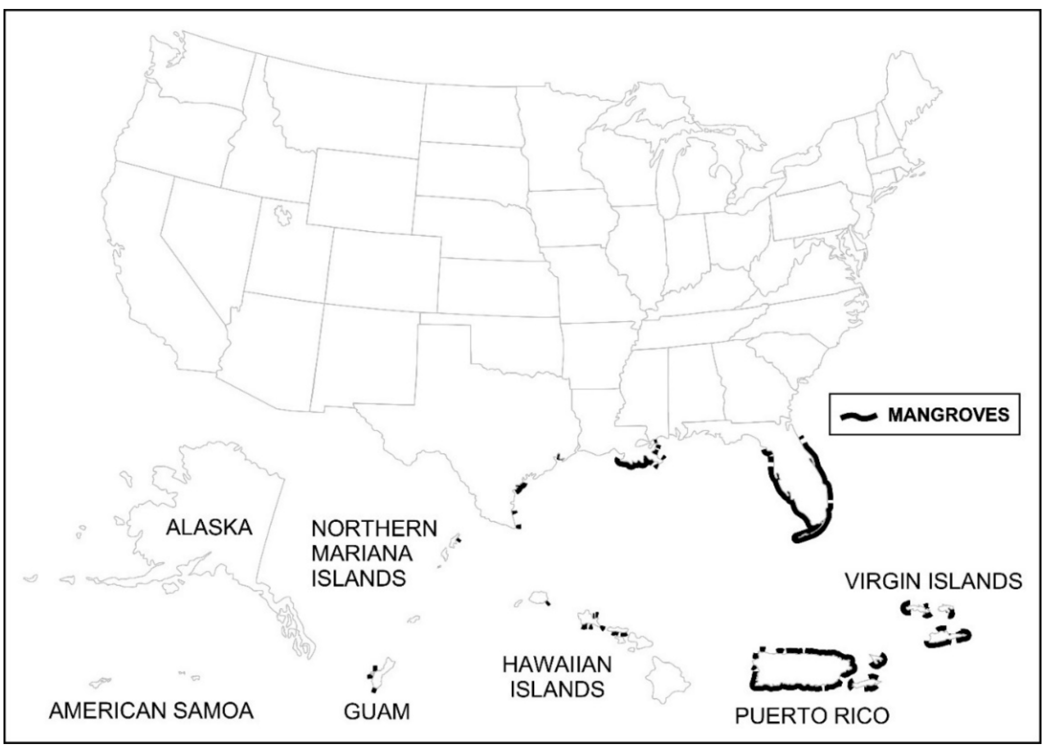

Figure 7-2. Distribution of mangroves in the U.S. and U.S. Territories
Modified after Saintilan et al. (2009).

Along the north Florida, Alabama, and Mississippi coast of the Gulf of Mexico, salt marshes are dominated by *J. roemerianus;* subdominant species include *S. alterniflora* and *S. patens*. Along the Gulf coast, *J. roemerianus* accounts for large percentages of the saltmarsh in Florida (60%), Alabama (52%), and Mississippi (92%). In Louisiana and Texas, *J. roemerianus* accounts for less than 4% of the total marsh area (Eleuterius 1970) and *S. alterniflora* is the dominant species; *Spartina patens,* and *J. roemerianus,* and *D. spicata* are subdominant species.

Pacific coast salt marshes are less developed than on the east and Gulf coast due to leading-edge shorelines with cliffs and few wide, flat river deltas and estuaries, although large marshes occur in sheltered areas such as San Francisco Bay. Vegetation may include fringes of *S. foliosa* and *D. spicata* bordered by stands of *Salicornia virginica* (pickleweed), which are the dominant species (based on area of coverage) along both the north and south coast of California (Zedler et al. 2008).

Subarctic and Arctic marshes, such as those along the coast of southern Alaska, are governed by processes and environmental characteristics similar to those of the temperate Pacific coast marshes but conditions can be more extreme due to scouring from ice rafting and thermal degradation of coastal permafrost (where present). Vegetation consists of sparse to close growth of halophytic, clonal graminoids such as *Puccinellia phryganodes* (creeping alkali grass), *Carex subspathacea* (Hoppner's sedge), and *Dupontia fisheri* (tundra grass), and the succulent forbs *Stellaria humifusa* (saltmarsh starwort) and *Cochlearia groenlandica* (scurvy grass). Arctic marshes support few plant species due to the severe climate and are often dominated by near-monotypic stands of *Carex lyngbyei* (Lyngbye's sedge) on mud flats and *Eleocharis palustri* (creeping spike-rush) or *Puccinellia* spp. (alkali grass) in the frequently inundated lower salt marshes (USGS 2016). The salt marsh zone along the Beaufort Sea is very narrow and patchy due to low tidal amplitude.

Plant species composition reflects salinity and elevation gradients in a salt marsh (**Figure 7-3**). For instance, fringing coastal salt marshes typically have higher salinities and lower species diversity than salt marshes with greater riverine influence. Hypersaline salt flats may occur in the high marsh where salt accumulates in evaporative pools and only blue-green algae and salt-tolerant plants survive. Increases in elevation from fringing to interior salt marsh typically form low (seaward) and high (landward) marshes, with greater species diversity in the high marshes due to inundation and salinity differences. Farther inland, salt marshes transition to lower salinity brackish and tidal freshwater marshes when freshwater input is greater (these lower salinity marshes share some species and similarities with salt marshes but are not directly covered here). Salt marsh plants are subjected to prolonged inundation and subsequent anoxia and accumulation of hydrogen sulfide that can inhibit plant growth. Adaptations to inundation include:

- Aerenchyma tissues that supply oxygen to roots;
- Thick cuticles and stomata that reduce water loss;
- Anaerobic metabolism that allows respiration in the absence of oxygen;
- Oxygenated root zone to reduce compounds harmful to plants; and
- C4 photosynthesis that increases water use efficiency.

Salt marsh plants have a number of additional adaptations that allow them to thrive under high salinities (Michel and Rutherford 2013). Some of these adaptations include:

- Salt secretion glands (eliminate excess salt), succulent stems and leaves (increase water retention that maintains internal salt balance), and waxy leaf coatings (minimizes contact with sea water);
- Salt exclusion (reduces salt uptake by roots); and
- Salt stress avoidance (by occupying higher levels of salt marsh) and periodic shedding of salt-saturated organs.

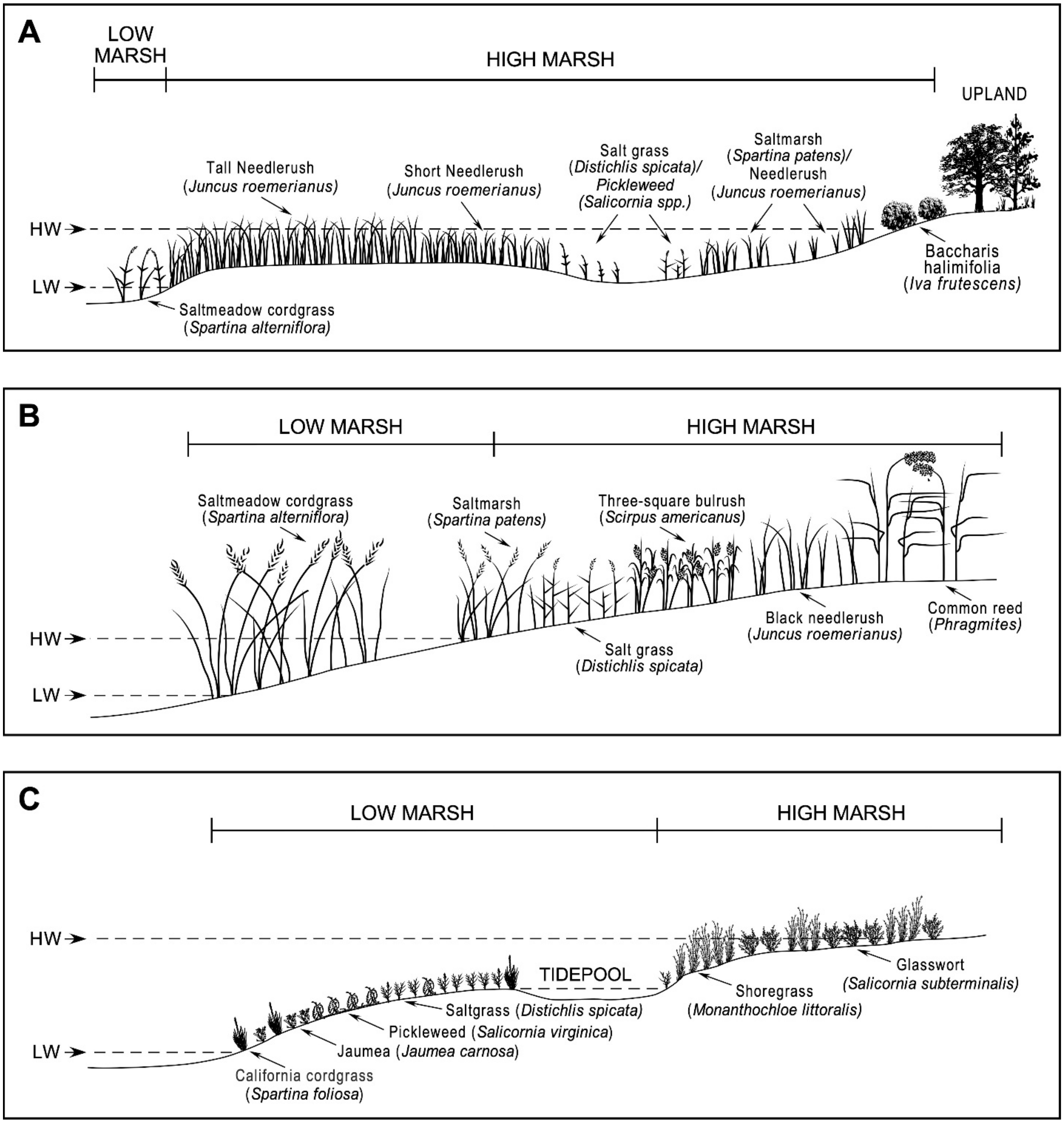

Figure 7-3. Tidal salt marsh plant zonation
A: Northeastern Gulf of Mexico salt marsh profile from Stout (1984); B: Mid-Atlantic salt marsh profile from Tiner and Burke (1995). C: Southern California salt marsh profile from Zedler et al. (2008).

Mangroves in the U.S. are dominated by four mangroves species or associates: *Rhizophora mangle* (red mangrove), *Avicennia germinans* (black mangrove), *Laguncularia racemosa* (white mangrove), and *Conocarpus erectus* (buttonwood). When these species occur together, they are typically distributed along an elevation gradient that corresponds to tidal regime (**Figure 7-4**). *R. mangle* and *A. germinans* are the two most common species in the U.S. and, like many mangroves, have conspicuous aboveground root structures (**Figure 7-5**).

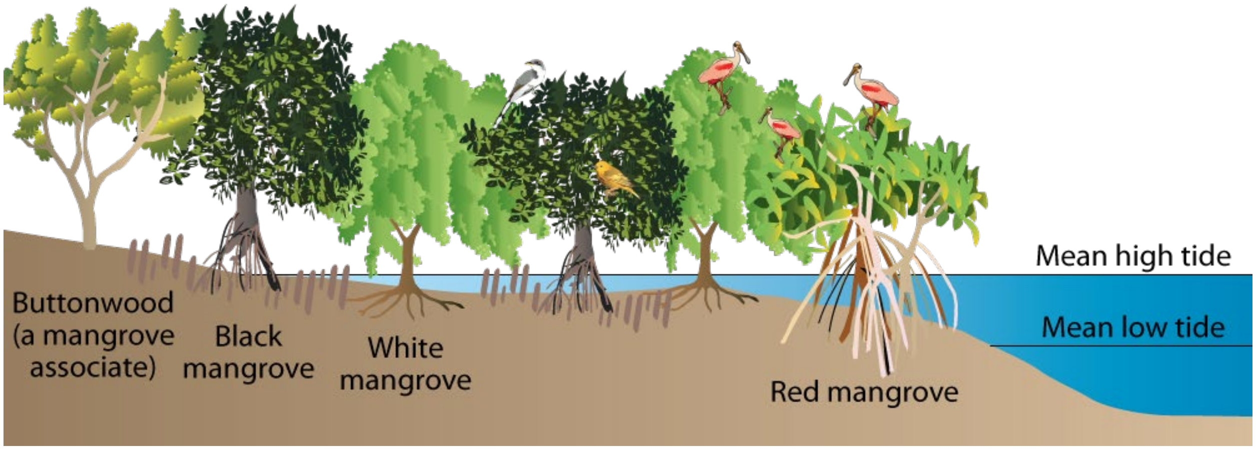

Figure 7-4. Mangrove distribution along a gradient of tidal inundation
Rhizophora mangle (red), *Avicennia germinans* (black), and *Laguncularia racemosa* (white), and the mangrove associate *Conocarpus erectus* may occur together, distributed along a gradient of tidal inundation. From Kruczynski and Fletcher (2012).

Figure 7-5. Dominant mangrove species in the U.S.
A. *Rhizophora mangle* (red mangrove). B. *Avicennia germinans* (black mangrove). C. *Laguncularia racemosa* (white mangrove). D. *Conocarpus erectus* (buttonwood mangrove). Photographs: A, B=RPI; C, D=R. Lewis.

A. germinans, which is more freeze tolerant than *R. mangle*, occurs in northern Florida, Texas, Louisiana, and Mississippi (McMillan and Sherrod 1986). Within mangrove forests, species zonation reflects tidal sorting of propagules based on buoyancy, size, differential survivability (Wang et al. 2019), and plant-animal interactions, such as differential predation of propagules by crabs (McKee 1995; Sousa and Mitchell 1999).

Mangroves have expanded north in the U.S. when compared with historic distributions, primarily due to reduced frequency and severity of hard freezes along the coast (Osland et al. 2013). Along the Texas coast, *A. germinans* is relatively common and is expanding northward. *R. mangle* is more cold-sensitive than *A. germinans* but has become established in bays in south Texas (Montagna et al. 2011).

Mangroves, although dominated by woody rather than herbaceous plants, are similar to salt marshes in that they, too, are adapted to high salinities, tidal inundation, and low soil oxygen. Adaptions of mangroves that allow them to tolerate high salinities and anoxic conditions are (after Hoff et al. 2014):

- Shallow root systems, generally in the upper 70 cm of the sediments, that avoid deeper (anoxic) soils and increase oxygen availability;
- Numerous lenticels and extensive aerenchyma tissue that facilitate oxygen uptake;
- Modified aboveground root structures such as pneumatophores and prop roots that allow direct gas exchange between the atmosphere and the root system and also provide physical support to trees in soft sediments;
- Salt glands that accumulate and then secrete salt, forming crystals on leaf surfaces that are then washed or blown away, or otherwise fall to the ground with the leaves; and
- Salt exclusion that reduces salt uptake by roots.

Mangrove forests have distinct types based primarily on hydrology and geomorphology that reflect differences in inundation regimes (Lugo and Snedaker 1974), described below.

- Fringe mangroves are tidally dominated, subjected directly to the full effect of tidal changes, and often have prop roots, buttresses, and pneumatophores;
- Riverine mangroves are often flooded by river flows, are the most productive mangrove forest type, and have moderate salinities due to freshwater flows into the forest;
- Basin mangroves often occur landward of berms, are the least productive of the three forest types, and are influenced more by rainfall than tide except at extreme high-tide conditions;
- Overwash forests are mangrove islands frequently inundated or washed over by tides; and
- Dwarf or scrub mangrove forests grow in areas where hydrology is restricted, resulting in conditions of high evaporation, high salinity, or low nutrient status.

7.2 Oil Behavior and Persistence in Marshes and Mangroves

7.2.1 Factors that Affect Oil Persistence

Oil behavior and persistence in marsh and mangrove habitats are influenced by the regional and local differences in wave energy, tidal frequency and amplitude, and vegetation described previously. Factors affecting the behavior and persistence of oil in marshes and mangroves are summarized below (Duke 2016; Hoff et al. 2014; Michel and Rutherford 2013).

Oil type. Lighter oils are more volatile and form thin slicks or sheens, thus they are less likely to heavily contaminate salt marshes and mangroves for long periods, unless oil reaches the subsurface soils. Heavier oils can form thick oil layers on the soil surface, with longer persistence times compared to lighter oils.

Light oils can penetrate the top few centimeters of sediments via interstitial spaces. Heavier oils can penetrate soils, but usually only if the water table is well below the soil surface and there is secondary porosity, such as crab burrows, to provide a mechanism for penetration.

Vegetation. Floating or trapped oil may adhere to vegetation in patches, in narrow or wide bands along the high tide reach, and/or in multiple bands along different tide reaches or water levels. Heavy shoreline oiling, coupled with tides that flood and then expose all or most of the vegetation, can coat all the aboveground vegetation. Marsh plants with smaller, more numerous stems (such as *S. patens*, which can have 10 times the number of stems per meter when compared with a plant such as *S. alterniflora*), provide a much larger surface area to which oil can adhere. In tall stands of mangroves, the tree canopy would rarely, if ever, be entirely flooded or exposed to oil; however, root structures and lower stems and branches can be completely coated with oil.

Exposure to waves and currents. Though marshes and mangroves occur in low-energy environments, fringing marshes and mangroves are more exposed to waves and tidal currents than interior vegetation. Thus, the residence time of oil along the fringes decreases as the energy of waves and currents increase.

Soil type. Oil can penetrate more readily and deeply into sandy soils compared to muddy soils; in soils with crevices, animal burrows, and root cavities; and in unsaturated soils where low water levels make the soil more permeable to oil. Oil penetration, burial, and/or response activities that actively mix oil into the soil can increase persistence of oil in the soil; oil at the surface may remain until the next growing season or at most several years while oil that penetrates the soils can persist much longer. Oil adsorbs or adheres to organic more so than mineral soils, thus oil may persist longer in organic soils.

Climate. Oil weathers more slowly in colder climates, where many of the physical, chemical, and biological processes that facilitate oil degradation are absent or act more slowly. In a mesocosm study, Sharma and Schiewer (2016) found that biodegradation decreased at lower temperatures and salinities. Snow and ice can impede oil reaching wetlands in the Arctic and portions of the subarctic.

Response actions. Many response activities, such as manual removal, trenching, sediment reworking, tilling, and similar activities, may further mix oil into soils, thereby increasing oil persistence.

Oil reaches marshes and mangroves with tides and currents and is likely to be trapped by vegetation (**Figure 7-6**), where it may weather, be removed by waves and tides, or persist in sediments at or beneath the surface, for decades in extreme cases.

7.2.2 Oil Persistence in Salt Marshes

Light refined oil products such as No. 2 fuel oil, home heating oil, and diesel, are more likely to evaporate or disperse into the water column rather than become stranded along shorelines, but are more acutely toxic to marsh vegetation than heavier crude oils (Michel and Rutherford 2013). Spills of crude oil tend to be more persistent because they are more viscous and can adhere to marsh vegetation and sediments. Cleanup is sometimes undertaken for heavily oiled marshes to reduce impacts of physical smothering and long-term persistence of crude oils.

In marshes where the sediments were heavily oiled with No. 2 fuel oil and no cleanup was conducted, oil was found to persist in the marsh soils beyond the conclusion of the studies. For example, oil persisted for over 7 years after the Exxon Bayway Refinery spill (Bergen et al. 2000) and more than 30 years after the *Bouchard 65* (Peacock et al. 2007) and *Florida* (Reddy et al. 2002) spills, all of which occurred in temperate climates. In these cases, the oil stranded heavily in sheltered areas (bays or rivers) and, in the case of the *Bouchard 65* and *Florida* spills, marshes were re-oiled over multiple tidal cycles.

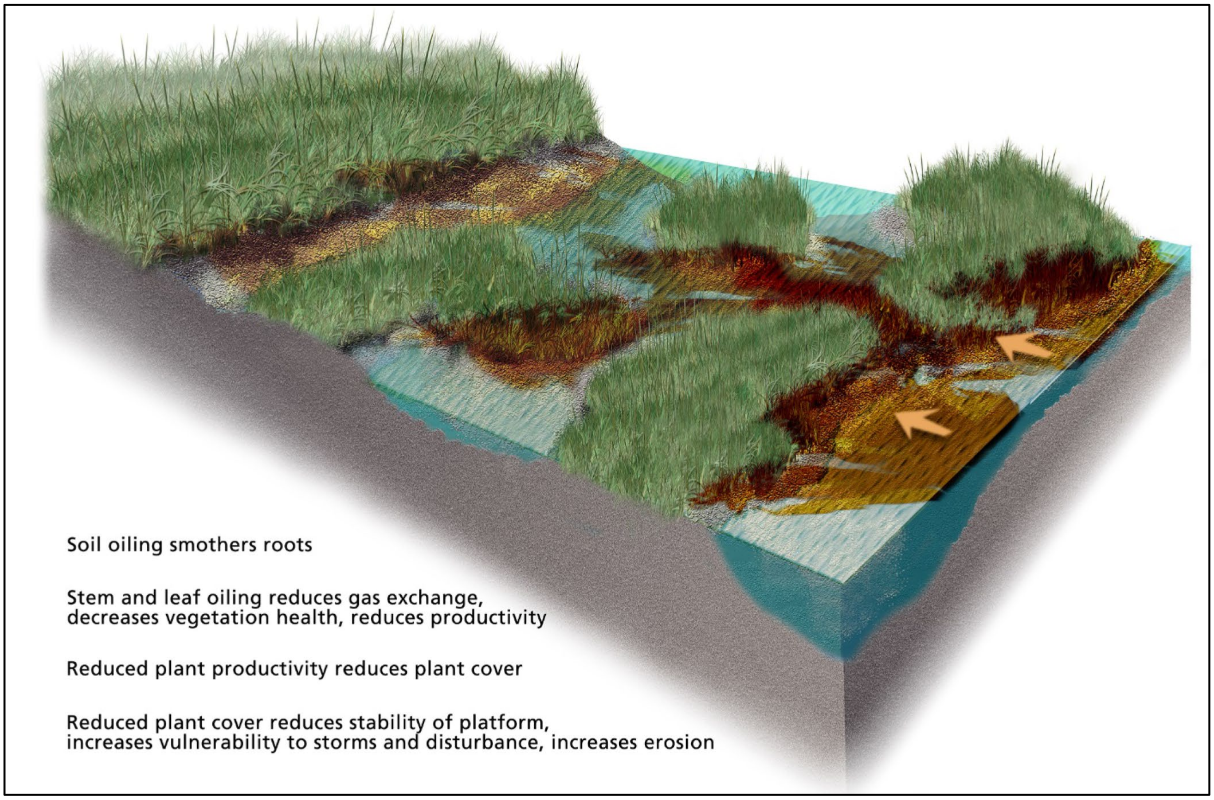

Figure 7-6. Pathways of oil into marshes (from NOAA 2016a)

In the Florida Keys, where air temperatures reached 90°F the week after the *Garbis* spill, crude oil covered the sediment surface, filled crab burrows, and was still present after a year (Chan 1977). Burning successfully removed 90-95% of the condensate remaining on the marsh surface a week after the Mosquito Bay pipeline spill but oil remained in the sediments and crab burrows after 13 months (Michel et al. 2003).

Light to moderate oil on marsh vegetation was generally gone 1–2 years following a spill in the cases studied. Only 20% of the spilled oil reached the shoreline after the *North Cape* spill of home heating oil; only a small fraction of that oil entered the salt ponds as surface slicks and oil sheens were absent 9 months post spill (Michel 1996). Marsh sediment cores take the first month after the spill had PAHs of 1.7 and 2.0 milligrams per kilogram (mg/kg), mostly attributed to background sources. In contrast, salt marsh vegetation did not recover where the soil concentrations of No. 2 fuel oil were >2,000 ppm at the *Florida* spill in Buzzards Bay (Burns and Teal 1979).

Response activities in which workers trampled sites resulted in deeper penetration of oil into sediments and longer persistence. Response activities that mixed oil into the soils (e.g., due to trampling) increased oil persistence after both the Fidalgo Bay (Hoff et al. 1993) and *Estrella Pampeana* (Moreno et al. 2004) spills. Ambient-water flushing from platform boards that reduced sediment disturbance appeared to successfully remove oil from vegetation after the Fidalgo Bay spill. Flushing was less successful when sunny weather increased the tendency of the oil to stick to vegetation after the *Julie N* spill; however, much of the oil had weathered, and the vegetation was covered by thin sediment coating after a large storm a month later (Michel et al. 1998).

7.2.3 Oil Persistence in Mangroves

Spilled oil carried into mangrove forests by winds and tides may be trapped and accumulated in low-energy depositional areas where oil adheres to surface sediments, prop roots, propagules, stems, and other structures when the tide recedes. Just as in salt marshes, lighter oils tend to spread more deeply into mangrove forests than heavier oils, and penetration into sediments results in greater persistence and toxicity. Heavy oils and emulsified oils can be trapped in thickets of *R. mangle* prop roots and *A. germinans* pneumatophores and are likely to adhere to these and other organic surfaces (e.g., wrack). Rising and falling water levels with tides can result in re-oiling, and rapid oil burial can lead to long-term persistence, but tidal action and precipitation can also flush oil from mangroves. Because mangroves are tropical and subtropical, oil degradation rates can be higher due to the higher temperatures in these environments, although rapid oil burial can lead to long-term persistence.

Oil that reaches interior mangroves may persist longer than oil in marshes due to more sheltered conditions afforded by the trees. Mangrove prop roots, propagules, and sediments were thickly coated with oil that penetrated sediments via fiddler crab burrows in the Florida Keys after the *Garbis* spill (Chan 1977). In addition, oily debris was temporarily retained in mangrove areas after the spill, resulting in re-oiling of mangroves and adjacent intertidal habitat for at least a month after the spill. Oiling persisted in and on the surface of the mangrove sediments for at least a year.

7.3 Impacts of Oil Exposure and Treatment on Salt Marshes

Salt marshes are vulnerable to the effects of oil spills, including oil that adheres to and coats or smothers plant surfaces and soils, direct oil toxicity (especially from lighter oil types and oil that is less weathered), and cleanup methods (Michel and Rutherford 2013). The ecological functions and services lost due to impacts on salt marshes from oil spills are a function of: 1) the amount and type of oil trapped in the marsh; 2) the amount (and area) of vegetation oiled; 3) the plant species impacted; 4) how much above- and below-ground vegetation is impacted; 5) the degree that marsh soils are oiled; 6) how quickly the oil degrades or is removed from the marsh (by natural processes or response operations); 7) the time of year of the spill; and 8) the intensity of response operations.

Sediment oiling in salt marshes can occur due to oil penetration via crab burrows and dead shoot/root cavities, oil burial, and/or soil mixing during response operations, which further slows oil degradation and exposes belowground vegetation (e.g., rhizomes) to oil (Levine et al. 2017; Zengel et al. 2015; Zengel and Michel 2013). This exposure can lead to longer-term damage and greater mortality (Hester and Mendelssohn 2000; Mendelssohn et al. 2012; Michel et al. 2009; Pezeshki and DeLaune 2015). Spills in marshes are typically not cleaned up unless there is heavy vegetation and sediment oiling because aggressive cleanup is likely to adversely impact recovery, delay it, or preclude it entirely.

A thorough review of the literature on impacts to salt marshes from oil spills identified ten spills with data adequate for inclusion in this analysis (**Table 7-1, Figure 7-7**). Among the spills evaluated for this synthesis, heavy oiling of vegetation and sediments resulted in severe and in some cases permanent impacts to temperate salt marshes. Oiling ranged from light to heavy overall but also often included small areas where oil pooled, or larger areas of heavy oiling due to high tides; pooled and heavy oiling were followed by severe mortality and soil erosion, especially if exposed to wave energy. Impacts to salt marshes were reported for seven spills for which impacts and recovery were not documented. For example, reports of some spills included the areal extent of marsh impacted or oiled in an initial damage assessment, but no monitoring was undertaken to document impacts or recovery. These spills are also included in **Table 7-1** but are not further analyzed due to inadequate data on either extent or duration of impacts.

Table 7-1. Studies with documented or estimated impacts to and recovery of marshes from spills of crude oil, condensate, or diesel-like oils (500–20,000 bbl)

Oil Spill	Degree of Oiling (oil volume & type)	Shoreline Cleanup	Species	Documented Effect/ Impacts	Recovery (years)
1996 T/B *North Cape*, South Kingstown, RI[a]	Light (19,700 bbl home heating oil)	No	*S. alterniflora*	1.17 ha of fresh and low salinity salt marsh oiled, chlorosis observed about 10 days later; no differences in stem density, stem height, and biomass were found between post-spill and pre-spill *S. alterniflora* marsh plots after 6 months.	<1
1996 T/V *Julie N*, Portland, ME[b]	Heavy to Light (2,058 bbl home heating fuel and 2,219 bbl IFO 380	Yes: ambient-water flushing, cutting	*S. alterniflora, S. patens*	10.4 ha oiled: 4.9 ha heavily oiled; 0.2 ha (5%) showed die-off in isolated patches, attributed to No. 2 oil impacts. 1 year: stem heights/density same for oiled and unoiled controls except for the 96 patches of dead vegetation. Impacts were not observed from vegetation cutting due to boat use instead of foot traffic. No signs of invertebrate mortality but absence of macro-invertebrates in heavily impacted areas possibly due to tidal flux.	1
1988 T/V *El Omar* Milford Haven, Wales[c]	Heavy to Light (670 bbl Light Iranian crude)	Not in marshes	*Spartina sp.*	Marshes mostly self-cleaned by wave and tidal action within 9 months; no substrate oiling except post spill when weathered oil initially captured in vegetation and seed heads of *Spartina* was buried in sediments with litter fall; heavily oiled areas took longer to recover.	>1
1975 T/V *Garbis* Florida Keys, FL[d]	Heavy to Light (1,500–3,000 bbl crude emulsion)	Not in marshes	*Batis maritima, Salicornia sp., Sesuvium portulacastrum, Monanthochloe littoralis, Borrichia frutescens*	Plants killed by oil on stems, leaves, substrate; lightly oiled vegetation recovered after 6 months; unoiled vegetation was not impacted; seasonal recovery did not occur. No epifauna survived oiling in the marsh; extensive substrate oiling and high temperature killed fiddler crabs but returned after 6 months; snails died due to isolation from water; other species survived.	>1
2001 Mosquito Bay, LA[e] (Williams pipeline)	Heavy to Light (2,380–3,000 bbl condensate)	Yes: burning	*S. alterniflora, S. cynosuroides, D. spicata*	4.9 ha of marsh oiled; 40 ha burned. Vegetation killed by condensate was not recovered after 13 months; burn removed oil but did not mitigate toxicity impacts. Plant shoots appeared after 1 week in lightly oiled and unoiled areas burned and was considered recovered; Fiddler crabs present 6 months later, suggesting minimal residual impacts.	>1

Oil Spill	Degree of Oiling (oil volume & type)	Shoreline Cleanup	Species	Documented Effect/ Impacts	Recovery (years)
1991 Fidalgo Bay, WA[f]	Heavy to Light (714 bbl Alaska North Slope crude)	Yes: flushing, vacuum	*Salicornia virginica, Distichlis spicata*	Monitoring of treatment and control plots for 16 months found most severe impacts due to trampling, including deeper oil penetration; washing/vacuuming removed oil with least impacts. Belowground structures slowly died; aboveground vegetation relied on less belowground biomass at treated sites. Clay substrate and dormancy reduced impacts. Damaged belowground roots/rhizomes slowed recovery. After 2nd growing season, several heavily oiled areas remained unvegetated.	2 after light-moderate oiling, if not trampled; 3–4 if heavily oiled, washed, not trampled
1999 T/V *Estrella Pampeana*, Rio de la Plata, Argentina[g]	Heavy (15,700 bbl Patagonia light crude)	Yes; cutting	*Zizaniopsis bonariensis, Schoenoplectus californicus*	After 36 months, density and biomass declined in oiled plots. Areas with trapped oil recovered more slowly than areas without. Only cut (trampled) sites did not recover to vegetation cover at unoiled sites due to trampling and subsequent reoiling and damage to roots, rhizomes.	<3 for not trampled; 4 for trampled areas
1990 Exxon Bayway Refinery, NY and NJ[h]	Heavy (13,500 bbl No. 2 fuel oil)	No. but site was planted	*S. alterniflora*	Loss of 7.6 ha of S. *alterniflora*; replanted when denuded areas did not recover. 70% of planted/oiled areas successful after 3 years; unsuccessful if exposed to erosion or geese grazing. Unplanted areas not recovered after 7 years (end of study) due to erosion from waves and loss due to geese grazing.	>7 where not eroded
1974 T/B *Bouchard 65* Buzzards Bay, MA[i]	Heavy to Light (600 bbl No. 2 fuel oil)	Not in marshes	*S. alterniflora, Salicornia spp., Distichlis spicata, Suaeda linearis, Liminium carolinianum*	Long-term oiling of vegetation and sediments in sheltered harbor. Heavily oiled marshes had 100% plant mortality and erosion rates >24 times control site after 3 years. 3 years: *S. alterniflora* absent from low marsh and species shifts occurred; seedlings and sprouts appeared but died each summer. 25 years: oiled sediments/peat substrate were eroded and vegetation recolonized at lower elevations. Number of infauna species declined by 92%; mussel bed mortality reached 100%; fiddler crab mortality extensive; interstitial fauna in oiled samples was 21 individuals vs. 261 in control plots. Ribbed mussel repopulated to numbers greater than control plots in absence of other species.	>25 Except for permanent loss where marsh platform eroded

Oil Spill	Degree of Oiling (oil volume & type)	Shoreline Cleanup	Species	Documented Effect/ Impacts	Recovery (years)
1969 T/B *Florida*, West Falmouth, MA[j]	Heavy (4,385 bbl No. 2 fuel oil)	Not in marshes	*S. alterniflora, S. patens, S. virginica*	Heavy and persistent surface oiling and long-term re-oiling from sediments due to sheltered conditions. Immediate marsh mortality. 2 years; live vegetation present in lightly oiled areas; stem density and above- and below- ground biomass lower in oiled areas, leading to sediment erosion and marsh loss. 40 years: above- and belowground biomass were lower at oiled sites, and sediment erosion persisted due to residual petroleum. 27 years: fiddler crabs chronically exposed to spilled oil avoided burrowing in oiled layers, had delayed escape responses, lower feeding rates, and 50% lower densities than at control sites. Nearly 40 years later, ribbed mussels transplanted into oiled areas had slower growth rates, shorter mean shell lengths, and decreased filtration rates.	>40, possibly indefinitely
1990 Apex barges collision, Galveston, TX[k]	Unknown (16,700 bbl partially refined crude)	Sorbents, flushing, bioreme-diation	*S. alterniflora*	Salt marshes reportedly impacted by oiling.	Not reported
2005 Barataria Bay spill, LA[l]	Unknown (600 bbl South Louisiana crude)	No	*S. alterniflora*	Study of fish and crustaceans in oiled marsh fringes.	Not reported
2007 Bayou Perot, LA[m]	Heavy to Light (8,500 bbl >45 API gravity condensate)	Not in marshes	Not reported	Intermittent oiling of remote mud flats and salt marshes over a 7,770 ha area; 60 km of shoreline oiled. Small vacuum systems operated on airboats were most efficient at removing oil from mudflats to protect adjacent marshes.	Not reported
1998 Equinox blowout, Lake Grande Ecaille, LA[n]	Light (1,535 bbl South Louisiana crude)	No	*S. alterniflora, J. roemerianus, D. spicata, A. germinans*	494 ha of salt marshes oiled, mostly by sheens, though oil was pooled as a result of the high water due to the hurricane in small areas of the marsh.	Not reported
1992 Greenhill well blowout, Timbalier Bay, LA[o]	Moderate to Light (60 bbl of crude per hour for 13 days; much of it burned)	No	*S. alterniflora*	49.4 ha of salt marshes experienced dieback or had other impacts to productivity; erosion was a concern.	Not reported

Oil Spill	Degree of Oiling (oil volume & type)	Shoreline Cleanup	Species	Documented Effect/ Impacts	Recovery (years)
1997 Lake Barre, LA[p]	Heavy to Light (6,561 bbl South Louisiana crude)	No	*S. alterniflora*	An estimated 1,751 ha of marsh impacted based on percent vegetation cover and other metrics; increase in chlorosis and potential reductions in primary productivity. The habitat value of the oiled marsh was reduced.	Not reported
2004 Nakika MP-69 pipeline Hurricane Ivan oil discharge, LA[q]	Light (4,528 bbl South Louisiana crude)	Vegetation flushing, cutting	Not reported	1.2 ha of fresh and low salinity salt marsh were oiled and chlorosis observed about 10 days later.	Not reported

[a]NOAA (1999) ; [b]Michel et al. (1998), Michel and Rutherford (2013), Reilly (1998); [c]Little et al. (1990), Little and Little (1991); [d]Chan (1977); [e]Michel et al. (2003); [f]Hoff et al. (1993), Hoff (1995); [g]Moreno et al. (2004); [h]Bergen et al. (2000); Burger (1994), Louis Berger & Associates Inc. (1991); [i]Culbertson et al. (2008a); Culbertson et al. (2007); Peacock et al. (2007); Sanders (1978); Teal et al. (1992); [j]Texas General Land Office et al. (1997); [k]Roth and Baltz (2009); [l]Locke et al. (2008), Henry et al. (2008); [m]Locke et al. (2008); [n]LOSC et al. (2005); [o]Finley et al. (1995); [p]Lorentz et al. (2001); [q]RPI (2005)

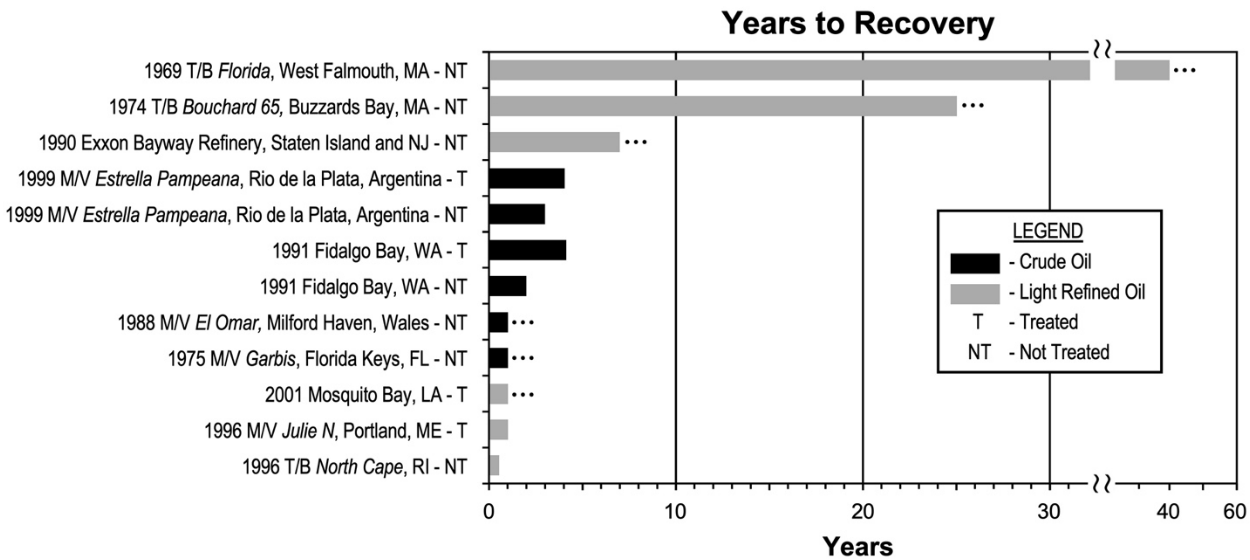

Figure 7-7. Recovery of salt marshes from the oil spills listed in Table 7-1
Black bars show recovery rates for spills of crude oil; grey bars show recovery rates for spills of light refined oils (diesel, No. 2 fuel oil, home heating oil). Dotted lines indicate incomplete recovery at the time of the most recent study.

Re-oiling over multiple tidal cycles in a sheltered setting had severe impacts on marshes in two spills in Buzzards Bay, MA–the *Florida* and *Bouchard 65*. Oil from these spills resulted in heavy oiling that killed above- and below-ground vegetation over 2–3 years followed by marsh edge erosion. Recovery of eroded areas had not occurred by the end of studies. The oil from both these spills was a No. 2 fuel oil, which resulted in even more severe impacts to vegetation due to the high concentration of toxic components in the oil (compared with crude oil) and deeper penetration into the soils. Vegetation in lightly oiled areas was still present after 2 years, and areas with freshwater seepage had lower plant mortality due to reduced oil penetration into sediments. Erosion rates were as much as 24 times greater in oiled areas than in unoiled areas after the *Bouchard 65* spill (Hampson and Moul 1978). Once oiled sediments and peat eroded away, *S. alterniflora* recolonized the unoiled, lower elevation areas, suggesting that if the marsh platform remains intact, vegetation can recolonize (Hampson 2000). After 27 years, above- and below-ground biomass of *S. alterniflora* was lower and variation in elevations was greater at oiled sites than non-oiled sites (Culbertson et al. 2008b). Peacock et al. (2007) concluded that the sediment accretion needed to maintain marsh vegetation may not be reached as a result of the sediment losses that occurred due to the *Florida* spill.

Replanting vegetation, even in oiled areas, can stabilize sediments and improve recovery times. Marshes heavily oiled by the Exxon Bayway Refinery spill and then planted with *S. alterniflora* recovered after 3 years if the vegetation was not exposed to shoreline wave energy or geese grazing; unplanted areas were still 90% bare at the end of the 7-year study (Bergen et al. 2000). Burning is also used to remove oil from marshes. Vegetation in marshes that were burned to remove condensate a week after a spill (Mosquito Bay) appeared to recover in weeks unless already killed by the condensate, whereas vegetation impacted by condensate prior to the burn had not recovered at the end of the 13-month study (Michel et al. 2003).

Light oiling in salt marshes generally has few adverse impacts, and marshes can recover over 1-2 growing seasons (Michel and Rutherford 2013). While lighter refined oils can be acutely toxic to vegetation when heavily stranded in the marshes and penetrated into the marsh soils (see above), evaporation and spreading into thin sheens prior to stranding usually reduce the amount of oil actually reaching and impacting a marsh. Although light oil sheens in marshes persisted for 6 months after the *North Cape* spill of home heating oil, no differences in vegetation abundance or growth patterns in oiled vs. unoiled areas and no differences in stem density, stem height, and biomass were found between post-spill and pre-spill salt marsh plots (NOAA 1999).

Cleanup activities can increase sediment oiling and damage roots and rhizomes in salt marshes, thus further delaying or preventing marsh recovery. Impacts of oiling during plant dormancy (when plants are reserving energy in roots and rhizomes) are typically less severe than if oiling occurs in the spring and summer growing season, but only if belowground roots and rhizomes are not damaged (Mendelssohn et al. 2012; Zengel et al. 2015; Zengel and Michel 2013).

Mortality typically occurs when plants and soils are heavily oiled (including oiling of most of the aboveground plant surfaces). Oiling from the *Garbis* spill in the Florida Keys killed marsh plants (e.g., *B. maritima, Salicornia sp., S. portulacastrum*) if plants or sediments were heavily oiled, while plants that were lightly oiled or occurred in lightly oil sediments recovered after 6 months (Chan 1977).

7.4 Impacts of Oil Exposure to Salt Marsh Invertebrates

Salt marsh invertebrates include intertidal benthic infauna (macroinvertebrates such as polychaetes, amphipods, and bivalves; and meiofauna such as copepods and nematodes, among many others); and intertidal benthic surface fauna and epifauna (macroinvertebrates such as crabs, shrimp, and gastropods). Some species occur in more than one of these groups; for example, fiddler crabs, *Minuca* and *Leptuca* spp., formerly *Uca* spp., that are both burrowing infauna and surface dwellers; and terrestrial insects and spiders that use the plant canopy and marsh surface.

Marsh invertebrates contribute to organic matter and nutrient cycling; affect vegetation, soils, microbes, and other invertebrates in a variety of ways (e.g., grazing benthic microalgae and marsh grasses, aerating the soils through bioturbation, stabilizing marsh shorelines); and are important prey for larger wildlife, fish, and shellfish species, including commercially and recreationally harvested species. Marsh-dependent or marsh-associated invertebrates such as penaeid shrimps, blue crabs (*Callinectes sapidus*), and eastern oysters (*Crassostrea virginica*), are of economic value.

Oil spills can affect salt marsh invertebrates in several direct and indirect ways, via toxicity, physical smothering and fouling, exposure to volatile oil components (for terrestrial species such as insects), habitat alterations (vegetation, soils, food availability), and influences on species interactions (competition, predation, facilitation). Heavy oiling can cause mortality and long-term impacts in marsh invertebrates, and while some invertebrates may recolonize after only a few years, changes to health and/or behavior can also occur. Three years after the *Bouchard 65* spill, Hampson and Moul (1978) reported a 92% reduction in the number of infaunal species; mussel bed mortality was as high as 100%; fiddler crab mortality was extensive; and numbers of individuals of infauna totaled 21 in oiled samples compared with 261 in control plots. In contrast, the ribbed mussel (*Modiolus demissus*) repopulated oiled areas in the absence of other species and after 3 years occurred in greater numbers in oiled than in unoiled plots. After the *Florida* spill, ribbed mussels were transplanted into oiled areas and 40 years later had slower growth rates, shorter mean shell lengths, lower condition indices, and decreased filtration rates, when compared with unoiled areas (Culbertson et al. 2008b). A study of fiddler crabs (*Uca pugnax*) 7 years after the *Florida* spill concluded that contaminated marsh sediments adversely impacted long-term survival and recovery in fiddler crabs. High oil content in sediments was directly related to reduced fiddler crab density, female to male ratios, and juvenile settlement, and sediment oiling was associated with high overwinter mortality, oil incorporation into body tissues, and behavior disorders such as abnormal burrow construction and impaired locomotion (Krebs and Burns 1977). Twenty-seven years post spill, fiddler crabs chronically exposed to oiling from the *Florida* spill avoided burrowing into oiled layers, suffered delayed escape responses, had reduced feeding rates, and had 50% lower densities than control sites. Six months after condensate was burned from marshes in Mosquito Bay, fiddler crabs were present in spite of oil that remained in some burrows (Michel et al. 2003).

7.5 Impacts of Oil Exposure and Treatment on Mangroves

Mangroves are especially vulnerable to oiling for many reasons. Most conspicuously, their aboveground root structures can be heavily oiled regardless of oil penetration into the soils, resulting in direct toxicity to tissues or physical smothering and elimination of gas exchange. Penetration of oil into soils increases oil persistence and exposure for root tissues, and oil that adheres to mangrove prop roots, stems, branches, and pneumatophores can damage the mangrove since these vegetative structures do not die and fall away from the roots to be quickly replaced by stems and shoots as in salt marsh grasses. The aboveground structures, including aerial roots, are also easily damaged by response activities. Sublethal effects to mangroves due to oiling include inhibited respiration with negative effects on mangrove propagules and seedlings (Shapiro et al. 2016). Because mangrove trees are long lived, recovery following forest mortality requires decades. Even if oiling does not result in immediate tree death, the surviving trees can become weakened and vulnerable to other natural stressors, eventually leading to death (Hoff et al. 2014). Impacts to propagules and seedlings can also have far-reaching effects on forest regeneration.

Mangrove seedling density, area of tree cover, and health indices are the only widely measured recovery indicators at many spills (Hoff et al. 2014). A more practical way to measure recovery may be to compare impacted and unimpacted systems that are similar in terms of location, species composition, tidal regime, and soil characteristics, using metrics such as tree height, trunk diameter, density, canopy cover, and abundance and diversity of associated invertebrates, fish, and plants. Because compromised ecosystems can be more vulnerable to stresses such as disease or predation, the recovering habitat should also exhibit the resilience of a functioning ecosystem. It is rare to find long-term, follow-up studies on mangroves beyond 1–2 years post spill. Studies that measure associated communities of invertebrates or other components of the mangrove forest other than the mangrove trees themselves are even rarer. Because mangrove tree cover and health are the only widely measured recovery indicators used at spills, they are used here to summarize impacts and recovery. Impacts to mangroves from spills of 500–20,000 bbl of crude oil, condensate, or diesel-like oils on marshes are summarized after Hoff et al. (2014) and others.

- Mangroves are especially susceptible to oiling impacts regardless of whether oil penetrates soils because of aboveground root structures;
- Lighter oils are more acutely toxic to mangroves than are heavier oils, although weathering generally reduces oil toxicity, and heavier oils can coat trees and reduce or prevent gas exchange necessary for photosynthesis and respiration;
- Oil-impacted mangroves may suffer yellowed leaves and defoliation, and they may die within a few weeks to several months (the onset of visible impacts can be delayed); more subtle responses include pneumatophore branching, germination failure, decreased canopy cover, and increased sensitivity to other stresses;
- Under severe oiling conditions, mangrove impacts may continue for years to decades, resulting in permanent habitat loss; and
- Mangrove-associated invertebrates and plants can recover more quickly from oiling than mangroves themselves (assuming sufficient habitat is present) because of the longer time for mangroves to reach maturity.

The number of spills reported to have adversely impacted mangroves far exceeds the number of studies documenting these impacts. The number of oil spills of 500–20,000 bbl known to have impacted mangroves from 1958 to 2016 includes 95 crude oil spills, two diesel spills, and one spill of condensate, based on data compiled by Duke (2016). Review of the literature on impacts from oil spills and response found data from five median-range size spills, one of which was experimental, were adequate for characterizing oil impacts to mangroves (**Table 7-2** and **Figure 7-8**). No studies were found that documented the impacts to mangroves from condensate or diesel for spills of this size range.

Table 7-2. Studies with documented or estimated impacts to and recovery of mangroves from spills of crude oil (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Oil Cleanup	Species	Documented Effect/ Impacts	Recovery (years)
1975 *Garbis*, Florida Keys, FL[a]	1,500–3,000 bbl crude oil emulsion	No	*R. mangle, A. germinans, Batis* spp.	Red mangrove seedlings with >50% oiling of leaves were dead ≤2 months; dwarf black mangroves with >50% oiling died; *A. germinans* with >50% oiling of pneumatophores, and some individuals in substrate still oiled 1 year later died. Even thin coatings on propagule stems damaged plant tissues; germination of oiled seeds declined 30% compared with unoiled plots. Coffee bean snail mortality due to habitat loss; *Uca* sp. emigrated to unoiled areas; benthic invertebrates in areas with high flow appeared unaffected.	20–30 for mature forest recovery
1983 Sao Paulo pipeline spill, Brazil[b]	16,250 bbl Brazil crude	No	*R. mangle, L. racemose, A. schaueriana*	1 year: Defoliation and deformation, and mortality of seedlings, saplings, and mature trees occurred. 3 years: Decline of 24% in density and 40% in basal area, primarily in *A. schaueriana* (most sensitive to oiling); *R. mangle* was least sensitive to oiling; *L. racemose* response was intermediate. 5–6 years: Recolonization but followed by 100% mortality of saplings. Recovery anticipated over >20 years. Herbivory on mangroves declined during first 2 years as grazers migrated elsewhere for food and then stabilized.	>20
2003 Moreton Bay pipeline spill, Queensland Australia[c]	11,900 bbl crude	Impounded for 1 week to float oil for removal	*A. marina, occasional R. stylosa, Aegiseras corniculatum*	Heavy oiling (up to 1 m thick at surface) resulted in loss of 0.08 ha (83%) of mangrove trees, 60% of pneumatophores, >50% of the foliage, and 100% of epiphytic algae, compared with unoiled and control sites. Mangrove sprouting occurred after 2 years. Mangrove crabs had initial 100% mortality; after 12 months crab biomass was 55% lower in oiled sites than control and reference sists. Species' distributions altered; reduced crab abundance and biomass corresponded to high mangrove mortality at oiled site.	>20
1979 *World Encouragement*, Botany Bay, NSW Australia[d]	803 bbl Arabian crude, bunker fuel	No	*A. marina*	After 4 months: Browning and defoliation. 5–8 months: dieback and mortality. 12 months: Severe mortality of trees >8 m in height in oiled sediment. 5 years: 4.4 ha of trees dead. 60% recovery after 18 years, including recolonization.	>20
1984 TROPICS field oiling experiment, Panama[e]	6 bbl Prudhoe Bay crude	No	*R. mangle*	4 months: Defoliation of about 25% of mangroves. 7 months: Loss of 25 trees (17% reduction). 10 years: 9.1% reduction in total basal area of mangrove trees at oiled site due to tree mortality. 32 years: Canopy cover, root density lower at oil only sites than reference sites; soil subsidence was greater. Mangrove tree snails declined about 50% at oil only sites and remained lower than pre-spill levels until about 1 year post oiling; vertical distribution was shifted. Oysters had no measurable increase in mortality.	>32

[a]Chan (1977); [b]Lamparelli et al. (1997), Santos et al. (2012); [c]Prosser (2004), Duke et al. 2005, Duke and Burns 1999; [d]Allaway (2009), Duke and Burns (1999); Renegar et al. (2017), DeMicco et al. 2011), Ballou et al. (1987), Dodge et al. (1995)

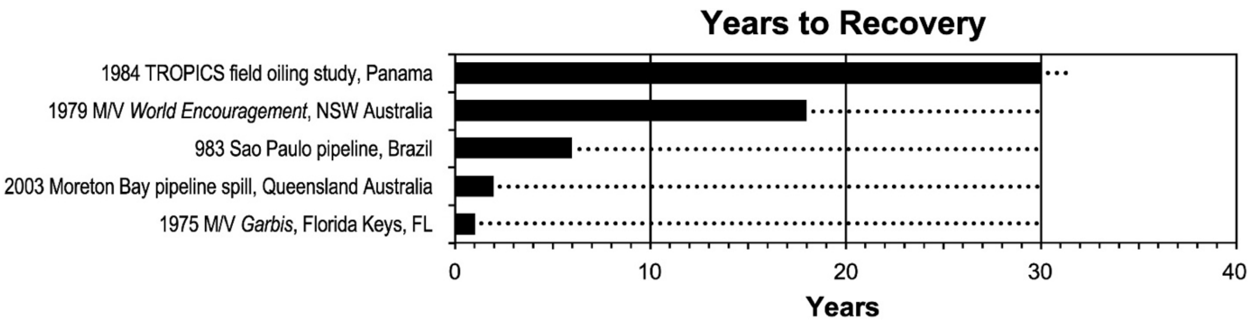

Figure 7-8. Recovery of mangroves from the oil spills listed in Table 7-2
Dotted lines indicate incomplete recovery at the time of the most recent study but projected time is based on time for the mangrove forest to reach maturity.

Mangrove responses to oiling can occur slowly, beginning with acute, short term impacts among smaller trees and followed by more severe chronic impacts in larger trees (**Table 7-3**). Depending on oiling and response activities, various pathways to recovery or permanent loss of habitat may occur. Pathways by which recovery (or loss) of undisturbed and oil impacted mangroves are outlined in **Figure 7-9**.

Table 7-3. Generalized responses of mangrove forest to oil spills (after Lewis 1983)

Stage	Observed Impact
Acute	
0 - 15 Days	Deaths of birds, fish, invertebrates
15 - 30 Days	Defoliation and death of small (<1 m) mangroves Loss of aerial root community
Chronic	
30 Days - 1 Year	Defoliation and death of medium (<3 m) mangroves Tissue damage to aerial roots
1 Year - 5 Years	Death of larger (>3 m) mangroves Loss of aerial roots Regrowth of roots (sometimes deformed) Recolonization of oiled areas by new seedlings
1 Year - 10 Years	Reduction in litter fall Reduction in reproduction Reduction in seedling survival Potential death or reduced growth of recolonizing trees Potential for increased insect damage
10 Years - 50 Years	Complete recovery

Recovery of mangroves from heavy and persistent oiling often takes decades, while sublethal impacts may be followed by recovery in 1–10 years if there are live trees from which to recover a canopy (Duke 2016; 1999; Duke and Burns 1999). Recovery following tree mortality requires much more time due to the need for seedling recruitment, establishment, and growth of trees, and as long as 25–30 years for recovery of a mature forest (Duke 2016), if recovery occurs at all. Using data from fourteen spills in Australia and Panama (some with dispersant), Duke and Burns (1999) demonstrated a linear response of mangrove recovery to time since spill (r=0.875); full recovery did not occur until 30 years after a spill. Tidal flushing is critical to oil degradation and mangrove recovery. However, wave exposure that weathers and removes oil from mangroves may also erode the oiled mangroves and substrates and can result in permanent loss.

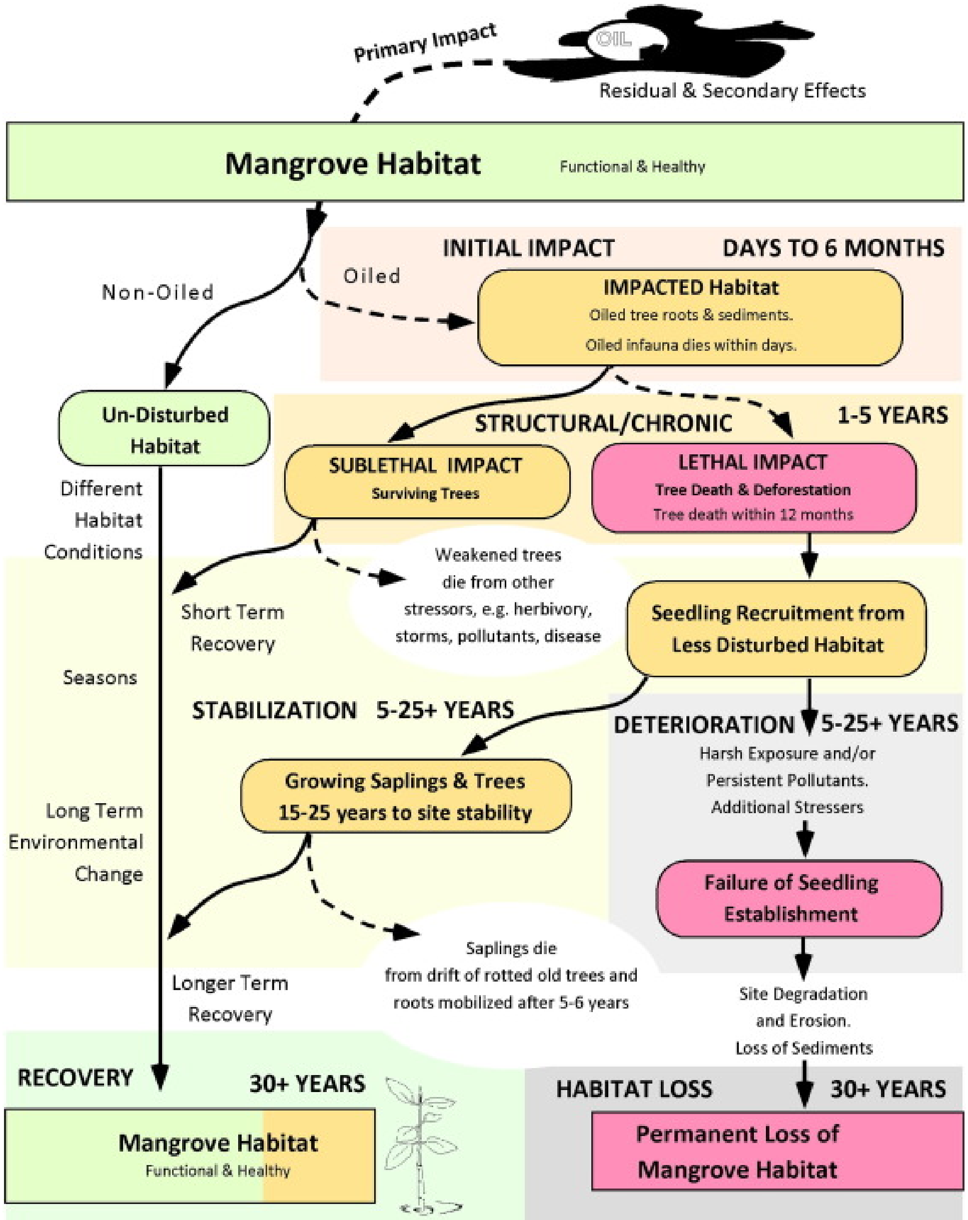

Figure 7-9. Schematic of the effects of a large oil spill on mangrove forests
Major pathways are shown as with recovery (on left) or permanent loss (on right) (adapted from Duke et al. 1999).

More than 50% oiling of mangroves may be enough to result in mortality over the short and long term. After the *Garbis* spill in the Florida Keys, red mangrove propagules with >50% leaf oiling were dead within 2 months; oiling of <50% of dwarf black mangroves (*A. germinans*) pneumatophores did not appear to affect trees at first but a year later, trees with <50% oiling surrounded by oiled soils died (Chan 1977). Even thin coatings of oil on propagules stems caused scars and damaged tissue, and oiled seeds showed a 30% decline in germination when compared with unoiled seeds.

Oily debris was temporarily retained in mangroves after the spill, resulting in re-oiling in the mangroves, which remained oiled for at least a year (no further monitoring followed). Extensive mangrove seedling mortality was observed in the mangrove fringe, and some died where the surrounding substrate remained oiled 1 year later. At the end of the year-long study, oil was not observed on the beaches or marshes but remained in the mangroves.

In the study on the Moreton Bay, Australia, pipeline spill, mangroves were impounded after the crude oil spill to float the oil off the sediments and remove most of the oil and then monitored at 12 and 20 months after the spill. Much of the impact was attributed to the toxicity of the volatile and dissolved fractions of the oil rather than smothering (Duke et al. 2005). Mortality after 1 year included loss of 0.08 ha of mangroves, 83% of the mangrove trees, 60% of the pneumatophores, and 100% of epiphytic algae. Mortality was greater than for the control (impounded/ unoiled) sites; there were no differences between control and reference sites (Prosser 2004). A seedling recruitment event at 4 months resulted in lower numbers of seedlings at the oiled site than in control and reference sites. After 20 months, many trees had a dead and decaying canopy, but some trees appeared to be recovering, evidenced by basal sprouting (Duke et al. 2005).

Mortality of mangrove seedlings, samplings, and pneumatophores is often observed in the first few months after a spill (i.e., acute or short-term effects). Unless long-term monitoring is implemented, the extent of the mortality and recovery, if it occurs, may not be investigated. For example, extensive browning and defoliation among mature trees and extensive mortality of seedlings were reported for heavily oiled portions of mangroves during 4 months of monitoring after the *World Encouragement* spill, while no mortality was reported in unoiled sites (Allaway 2009). Observations at 5 months indicated die-back among mangroves (i.e., browning of all leaves of the trees), followed by complete defoliation and mortality after 8 months when monitoring ended, and further damage was not observed. A year later, visits to the site revealed extensive and severe mortality among mangroves as large as 8 m in height on heavily oiled sediments; 5 years later, 4.4 ha of mangroves were dead (Duke and Burns 1999, Allaway 2009). The extent of mortality was not quantified or followed further.

The benefit of long-term monitoring of oiled mangroves is information on which to base selection of response options to reduce the effects of the oil (e.g., use of dispersants offshore). Data from nearly 10 years of monitoring (1984-1992) mangroves after a pipeline spill of crude oil documented initial mortality of mature trees, seedlings, and propagules, followed by recolonization and 100% mortality at 6 years post spill, and then survival and growth of new trees at 9 years (Lamparelli et al. 1997). Leaf defoliation and leaf deformation (e.g., fading, twisting) was high but varied by species' oil sensitivity (and corresponding osmotic regulatory mechanisms). Defoliation was least (25.9%) for *R. mangle*, a salt-excluder and the least sensitive of the three species; intermediate (43.4%) for *L. racemose*; and greatest (64.5%) for *A. schaueriana*, a salt-excreter and the most sensitive to oiling. Tree density declined by 24% by year 3, but occurred mostly by 1 year post spill, and corresponded to a 40% reduction in total basal area (primarily in *A. schaueriana*). At 7 years post spill, dead trees, exposed substrate, and recovering mangrove stands with reduced structural development were reported, based on aerial imagery (Santos et al. 2012). Observations 10 years post spill indicated survival and beginning recovery, which was anticipated to require more than 20 years (Lamparelli et al. 1997).

Experimental (i.e., intentional and controlled) mangrove oiling was undertaken as part of the 1984 TROPICS spill in Panama and provided extensive data demonstrating the adverse and long-term impacts to mangroves from oil only and dispersed oil. Defoliation at the oil only site was observed in about 25% of the mangrove area by 4 months post spill (Ballou et al. 1987). Mortality immediately following exposure included 18 trees and increased to 25 trees (17% reduction) after 7 months. At 10 years post spill, there was a 9.1% reduction in total basal area of mangrove trees at the oil only site due to mortality (number of trees reduced by 46%, from 149 to 80 trees) (Dodge et al. 1995). Renegar et al. (2017) reported no measurable long-term effects on mangroves at the dispersed oil site. After 25 years, the number of small trees at the oil only site increased compared with numbers similar to baseline conditions at the dispersed oil (124 trees) and reference (392 trees) sites (DeMicco et al. 2011). In 2013, 29 years later, the counts of adult trees had recovered though there were still an abundance of small trees at the oiled site and curling and distortions of prop roots were conspicuous (Baca et al. 2014).

Thirty-two years after the TROPICS experimental oiling, damage to mangroves was still apparent. Mangrove canopy cover and mangrove root density were lower at the oil only site compared to the dispersed oil or reference sites, although no differences between sites were documented for mature tree density or sapling-to-mature tree ratio score. Trees showed signs of damage including curling and distortions of prop roots in small trees and seedlings (Baca et al. 2014). Renegar et al. (2017) concluded that non-dispersed oil penetrated and remained at least partially trapped in the mangrove substrate, resulting in chronic re-oiling in low concentrations over many years.

7.6 Impacts of Oil Exposure to Mangrove Invertebrates

Impacts to mangrove-associated invertebrates from oiling can be severe because oil-covered prop roots no longer provide habitat for epifauna, and oil that penetrates soils exposes benthic infauna to oiling toxicity. Invertebrates can move to unoiled areas after a spill, followed by changes in their distributions (vertical or horizontal), and/or recolonize once conditions improve. Others may be trapped and die, or their behavior may be altered, which may or may not increase rates of survival.

After the *Garbis* spill, *Uca* sp. emigrated from oiled to clean substrates when flood tides delivered oil, which flowed into crab burrows. The coffee bean snail (*Melampus coffeus)* that normally travels up and down mangrove prop roots and trunks with the tide to feed on detrital material at low tide was precluded from feeding by the oiled trunks and prop roots. Where mangroves were heavily coated with oil, the snail did not cross the oil to feed at low tide until the oil became tacky (about 4 weeks) and the snails eventually died (Chan 1977). Oysters (*Crassostrea virginica)* were covered and/or killed by the oil; flat tree oysters (*Isognomon alatas)* also suffered mortality in cases of heavy physical coating. Some invertebrates survived in high-flow areas, and live horseshoe crabs (*Limulus)*, hermit crabs (*Paguristes* sp.), blue crab (*Callillectes sapidus)*, and the gastropod *Batillaria minima* were found in the oily water, seemingly unaffected.

Mangrove tree snails (*Littorina angulifera*) declined in abundance by about 50% and their vertical distribution was altered within 4 months of the TROPICS experimental spill (Ballou et al. 1987). Numbers of snails remained low until about 1 year later, although their distribution shift to higher elevations in the mangroves persisted. Flat tree oysters (mainly *I. alatus* but also *Crassostrea virginica*) showed no measurable increase in mortality after the spill despite uptake of high levels of hydrocarbons.

Recovery of invertebrates such as mangrove crabs from oiling is related to the loss of mangrove habitat. Mangrove crabs recovering after 100% mortality following the Moreton Bay (Australia) spill still had 55% lower biomass in oiled sites than unoiled and control sites 16 months after the spill (Prosser 2004). Three crab species were found at oiled and unoiled sites (*Parasesarma erythrodactyla, Australoplax tridentata*), and *Helograpsus haswellianu; Perisesarma messa* was absent only from the oiled site. One species (*H. haswellianus*) expanded its vertical distribution in the oiled mangrove, indicating a potential species shift. Fewer crab burrows and lower crab abundance and biomass corresponded to high tree mortality in the oiled site, and the reduced number of pneumatophores (refuge from predators) was thought to increase risk of predation on the crabs.

7.7 Summary and Information Needs for Assessing Impacts to Salt Marshes and Mangroves

Every oil spill represents a different combination of oiling and environmental conditions that influence oil behavior and persistence as well as habitat impacts and recovery. The understanding of how oil interacts with sediments, soils, and vegetation in marshes and mangroves is, however, adequate to support impact evaluation. Heavy oiling of marshes from No. 2 fuel oil (e.g., Exxon Bayway Refinery, *Florida*, and *Bouchard 65* spills) can have long-term impacts on salt marshes due at least in part to the acute toxicity of the oil, followed by root and rhizome death and sediment subsidence. After several years, vegetation recolonized areas where oiled sediments had subsided as long as the marsh platform remained. Mangroves killed by prop root oiling and sediment toxicity (e.g., *Garbis,* Sao Paulo*,* and TROPICS spills) require 20–30 years for new trees to establish and mature. Mangroves oiled by the experimental TROPICS spill had lower canopy cover and biomass when compared with reference sites even 40 years after the spill; mangroves survived and appeared to begin recovery after 10 years following the Sao Paulo pipeline spill.

For lightly oiled salt marshes, recovery typically occurs within 1–2 growing seasons, in the absence of damage from cleanup treatments. Salt marshes that are heavily oiled at the surface may require 10 years or more for recovery, and recovery may be precluded altogether if belowground biomass is damaged to the degree that the marsh edge subsequently erodes. Recovery of salt marshes is longest for spills with the following conditions: heavy, persistent, and prolonged oiling; cold and/or arid climates; sheltered settings; oils that penetrate or become mixed into the soils; oils that form persistent thick surface residues; and intensive or overly aggressive cleanup. Recovery was shortest following spills with the following conditions: warm, humid climates; light to heavy oiling of the vegetation only (not the soils); or less intensive (or no) cleanup activities. Ambient-water flushing may be useful for oil cleanup in marshes as long as soils are not disturbed. Burning of surface oil in salt marshes can remove surface oiling and may not damage roots and rhizomes, but subsurface oiling will likely persist.

Mangroves are more susceptible to severe impacts than salt marshes, primarily due to aboveground roots that can be heavily impacted by surface oiling. Only those mangroves that are lightly oiled, and where oil is removed quickly by currents and tides so that the impacts are relatively temporary, are likely to recover. Full recovery in mangroves where mortality or severe damage has occurred is likely to take 20–30 years for a mature forest canopy to re-develop.

The limited number of oil spills in the range of 500–20,000 bbl with adequate information for assessing impacts to salt marshes and especially mangroves demonstrates that few studies directly document the effects of extensive oiling in these habitats, especially over long time periods. Marsh and mangrove habitats impacted by the *Florida* and *Bouchard 65* spill were not recovered after as many as 40 years, making it difficult to predict how or when they will recover. Studies of the Fidalgo Bay and *Estrella Pampeana* spills indicate that trampling can be a major cause of damage from even less aggressive response activities (e.g., cutting, ambient-water washing) and can damage roots and rhizomes physically or increase soil oiling and subsequent toxicity to below ground roots and rhizomes in marshes. Recovery in some heavily oiled areas after the Exxon Bayway Refinery spill demonstrated the potential benefits of re-planting, even without removing oiled substrate, to stabilize soils and reduce erosion.

Previous studies often have inadequate sampling or replication, limited scope in space and time, and confounding factors that make it difficult to quantify the degree of impacts as related to oil exposure. Even fewer studies have documented full ecological recovery. Further field studies and collection of empirical data will be helpful, especially with respect to recovery times for mangroves. The extent of mangrove loss and recovery can be measured using aerial photography, which has been used successfully for long-term monitoring (Duke 2016). This method can be used to differentiate between sublethal and lethal impacts and to monitor previously impacted mangroves that have not been tracked for many years. However, field verification of remote-sensing studies in mangroves is also very important. Long-term transplant experiments may prove important for regeneration of mangroves with respect to oil persistence in sediments.

The continued incidents of oil spills and the relative paucity of empirical data available for spill impacts in salt marshes and mangroves suggest that long term monitoring should be a standard response to spills. Rigorous field study designs should be prepared and readied for implementation in the event of a substantial spill. Peterson et al. (2001), Cosco Busan Oil Spill Trustees (2012), Peterson et al. (2008), and Duke (2016) are good references for developing successful post-spill impacts studies for salt marshes and mangroves, respectively.

8 Beaches and Tidal Flats

8.1 Habitat Description, Communities, and Ecological Functions and Services

This chapter addresses potential impacts of spills of 500–20,000 bbl of crude, condensate, or diesel to the following coastal and estuarine habitats: sand beaches, exposed (sand) tidal flats, gravel beaches (including sand and gravel beaches), and sheltered (muddy) tidal flats. These sedimentary habitats have typical morphologies that are a function of their grain size, exposure to waves, tidal range, and sediment source. Furthermore, these areas, and the adjacent habitats to which they are connected by physical and biological processes, supply a broad range of ecosystem services including: nesting, feeding, and roosting areas for birds; nesting for sea turtles; rookeries and haul-outs for pinnipeds; habitat and trophic support for fish and marine and terrestrial invertebrates; sediment storage and transport; nutrient mineralization and recycling; fisheries production; subsistence; coastal protection; and recreation. Impacts to dune habitat and communities are discussed under Chapter 16 – Terrestrial Habitats and Wildlife.

Sand beach invertebrate communities are often dominated by a handful of species, which contribute greatly to the overall energy budget of these habitats. Sand beaches provide habitats for diverse and ecologically functional fauna ranging from interstitial primary producers and decomposers (bacteria, protozoans, microalgae) to small and large secondary invertebrate consumers (meiofauna–organisms retained by a 100-µm mesh sieve, and macrofauna–organisms retained by 1-mm mesh sieve) including epifauna and infauna. Interstitial and actively burrowing meiofauna and macroinvertebrates, such as crustaceans, mollusks, and polychaetes, are especially characteristic of sand beaches, and the continuous supply of wrack supports a rich supratidal fauna of crustaceans and insects. **Figure 8-1** shows typical sand beach zonation and dominant macrofauna for the Gulf of Mexico and southern Atlantic regions.

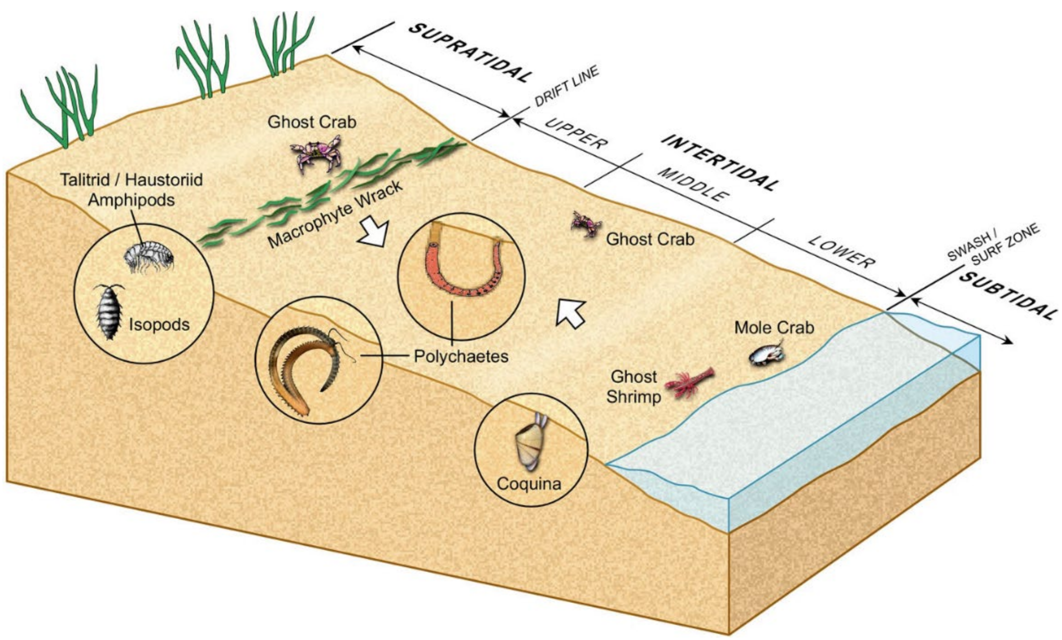

Figure 8-1. Typical sand beach zonation and dominant macrofauna
Distribution of representative sand beach invertebrates for the Gulf of Mexico and southern Atlantic regions. The supratidal zone is above the macrophyte wrack drift line. The upper, middle, and lower intertidal zones grade into each other and are not necessarily rigid in their extent. The fauna of the middle intertidal zone is a combination of those found in the upper and lower intertidal zones (represented by white arrows). From Michel et al. (2017).

Gravel beaches have very different morphologies and macrofaunal communities than sand beaches, depending on the climate, grain size, and degree of exposure to wave energy. Gravel beaches have high primary and secondary productivity, particularly those that are semi-exposed to wave energy. For example, in Puget Sound, red, brown, and green algae and kelp cover the gravel at the lower intertidal zones, and epifauna attached to or hiding under the cobbles include barnacles, anemones, crabs and smaller crustaceans, and many types of snails. Infauna are likewise diverse, with many species (wide diversity of polychaetes and other worms, small crustaceans, and other invertebrates) and high biomass (Dethier 2010). Gravel beaches provide many of the same ecosystem services as sand beaches.

Figure 8-2 shows the morphology and representative macrofaunal communities of an exposed, coarse-grained gravel beach. The landward half of the profile is primarily depositional, with the storm berm being activated during major storms and the minor, accretionary berms building up between storms. The seaward half of the profile is an area where structural strengthening occurs by either clast imbrication (a process whereby the maximum projection areas of the clasts dip toward the strongest force acting upon them, in this case the uprush of the breaking wave), or by the formation of a surface armor of coarse gravel (a well-sorted surface layer of the coarsest gravel on the beach that is extremely stable). Once imbrication and/or armoring occurs, very strong currents are required to remobilize the sediments, particularly those in the subsurface. Hayes and Michel (2001) identified the formation of coarse-grained surface imbricate and armor zones on the mid- and lower intertidal elevations as a factor in the persistence of subsurface oil. The armor slows the rate of reworking of the sediments, which also provides a more stable habitat for macrofauna.

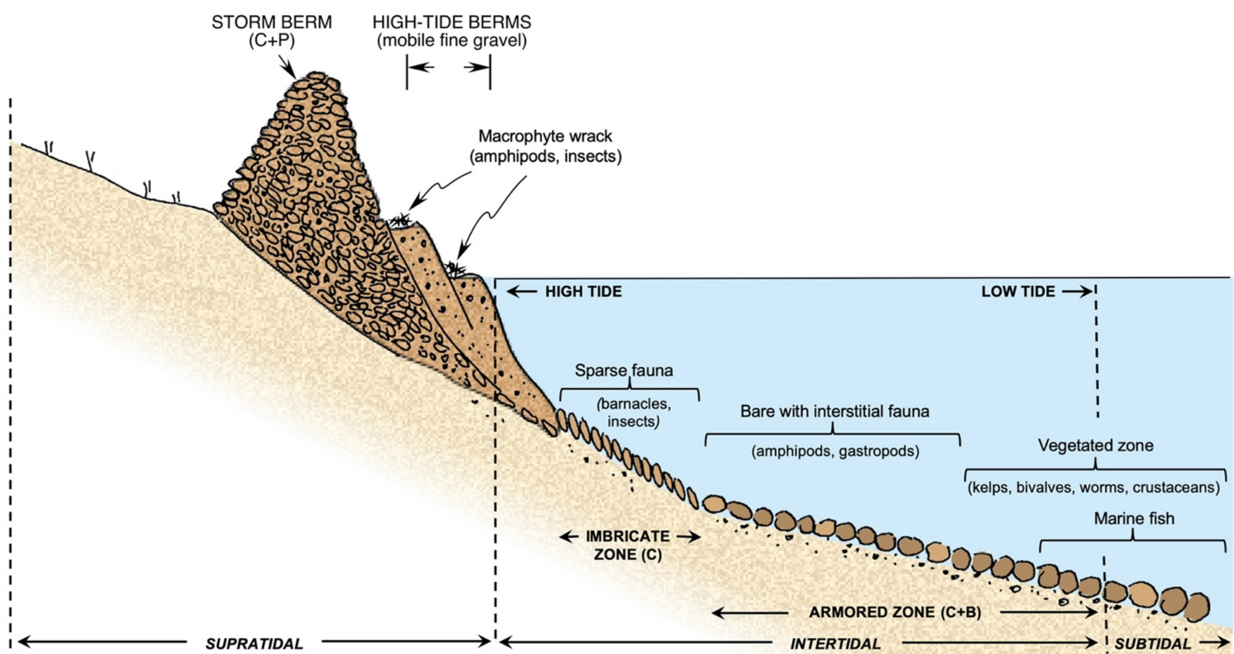

Figure 8-2. Morphology and representative macrofaunal communities of an exposed, coarse-grained gravel beach

Typical beach profile, sediments, and distribution of representative macrofaunal communities of an exposed, coarse-grained gravel beach. P = pebbles. C, cobbles; B, boulders. The supratidal zone includes multiple berms, with the storm berm the highest and multiple high-tide berms. Modified from Hayes and Michel (2001).

Tidal flats are unvegetated, flat, intertidal habitats that are composed of mostly sand and/or mud. Sand flats are exposed to enough wave and tidal energy to winnow the finer-grained sediments, often at the entrances to estuaries or adjacent to tidal inlets. Mud flats occur further up the estuary and away from inlets. Both types of tidal flats have rich benthic communities (**Figure 8-3**) and play an important role in the ecological function of coastal and estuarine ecosystems, particularly in regard to primary production, secondary production, and water quality. Tidal flats provide the following ecological functions: 1) nursery grounds for early stages of many species; 2) refuges and feeding grounds for a variety of forage species and juvenile fishes; 3) important trophic support to birds, sea otters, fish, and shellfish; and 4) modulation of sedimentary nutrient fluxes (they have rich microalgal and bacterial communities).

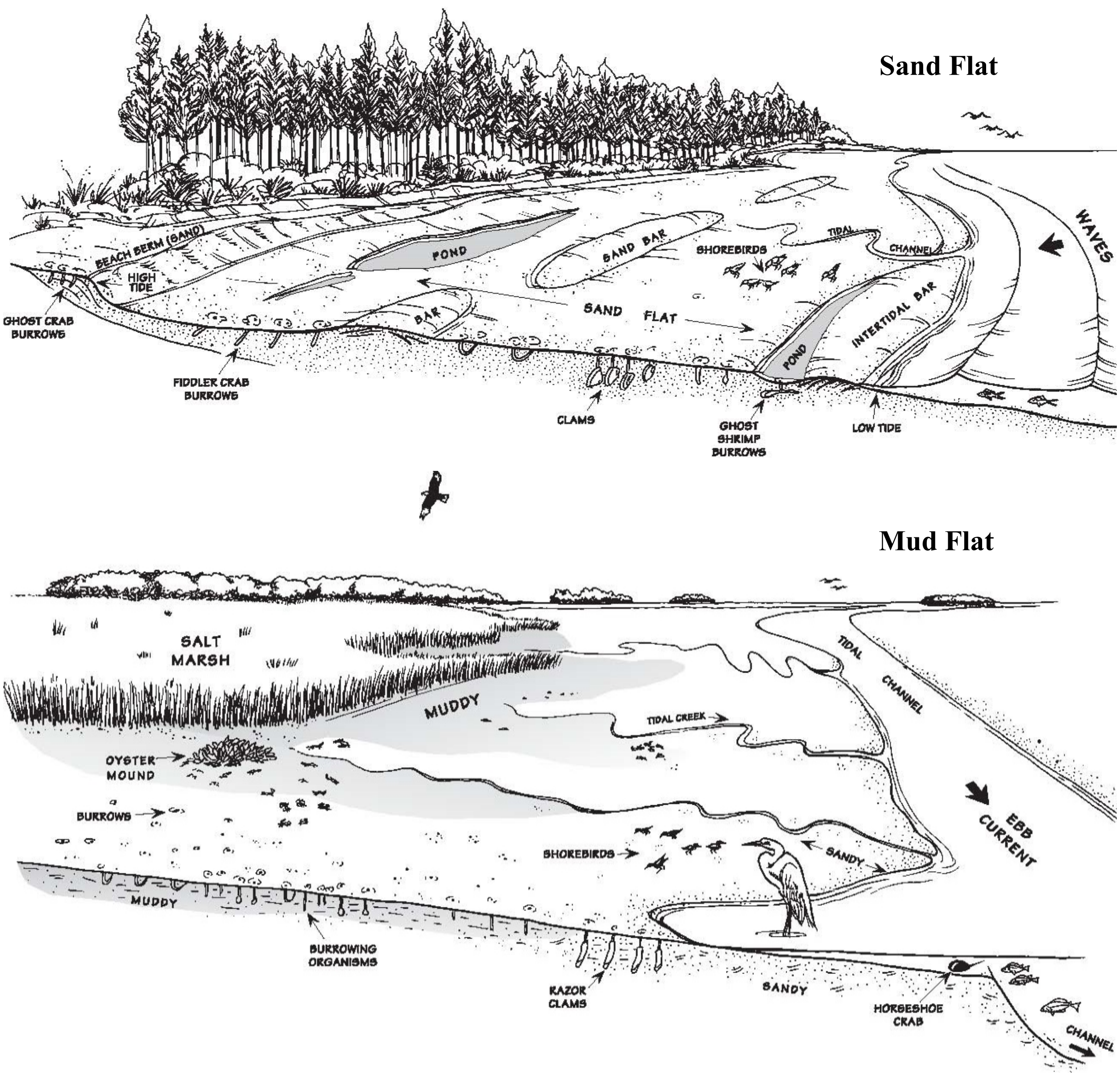

Figure 8-3. Examples of sand and mud flats and associated biota
Top: Sand flat habitat and associated communities. Bottom: Mud flat and associated communities. From NOAA (2010a).

8.2 Oil Behavior and Persistence on Beaches and Tidal Flats

8.2.1 Oil Behavior and Persistence on Beaches

Oil penetration and burial in beaches and tidal flats are functions of oil loading, oil viscosity, grain size (permeability), and sediment dynamics including transport and deposition rates. The degree of oil penetration and burial affects oil persistence, selection of response options, and potential impacts to infauna.

Of the sixty-two oil spills and field experiments (BIOS and TROPICS) listed in **Table 1-1**, eleven spills had sufficient information on which to describe oil distribution and behavior and to estimate oil persistence on beaches (**Table 8-1**). These data show that light refined oils such as diesel, No. 2 fuel oil, and home heating oil have the shortest persistence, from weeks to less than 1 year. These oils have higher penetration into the permeable sediments, up to 1 m, but only for spills where the oil was released very close to or right on the shoreline, allowing for heavy accumulations, such as the *World Prodigy* and *North Cape* spills, respectively (Mulhare and Therrien 1997).

Table 8-1. Spills with information on oil behavior and persistence on beaches

Oil Spill (Shoreline Oiling, km)	Degree of Oiling (oil volume & type)	Oil Cleanup	Documented Oil Distribution and Behavior/Beach Type	Oil Persistence (years)
1972 Long Island Sound, CT[a]	Light to Heavy (1,905 bbl No. 2 fuel oil)	No	Localized heavy oiling of coarse sand beach, large storm on day 2, no oil detected on day 16.	<0.1
1997 Torch Platform, Offshore Santa Barbara, CA (27 km)[b]	Light to Moderate (163 bbl crude oil emulsion, 21 bbl diesel)	Yes, manual, heavy equipment	Sand beaches had up to 50% coverage of thick oil ribbons and patties; reoiling required cleanup to extend to over 6 weeks.	<0.25
1984 Eagle Creek, Queen Charlotte Island, Canada[c] (<1 km)	Heavy (1,000 bbl diesel; 240 bbl gasoline)	No	Release was into the surf zone on an exposed shoreline, where the oils penetrated into the sand beach to the water table; removal was by tidal pumping of the ground water.	<0.5
1989 T/V *Bahia Paraiso,* Antarctica (a few km)[d]	Heavy (3,760 bbl diesel fuel arctic)	No	Released into the harbor; Most of the diesel in the pebble beaches evaporated or was removed by wave action. Reoiling from chronic leaks from the vessel occurred 2 years later.	<1
1989 T/V *World Prodigy*, Narraganset Bay, RI[e]	Heavy (6,500 bbl home heating oil)	Yes, excavation and aeration on high-use public beach	High public use sand beach in Mackerel Cove was heavily oiled, to depths of ~ m.	<1
2015 Refugio Beach, CA (214 km)[f]	Light to Heavy (500 bbl Monterey crude)	Yes, manual and often	Oil stranded at the upper intertidal and supratidal zones, with penetration of up to 5 cm and burial by 10+ cm on sand and gravel beaches.	<0.5 Based on tPAH in porewater
1996 T/B *North Cape*, South Kingstown, RI[g]	Light (19,700 bbl home heating oil)	No	Oil stranded on sand beaches during a major storm and penetrated to 1.3 m, with highest concentrations at the storm high tide line at the barge grounding site. Oil persisted for at least 7 months.	<1

Oil Spill (Shoreline Oiling, km)	Degree of Oiling (oil volume & type)	Oil Cleanup	Documented Oil Distribution and Behavior/Beach Type	Oil Persistence (years)
1988 T/V *El Omar* Milford Haven, Wales[h]	Heavy (670 bbl Iranian crude)	Yes	After 3 months, surface oil on gravel beaches was removed (TPH decreased from 70,000 to 850 ppm), whereas subsurface oil persisted (decrease from 4,750 to 1,930 ppm) and was less weathered.	>1
1975 T/V *Garbis*, Florida Keys[i]	Moderate (1,500–3,000 bbl crude oil emulsion)	Limited	Oil on sheltered sand beaches formed a hard, tarry layer of sand/waxy residue over the oiled sand at 6 weeks. Oil beneath this crust remained fluid. Small patches remained at 6 months. No oil was observed 1 year later.	1
1985 T/V *Arco Anchorage*, Puget Sound, WA (24 km)[j]	Heavy (5,690 bbl Alaska North Slope crude)	Yes, agitation	Heaviest oiling was in Ediz Hook sand and gravel beaches, with initial TPH concentrations up to 20,000 ppm; TPH reached background by July 1987.	1.5
1999 T/V *Estrella Pampeana*, Rio de la Plata, Argentina[k]	Heavy (15,700 bbl crude)	Yes	On outer sand beaches, aliphatic compounds degraded in 13 months, and PAHs in 42 months.	3.5
1989 T/V *World Prodigy*, Narragansett Bay, RI[l]	Light to Heavy (6,900 bbl home heating oil)	No, difficult vehicular access; not a public bathing beach	Mixed sand and gravel beach in Hull Cove had no sheening so left for natural recovery. Initial TPH concentration was 15,000 ppm.	5.5
1981 Baffin Island, Canada BIOS Field Experiment (0.4 km)[m]	Heavy (94 bbl aged Venezuelan Lagomedio crude)	No	In the first 3 ice-free months, there was a 70% oil volume reduction on the mixed sand and gravel beach; asphalt pavement formed after 4 ice-free months. In next 18 ice-free months, 80% reduction; little change for 1993-2001, with small patches containing both fresh and highly degraded oil.	>20

[a]EPA (1973); [b]Torch/Platform Irene Trustee Council (2007); [c]McLaren (1985); [d]Kennicutt et al. (1991b), Kennicutt and Sweet (1992); [e]Mulhare and Therrien (1997); [f]Nixon (2018), Refugio Beach Oil Spill Trustees (2020); [g]Mulhare and Therrien (1997); [h]Little et al. (1990); [i]Chan (1977); [j]Blaylock and Houghton (1989); [k]Colombo et al. (2005); [l]Mulhare and Therrien (1993); [m]Owens et al. (2002)

For open-water spills, diesel-like oils tend to spread quickly into thin slicks without enough oil to saturate beach sediments. Diesel-like oils have very low viscosity, and surface sheens are readily dispersed into the water column when winds exceed 3.5 m/second or sea conditions are >0.5 m (NOAA 2019). These types of oils are not very sticky or viscous, compared to heavier oils. All of these factors lead to less shoreline oiling and higher removal rates for diesel-like oils.

As shown in **Table 8-1**, shoreline cleanup was only conducted on one of the five spills of diesel-like oils. The rapid removal/degradation of these light oils often leads to the decision to avoid further impacts that could result from oiled sediment removal. The exception was the *World Prodigy*, where a heavily oiled and high-use amenity beach was oiled at the peak of the summer season. The decision was made to excavate the sand for placement above the high-tide line and to aerate it to speed the oil removal, with later replacement of the treated sand back on the beach after Labor Day.

Crude oil spills tend to be more persistent than diesel-like spills, and shoreline cleanup is almost always used when this type of oil strands on beaches. Oil can cover the entire intertidal zone at low water (**Figure 8-4 Left**), though in most cases the oil is lifted by the rising tide and deposited in the upper intertidal zone and, during high wave conditions, in the supratidal zone.

Figure 8-4. Heavy Monterey crude oil on mixed sand and gravel beaches at the 2015 Refugio Beach spill
Left: Shortly after the oil stranded on a sand beach, covering the entire intertidal zone at low tide; Right: 1–2 cm of oil buried by 10–15 cm of clean sand days later.

Oil penetration and persistence is greatest on gravel beaches. Hayes and Michel (2001) identified the following features that enhance the persistence of oil in gravel beaches:

- High porosity and permeability that allow deep penetration from the surface. The open framework of gravel beaches allows penetration to depths as great as 1 m.
- Potential for deep and rapid burial by clean sediments.
- Gravel tends to be highly mobilized during peak and waning stages of storm activity. The finer components of gravel–granules, pebbles, and small cobbles–also are moved readily by normal waves. The gravel may be moved either perpendicular to the beach, in the form of berms or swash bars, or parallel to the beach, in the form of rhythmic topography. Thus, gravel movement can bury oil for months to years.
- Presence of localized, sheltered areas where oil can persist for years. Shorelines with gravel beaches tend to be irregular in outline, and localized sheltering from wave action is common. At spills, oil has persisted for years in wave shadows formed behind large boulders, or at the edges of beaches where they transition into rocky headlands.
- Complex patterns of sediment reworking during storms. It is known from years of research on sand beaches that sand is eroded from the beach and transported offshore during storms or during periods of continuous large swell, then is redeposited on the beach during calmer periods (**Figure 8-5 Bottom**). However, gravel beaches normally respond during storms in a contrary fashion to sand beaches, with coarser gravel transported landward, forming high storm berms (**Figure 8-5 Top**). Oil that penetrates or is buried in the storm berm on gravel beaches can persist for many years. This documented persistence of oil in the upper and supratidal zones on gravel beaches has led to use of berm relocation as a treatment option for residual oil. For example, high-tide berm relocation was conducted 1.5 years after the *Exxon Valdez* spill at 30 gravel beach locations for a total length of 3,000 m (Owens et al. 1991).

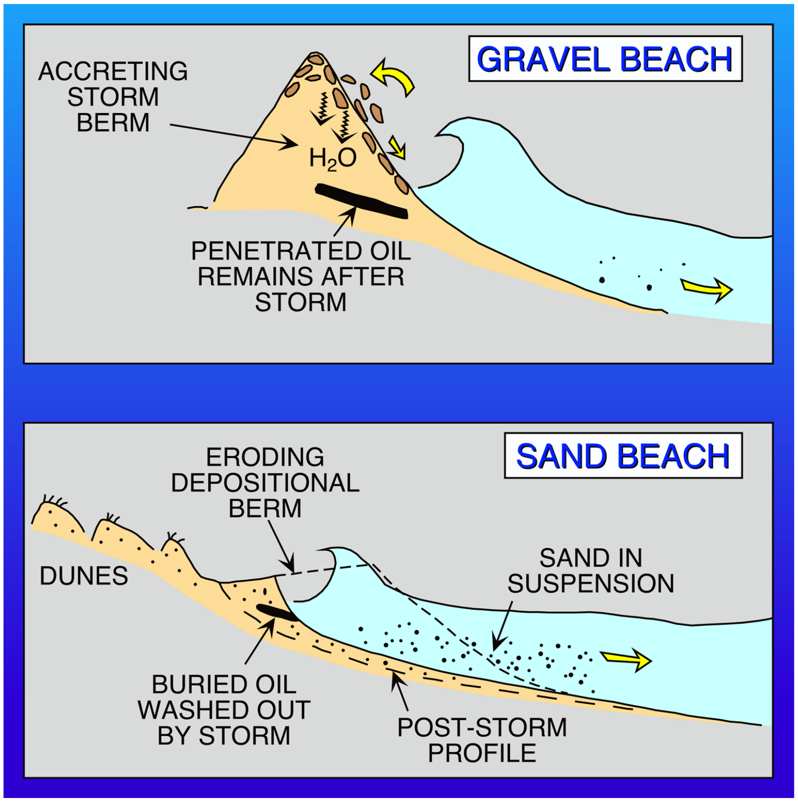

Figure 8-5. Comparison of the impact of storm waves on a sand beach and the storm berm of a gravel beach
Storm waves erode the sand beach, removing any subsurface oil, whereas they either build up or do not change the storm berm of the gravel beach, allowing the subsurface oil to remain in place. Modified from Hayes and Michel (2001).

Oil persistence would be highest on gravel beaches in the Arctic, where there are only a few ice-free months per year. The BIOS field experiment showed that, without shoreline cleanup, 70% of a weathered crude oil was removed by natural processes in the first 3-month, ice-free period. However, the remaining oil formed asphalt pavements that persisted for over 20 years and contained both fresh and weathered oil (Owens et al. 2002).

Burial of oiled sediments by clean sediments poses particularly challenging response issues. Burial can be rapid on sand beaches. Particularly if the oil came ashore during a storm, it could become buried within one to two tidal cycles as constructional waves transport the sand back onto the beach. During the 2015 Refugio spill in Santa Barbara, oil in the upper intertidal zone was buried by clean sand within days (**Figure 8-4, Right**).

Tidal flats are almost always located seaward of other habitat types, and the water table is very shallow, compared to beaches. Thus, spilled oil that covers the flat during low tide tends to be lifted off the flat by the rising tide and deposited on the landward habitat. No studies of the impacts to tidal flats from spills of 500–20,000 bbl of crude oil, condensate, or diesel were identified. However, one case study is informative on the response challenges. During the January 2007 well blowout of 8,500 bbl of condensate in Bayou Perot, Louisiana, over 8 ha of tidal flats were oiled (Locke et al. 2008). The emulsified oil was 2–3 cm thick and did not readily refloat with higher water levels in the marsh. **Figure 8-6** shows the nature of the oil on the muddy tidal flats and the different methods used to remove the oil.

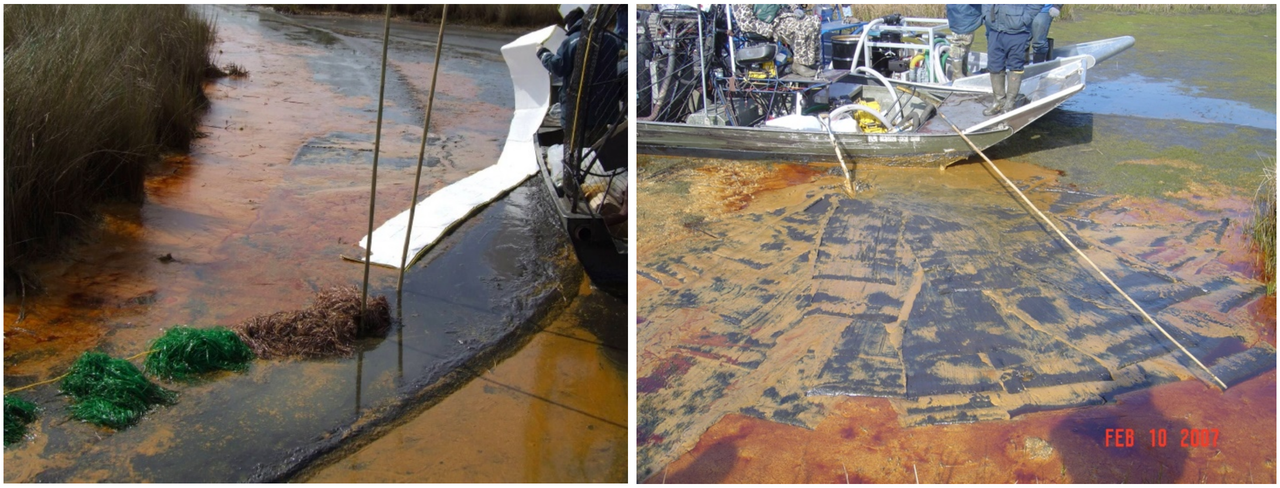

Figure 8-6. Emulsified condensate on muddy tidal flats after the 8,500 bbl 2007 Bayou Perot spill in Louisiana
Left: Testing the use of snare and sweep sorbents. Right: Testing the use of squeegees to remove the surface oil. Note there is no penetration into the water-saturated mud flat sediments.

It took nearly 3 months to complete the shoreline cleanup. No studies were conducted on the ecological impact of this spill.

8.2.2 Impacts of Oil Exposure and Treatment on Beaches and Tidal Flats

Though beaches are frequently oiled during oil spills, few field studies have quantified the impacts and recovery of the biological communities for this habitat type (for the oil types and volumes examined herein). Injury and recovery on sand beaches and tidal flats are mostly quantified by measurement of meiofauna and macroinvertebrates. Studies of gravel beaches include epibiota and attached macroalgae. From a thorough review of the literature on impacts from oil spills and response, eleven oil spills were identified as having sufficient data from field-based studies of spills of 500–20,000 bbl of crude oil or diesel-like oil to quantify impacts and recovery rates (**Table 8-2** and **Figure 8-7**).

Five of the studies were spills of light refined products: diesel, No. 2 fuel oil, and No. 1 fuel oil; six were of spills of crude oil. All the studies were of impacts to beaches (sand, mixed sand and gravel, and gravel); no studies on the impacts to tidal flats were found. Seven of the studies indicated recovery of the beach fauna within 1 year. The longer recovery periods were for crude oil spills that heavily impacted mixed sand and gravel beaches. The longest recovery period of 2–4 years was for the *ARCO Anchorage* spill where heavily oiled sand and gravel beaches inside the harbor of Ediz Hook were aggressively cleaned using mechanical methods. However, Blaylock and Houghton (1989) also noted that the spill site was industrialized and subject to periodic episodes of pollution (new oil contamination of sediments 2 years after the spill), which likely slowed the recovery rate.

The only study that referenced impacts to macroalgae was the *Bahia Paraiso* spill of diesel fuel arctic by Kennicutt et al. (1991a). The authors indicated that, after being oiled, macroalgae were cleaned within a few days and were resilient to the oiling because of the short time of exposure at the lower intertidal zonation of the macroalgae.

Table 8-2. Studies with documented or estimated impacts to and recovery of sand and gravel beach invertebrate communities from spills of crude oil or diesel (500–20,000 bbl)

Oil Spill (Shoreline Oiling, km)	Degree of Oiling (oil volume & type)	Oil Cleanup	Invertebrate group (indicator species) and grain size	Documented Effect/Impacts	Recovery (years)
1972, Long Island Sound, CT[a]	Light to Heavy (1,905 bbl No. 2 fuel oil)	Yes, sorbents	Macroinvertebrate infauna Coarse sand beach	Initial high mortalities of polychaetes, snails, and amphipods in heavily oiled beaches, normal abundance in lightly oiled beaches.	<0.5
2001 T/V *Jessica,* Galapagos Islands, Ecuador[b]	Very Light to Light (2,800 bbl diesel and 2,160 bbl bunker fuel oil)	No	Macroinvertebrate abundance and diversity (snails, anemones, crabs) Boulder beach	No consistent pattern of differences in density of individual species between paired oiled and non-oiled sites at 4 and 11 months post spill.	<0.5
1990 T/V *American Trader*, Huntington Beach, CA (22 km)[c]	Light to Heavy (9,919 bbl North Slope crude)	Yes, manual	Bean clams (*Donax gouldii*) Sand beach	Bean clams: Up to 70% mortality in the upper intertidal zone; overall 24% mortality. Sand crabs (*Emerita*) had large increases in oil uptake that persisted for 4 months.	<0.5
1971 Texaco Refinery, Puget Sound, WA[d]	Light (4,700 bbl diesel)	Yes, straw and sorbents	Macroinvertebrate epifauna and infauna Cobble beach	Study of a cobble beach over 6 months found high mortality (30–100% of epifauna, (barnacles, snails, hermit crabs) and some clams over 6 weeks, and larval recruitment as of 6 months, though diesel persisted in sediments over that period.	<0.5
1999 T/V *Laura d'Amato*, Sydney Harbour, Australia[e]	Light to Heavy (1,750 bbl Murban light crude)	Yes	Amphipods (*Exeodiceros fossor*) Sand beach	Reduced abundance relative to unimpacted beaches.	0.3–>0.9
1975 T/V *Garbis*, Florida Keys, FL[f]	Moderate (crude oil emulsion)	Limited	Macroinvertebrate abundance (amphipods, crabs) Sand beach	Macrofauna were not found in the oiled wrack or in the oil-soaked sand.	<0.5
2010 T/V *Bunga Kelana*, Singapore (35 km)[g]	Light to Heavy (17,250 bbl Bintulu crude)	Yes, large quantities of oil-soaked sand removed	Ghost crabs (*Ocypode ceratophthalmus)* Sand beach	Ghost crabs had 100% mortality but recruits of juveniles started in 3 months to pre-spill densities, with second wave of recruitment at 7 months; however, they avoided the oiled areas.	0.25–>1
1973 Lamma Island, Hong Kong[h]	Heavy (14,000–21,000 bbl heavy marine diesel)	Unknown	Meiofauna community (harpacticoid copepods, nematodes) Sand beach	Drastic quantitative changes of the meiofauna community within the first eight months after the spill.	1.25

Oil Spill (Shoreline Oiling, km)	Degree of Oiling (oil volume & type)	Oil Cleanup	Invertebrate group (indicator species) and grain size	Documented Effect/Impacts	Recovery (years)
1981 T/V *Sefir* Baltic Sea, Sweden (10 km)[i]	Heavy (2,800 bbl No. 1 fuel oil and diesel, leaked from sunken vessel for 6 weeks)	Yes	Macroinvertebrate infauna and epifauna (polychaetes, snails, amphipods, isopods, and ostracods) Sand and pebble beaches	Total macrofauna mortality was 98%, biomass was reduced by 98.5% for both burrowing and surface animals. 1 year: biomass and abundance were only 10% of reference sites.	>2
2015 Refugio Beach, Sand Barbara, CA (214 km)[j]	Very Light to Heavy (500 bbl Monterey crude)	Yes, manual	Macroinvertebrate infauna (talitrid amphipods, *Megalorchestia*) biomass and abundance Sand and gravel beach	One month post spill, abundance was reduced by 85%, biomass by 80%, declining further at 4 months, particularly for adults. 1 year: Recovery was progressing at most sites. 2 years: Abundance, but not biomass, approached baseline due to a lack of larger adults and impaired population size structures and reproductive capacity.	>2
1985 T/V *ARCO Anchorage*, Puget Sound, WA[k] 24 km)	Heavy (5,690 bbl Alaska North Slope crude)	Yes, agitation	Macroinvertebrate infauna biomass, abundance, diversity (bivalves, crustaceans, polychaetes) Sand and gravel beach	Oiling and beach cleanup caused near-complete mortality of the infaunal community.	2–4

[a]EPA (1973);[b]Edgar et al. (2003a); [c]American Trader Trustee Council (2001); [d]Woodin et al. (1972); [e]Jones (2003); [f]Chan (1977); [g]Lim and Yong (2015); [h]Wormald (1976); [i]Linden et al. (1983); [j]Dugan (2018); [k]Blaylock and Houghton (1989)

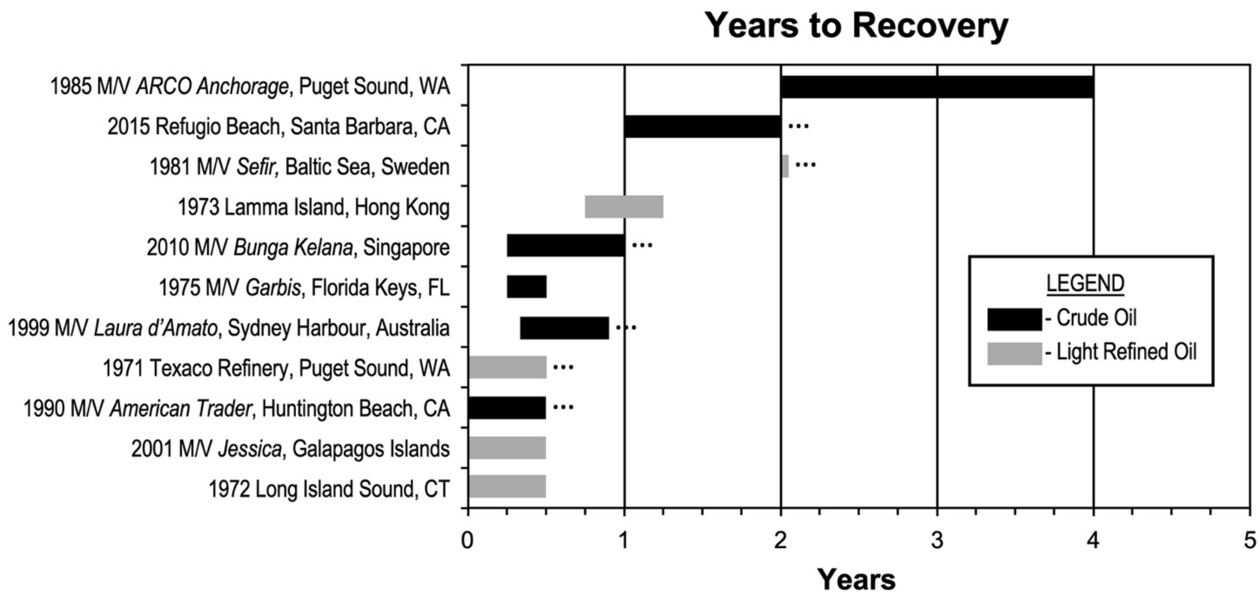

Figure 8-7. Recovery of sand and gravel beach invertebrate communities for the oil spills listed in Table 8-2

Black bars show recovery rates for spills of crude oil; grey bars show recovery rates for spills of light refined oils (diesel, No. 2 fuel oil, home heating oil, No. 1 fuel oil). Dotted lines indicate incomplete recovery at the time of the most recent study

Another important point about the recovery rates of sand and gravel beach macroinvertebrate infauna is that full recovery was not reached for five of the spills by the end of the study period, indicated by dotted lines in **Figure 8-7**. Thus, the actual recovery rate is unknown. Also, it is important to note that the specific definition of recovery varies from case-to-case and by the species or taxonomic groups and metrics studied. In this report, we summarize and synthesize the limited information that is available and acknowledge that recovery in this context does not necessarily equate to full ecological or ecosystem recovery.

Based on the available literature, impacts and recovery of invertebrate communities of beaches and tidal flats can be described in three phases: 1) an impact phase, where the invertebrate community experiences a measurable reduction in abundance and species diversity caused by direct mortality from the oil's toxicity, fouling effects, and disturbances from treatment methods, as well as indirect effects from degradation of habitat; 2) an initial recovery phase, where there is an increase in opportunistic and/or oil-tolerant species when the oil levels in the sediments are generally lower but still high and bioavailable enough to have sublethal effects such as reductions in fertility and growth rates on less tolerant species; and 3) the return of normal species composition, diversity and density indicating the start of full recovery, once the toxicity of any residual oil is reduced to a value below effects levels. Full recovery is achieved when the invertebrate community reaches species composition, diversity, density, and age/size structures comparable with unoiled reference sites (though studies seldom are conducted long enough or rigorously enough to document such full recovery).

The extent of adverse impacts to macroinvertebrates during the impact phase is a function of: 1) the type and amount of oil that strands on the substrate; 2) where the oil strands in the intertidal and supratidal zones (including the degree of oiling of the wrack habitat and community); 3) how deeply the oil penetrates or is buried in the sediment; 4) how quickly the oil is removed (by natural processes or cleanup operations); and 5) the intensity of cleanup operations (which can have large spatial variations). Often, studies are conducted only of the more heavily oiled habitats.

Heavy shoreline oiling by light refined oils often results in high mortality of benthic fauna, up to 100%. However, the low persistence of this type of oil allows for rapid recovery in most cases, particularly if the heavily oiled area is of limited extent. Adjacent, unaffected areas provide sources for rapid recruitment. Heavy shoreline oiling by crude oils can also result in high toxicity, by both chemical toxicity and smothering effects.

The persistence of oil residues in the sediments will also affect recovery rates. As shown in **Table 8-1**, most of the studies showed that, for crude oil spills on beaches, much of the oil was removed during cleanup operations, which would be the start of the recovery period. On tidal flats, passive recovery using sorbents is often the only response option conducted (NOAA 2010).

The final recovery phase after an oil spill is a function of the life histories of the dominant macrofauna, which follows one of two cycles (McLachlan and Brown 2006), as described by Michel et al. (2015).

The first is the bentho-pelagic life cycle in which mature adults release fertilized eggs or larvae into the water column. The larvae drift passively for days to weeks, rarely months, while they develop. Once they reach the last stage of larval development and if they have drifted to appropriate habitat, they metamorphose into juveniles. Similarly, the early life-history stages of many terrestrial species associated with beaches and wrack occur in a location separate from the beach (although some Diptera species use the wrack as their larval habitat, and tiger beetle larvae occur in intertidal sediments). Recruitment and recovery can be rapid for species with a bentho-pelagic life history, as long as there are sources of larvae up-current from the oil-impacted site, and the oiled site has recovered enough to support the new recruits.

The second dominant life-history cycle for macrofauna occurs in species with non-planktonic or non-dispersing eggs or larvae, such as brooding species, which includes many amphipods and isopods in both the supratidal and intertidal zones. Brooding species retain fertilized eggs on their body or in burrows, and juveniles are released directly into the same location where the parental adults reside. If oil impacts to brooding species are substantial, recovery can only start once adults immigrate into the affected area from surrounding locations the following year.

Other factors affecting the rate of macroinvertebrate recovery following an oil spill are reproductive timing and frequency, life span, and vulnerability of species. If a spill severely reduces the annual reproductive cohort prior to spawning, recovery could be delayed until new recruits arrive and mature into adults. For brooder species, such as talitrid amphipods often associated with beach wrack and haustorid amphipods in the intertidal sediments, even the start of recovery can be delayed 1 year.

With respect to life span, many of the dominant sand beach macrofauna are functionally annual species, though some proportion of the spawning population can reproduce a second year (McLachlan and Brown 2006). However, some species can live longer, as compiled in Bejarano et al. (2015): ghost crab can live 2–3 years; lugworm polychaetes can live up to 6 years; ghost shrimp can live 2–4 years; some polychaetes can live for over 2 years.

Bejarano and Michel (2016) synthesized information on the vulnerability of benthic communities of sand beaches and flats to impacts associated with oil spills and concluded vulnerability is influenced by three key factors: 1) traits that facilitate a direct physical pathway of exposure to oil; 2) traits that influence the sensitivity of a species, altering its ability to survive, develop, and reproduce; and 3) traits that influence a species' ability to recover, which are directly liked to demographic parameters and recolonization capabilities. They developed a ranking approach shown in **Table 8-3**.

Table 8-3. Species-specific traits categorized into factors that may influence invertebrate species vulnerability to impacts from oil spill exposure and response activities on sand beaches, organized into three relative vulnerability categories. From Bejarano and Michel (2016)

Vulnerability component	Trait	Low Vulnerability	Moderate Vulnerability	High Vulnerability
Traits that facilitate physical exposure to oil	Habitat preference	Only a few life stages are associated with the substrate (ontogenetic habitat shifts).	Most life stages (e.g., juveniles and adults) are tightly associated with the substrate.	All life stages are tightly associated with the substrate.
	Distribution within habitat	Broad and widespread distribution; different habitat usage across life stages.	Broad but patchy distribution.	Patchy distribution; restricted to areas within and around wrack.
	Burrowing behavior	Deep burrows (>1 m).	Intermediate burrow depths (10 cm to 1 m).	Shallow burrows (<10 cm); wrack associated.
	Mobility, degree of avoidance, swimming capacity[a]	High mobility or swimming capacity (10s of meters); capable of avoiding exposure.	Moderate mobility (meters); moderate ability to avoid exposure.	Low mobility (centimeters).
Traits that influence the sensitivity or a species ability to survive following exposure to oil[b]	Body size; surface area-to-volume ratio	Large body size (>1 cm); small surface to volume ratio.	Medium body size (≥0.1–1 cm); intermediate surface to volume ratio.	Small body size (<0.1 cm); large surface to volume ratio.
	Body type	Hard exoskeleton.	Armored body with integuments; flexible exoskeleton.	Soft body.
	Detoxifying mechanisms; biotransformation machinery	High ability to metabolize hydrocarbons; presence of enzymes with biotransformation capabilities.	Moderate ability to metabolize hydrocarbons.	Limited ability to metabolize hydrocarbons, lack of enzymes with biotransformation capabilities.
Traits that influence a species ability to recover following impacts resulting from oil spills	Occurrence	Abundant species.	Common species	Rare species
	Life span	Short life span (<1 year).	Life span of intermediate duration (1–2 year)	Relatively long-life span (>2 years).
	Age at first reproduction	Early maturity (days to weeks).	Intermediate maturity (weeks to months)	Slow development and late maturity (months to years).
	Reproductive frequency and timing	Multiple reproductive events per year; long reproductive periods.	One reproductive event per year; short reproductive periods; all adults reproduce within a short time period.	One or a few reproductive events per life span; all adults reproduce within a short time period.
	Fecundity	High fecundity (many thousands of offspring).	Intermediate fecundity (hundreds to thousands of offspring).	Low fecundity (tens to low hundreds of offspring).
	Reproductive strategy	Broadcast spawners.	Sperm casting and internal fertilization.	Internal fertilization.
	Development strategy	Planktonic, pelagic larvae.	Lecithotrophic larvae.	Direct development (brooding, lack of larvae stage).

Vulnerability component	Trait	Low Vulnerability	Moderate Vulnerability	High Vulnerability
	Adult's dispersal capacity	High dispersal ability (km).	Intermediate dispersal ability (m to km).	Limited dispersal ability (cm to m); resident populations.
	Recruitment	High recruitment from nearby areas.	Moderate recruitment from nearby areas.	Limited to no recruitment from nearby areas.

[a] This trait is also related to recovery.
[b] Important traits may also include factors that influence the update and depuration of toxic fractions of oil, such as respiration strategy, lipid-body burdens, the presence of specific enzymes, etc. However, some of this information is generally scarce for most marine invertebrates.

Figure 8-8 presents these factors as they contribute to degree of impact and recovery time from exposure to oil following spills on sand beach habitats. Note that **Figure 8-8** does not include additional impacts resulting from treatment operations, which are discussed below.

Cleanup operations on beaches and flats often involve manual removal methods to minimize the amount of sediment removed and damage from vehicular traffic. However, when large amounts of oiled sediments need to be removed quickly, responders have to evaluate the tradeoffs between manual methods and the use of mechanical methods, including backhoes, graders, bulldozers, etc. As part of the NRDA of the 2010 *Deepwater Horizon* oil spill, which included intensive mechanical treatment was conducted on nearly 93 km of 965 km of oiled sand beaches, Michel et al. (2017) developed 'response injury' categories that reflect both intensity and frequency of beach cleanup treatment methods. They used the literature on similar types of disturbances to sand beach communities (foot traffic, vehicular traffic, wrack removal, beach nourishment) to describe the expected impacts and predicated recovery timeframes for different types of sand beach cleanup operations. As for oil spills, the timing, areal extent, degree, and duration of physical disturbances during intensive beach treatments affect invertebrate recovery rates post-treatment.

8.3 Summary and Information Needs for Assessing Impacts to Beaches and Tidal Flats from Spills of 500–20,000 bbl of Crude Oil, Condensate, or Diesel

Every spill is a unique combination of circumstances; however, there is an adequate understanding of how oil interacts with beach and tidal flat sediments and of oil fate and persistence. The spill response community is very aware of the tradeoffs to be considered during selection of appropriate response methods to mitigate oil on beaches and flats.

As is clear in the above discussions, there are few studies that directly document the effects of oiling on invertebrate faunal abundances from median-range spills on beaches. No studies were identified on the impacts of oil on tidal flat invertebrate communities. Existing studies suffer from one or more issues (inadequate replication, weak sampling designs, limited scope in space and time, confounding by conditions unrelated to oiling) that make it difficult to quantify the impacts as related to oil exposure.

There are even fewer studies that have documented full ecological recovery for oiled beaches and tidal flats. Factors controlling impacts and recovery include the type of oil; degree and extent of oiling over space and time; the degree of weathering of the oil that strands onshore; oil persistence and weathering in the sediments and its bioavailability; species-specific sensitivities to oil exposure pathways; and species

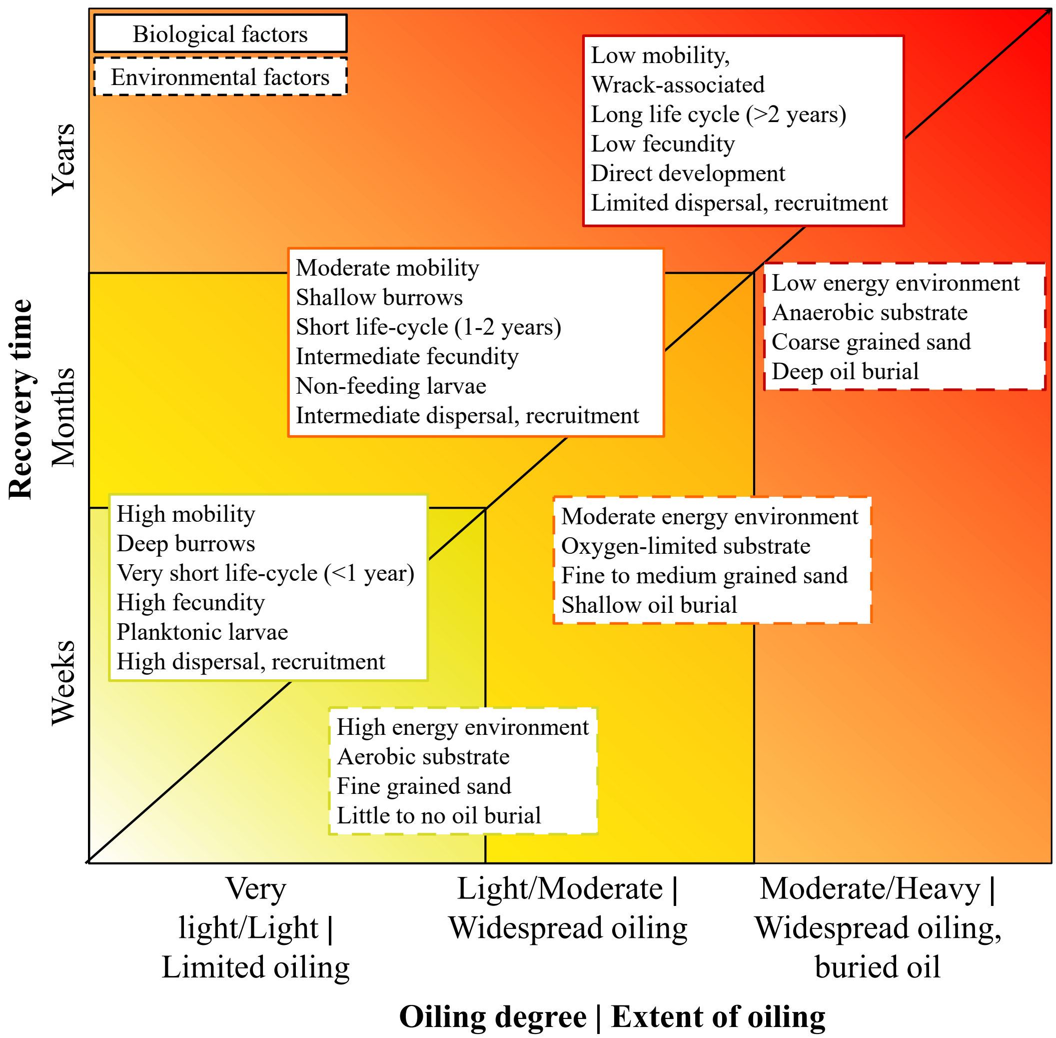

Figure 8-8. Factors that may contribute to the recovery of invertebrate communities in sand beaches following an oil spill

This figure maps out the factors in Table 8-3 that result in short to long recovery times. The anticipated recovery times are a continuum rather than exact times, and they are a function of site-specific physical and biological factors and their continuum. From Bejarano and Michel (2016).

life-history traits, including reproduction and dispersal. The available literature indicates that the invertebrate community on oiled beaches can recover in weeks to 1–2 years under most spill conditions. However, recovery can take up to 4 years where there is heavy oiling, intensive treatment, or other sources of contamination that limit recovery.

To better quantify the potential impact and recovery of beaches and tidal flats following an oil spill, it is recommended that rigorous field study designs for each habitat be prepared and ready for implementation in the event of a spill. Peterson et al. (2001) identified 18 study design decisions that affected the outcomes of shoreline impact studies conducted following the *Exxon Valdez* spill. These are excellent considerations to use in design of any study assessing the effects of oiling on beaches and flats.

9 Rocky Shores

9.1 Habitat Description, Communities, and Ecological Functions and Services

Rocky shores are typically classified in terms of tidal zone and their exposure to wave energy (Little and Kitching 1996). The rocky shore habitat can exhibit complex morphology, featuring variable slopes, overhangs, tide pools, crevices, and varying surface textures. Depending on geological formations and oceanographic processes, rocky shore habitat may range from vertical cliffs to wide platforms with abundant tide pools (Petersen et al. 2019). Rocky shore substrates may be relatively impermeable with homogenous topography or highly porous with large interstitial spaces. These habitats provide a range of ecological functions and services including trophic support for both marine and avian species, fisheries production, coastal protection, and recreation (IPIECA 2005; Little and Kitching 1996).

The rocky shore community comprises species that are well-adapted to a stressful and dynamic environment. Rocky shores exhibit a distinct vertical habitat zonation based on tidal zones (**Figure 9-1**). Each zone supports different communities of flora and fauna based on conditions that affect desiccation, temperature, salinity, wave energy, access to food, and predation (Foster et al. 1988; Little and Kitching 1996). In general, lower zone communities are more diverse and productive than those of the upper shore and have more soft-bodied organisms (e.g., sea stars, sea slugs) and macroalgae (e.g., rockweed, kelp). The upper shore supports organisms that can adhere to the rocky substrate and have adapted to long periods exposed to air, such as filter-feeding barnacles and limpets (de la Huz et al. 2011). Tide pools, caves, crevices, and other sheltered microhabitats in the upper zones support a diverse community typical of the lower tidal zones, as they may serve as a wet refuge from the harsh conditions that characterize the upper zones.

Macroalgae are the dominant flora of rocky shore communities and serve as a major source of organic material to marine and coastal fauna (Little and Kitching 1996). During high tides, fish and other marine species feed on the algae; at low tides, when algae are exposed to the air, they serve as a food resource for birds and mammals. Algae also provide reproductive habitat for rocky shore invertebrates, which release eggs and larvae into coastal waters that are consumed by juvenile fish and other species (de Putron and Ryland 1998).

Interactions between flora and fauna strongly influence the structure of the rocky shore community. In particular, grazers such as limpets and snails dominate the rocky shore fauna and typically control the algal population. In the absence of these grazers, algal populations may explode and overwhelm the habitat, whereas an overabundance of grazers can decimate algae. Limpets and sea stars, in particular, are considered keystone species in rocky habitats (Crump et al. 1998; Paine 1966). Disturbances to grazing species can therefore dramatically impact the entire rocky shore community (Little and Kitching 1996).

9.2 Oil Behavior and Persistence in Rocky Shore Habitats

Oil persistence in rocky shore habitats is strongly influenced by habitat characteristics such as shoreline exposure, slope, habitat zonation, and wave action (Petersen et al. 2019). The Environmental Sensitivity Index (ESI) shoreline ranking system is used to categorize habitats based on the likely persistence of spilled oil (Petersen et al. 2019). ESI rankings range from 1A to 10F, where 1 represents a shoreline where oil is least likely to persist, and 10 represents a shoreline where oil is most likely to persist and affect sensitive habitats. Petersen et al. (2019) indicate four major factors contributing to ESI rankings:

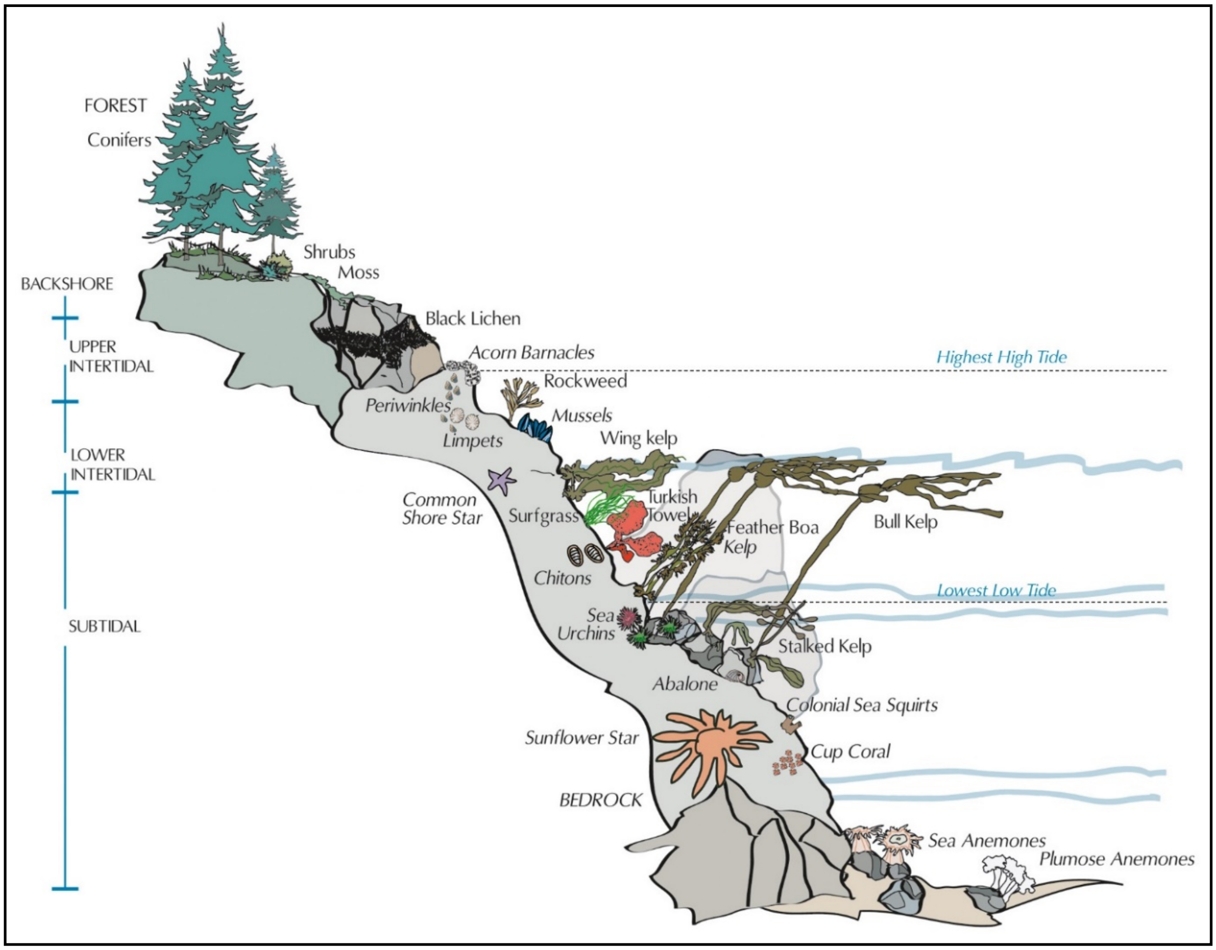

Figure 9-1. Typical rocky shore zonation and dominant macrofauna for the Pacific Northwest region

Distribution of representative rocky shore flora and fauna across tidal zones. Species are distributed within the intertidal zone based on their ability to withstand exposure to air and sun during low tide. From Stewardship Centre of British Columbia (2004).

1) Relative exposure to wave and tidal energy;
2) Shoreline slope;
3) Substrate type; and
4) Biological productivity and sensitivity.

Rocky shores are assigned varying ESI rankings depending on the above characteristics and range from 1A to 8D (**Table 9-1**). These ESI rankings support the understanding that oil may have high or very low persistence on rocky shore communities, depending on specific habitat characteristics.

Shoreline exposure, slope, and substrate type strongly influence how oil reaches and interacts with the shore as well as the degree to which waves and tides naturally "clean" contaminated shores. The nature of both shoreline oil contamination and cleaning are dictated by interactions with wave energy. Oil that may otherwise reach exposed rocky shores is often held offshore by wave action and any stranded oil is also removed by wave action, whereas oil on sheltered rocky shores may persist for long periods due to a lack of wave action (Lopes et al. 1997; Petersen et al. 2019). For example, following the *Universe Leader* spill, oil remained trapped for 3 months in sheltered coves with limited wave action (Cullinane et al. 1975). Conversely, shoreline oiling was minimal during the *Jessica* spill due to the area's exposed shorelines and substantial wave energy (Edgar et al. 2003a; Gelin et al. 2003; Lougheed et al. 2002).

Table 9-1. ESI rankings for rocky shores, as defined in Petersen et al. (2019)

ESI Ranking	Rocky Shore Type	Oil Behavior and Persistence
1A	Exposed, rocky shores	Oil is mostly kept offshore by wave reflection; impermeable so oil remains on the rock surface; persistent oil is usually as a band at the high-tide or splash zones.
1C	Exposed, rocky cliffs with boulder talus base	
2A	Exposed, wave-cut platforms	Similar to above, except that there can be some sediments on the platform and at the high-tide zone where oil can persist for weeks or months.
8A	Sheltered, impermeable rocky shores	Oil adheres to rough rock surfaces and persists due to low wave energy.
8B	Sheltered, permeable rocky shores	Oil can coat the surface and become trapped below a veneer of coarse material, persisting for long periods of time.
8D	Sheltered, rocky rubble shores	

Patterns of oil persistence vary across rocky shores of differing slopes as a result of wave interactions. On steeper rocky shores, strong wave reflection patterns tend to keep the oil offshore, with any stranded oil forming a band along the high-tide line or splash zone (Petersen et al. 2019).

The substrate comprising rocky shores varies in its permeability and topography, with oil tending to accumulate in the cracks and crevices of more contoured rocky shores. Oil trapped in these interstitial areas persists when wave action is limited. Impermeable, relatively flat shores keep oil at the surface where it is more quickly removed by wave action (Petersen et al. 2019). Tide pools may accumulate oil during low tides. On exposed shorelines, oil in tide pools is often quickly removed by direct wave action (Petersen et al. 2019).

At the habitat level, oil persistence varies according to tidal zonation. Because oil is more likely to adhere to dry rock substrate than wet rock or algae, the drier upper tidal zones are more likely to retain oil than are the lower zones. In addition, the onset of a high tide may resuspend oil that has adhered to the substrate, allowing it to spread to other parts of the shore (Little et al. 1990). Because oil tends to pool in "natural sinks," oil may also become trapped at the lower tidal zones of a more contoured, cobble shore as the tide recedes (Little et al. 1990). **Figure 9-2** shows a rocky shore with patches of cobbles, with variable degrees of oil persistence—the weeds and rocks of the upper shore are heavily oiled, as is the lower shore due to its contoured topography; the smooth middle shore is mostly free of oil, apart from crevices and tide pools (Little et al. 1990).

Weather may also play a role in the behavior and persistence of oil on rocky shores. Following the *Nella Dan* spill, oil was immediately blown onshore, but a change in weather about one day later blew the oil out to sea, mitigating shoreline oiling (Smith and Simpson 1995). In the case of the *El Omar* spill, weather patterns had a deleterious effect, as changes in wind direction following the spill caused previously unaffected shores to become oiled (Little et al. 1990).

9.3 Impacts of Oil Exposure and Treatment on Rocky Shores

This section reports findings on the impacts to and recovery of rocky shores following spills of 500–20,000 bbl of crude oil, condensate, or diesel. Five such spills of crude oil, three spills of diesel, and one spill of both diesel and bunker fuel were identified in the literature as having data on rocky shore impacts (**Table 9-2**). Of the studies identified, several lacked concrete quantitative data. As such, qualitative data were used to describe the impacts. The persistence of oil in rocky shore habitats plays a major role in the degree to which biota are impacted by oil spills (Chia 1971). Studies of these spills found that factors influencing oil persistence had a strong effect on the extent of biological impacts. Where treatment methods were employed to remove oil or oiled substrate, the treatment itself often adversely affected some community components.

Figure 9-2. Oil behavior on a rocky shore during the *El Omar* oil spill, Wales
Oil from the spill adhered to the upper and lower rocky shore (indicated by the black bands at the top and bottom of the photograph) due to tidal zonation and variable substrate, while the smooth middle shore remained mostly clean except in tide pools and crevices. From Little et al. (1990).

The definition of recovery for rocky shore communities varies in the literature but generally focuses on recovery to pre-spill or reference abundance, density, or diversity. The nature and duration of impacts to rocky shore communities are described in **Table 9-2** and plotted in **Figure 9-3**.

Wave and tidal energy have been major determinants of patterns of oil exposure and effects for several spills, causing dramatic variation in the extent of impact across shorelines and tidal zones (Chia 1971; Cullinane et al. 1975; Lopes et al. 1997). High wave action following the *Universe Leader* spill was influential in limiting the persistence and impact of the oil. Along sheltered shorelines affected by the spill, however, a lack of natural cleaning due to low wave action allowed oil to remain trapped in coves up to 3 months post spill (Cullinane et al. 1975). In these sheltered areas, researchers observed stress on rocky shore flora and fauna. Toothed wrack was the most heavily impacted species of macroalgae, and large amounts were observed cast up from their anchors onto the shore. Two months post spill, saddle oysters and limpets detached from their shells due to heavy oiling (Cullinane et al. 1975).

Impacts to rocky shores from the Tebar V refinery spill were minimal due to heavy wave energy and exposed shorelines (Lopes et al. 1997). An existing monitoring program of the Sao Sebastio Channel's rocky shore communities allowed for a before-after control-impact study following the spill. One year post spill, researchers observed no measurable difference in the percent cover of both mussels and barnacles on rocky shores compared to both pre-spill levels and control (not oiled) sites (Lopes et al. 1997). The *Jessica* spill occurred near exposed shorelines, reducing the persisting and potential impacts to the rocky shore community from the oil (Edgar et al. 2003a; Gelin et al. 2003). Ocean currents also mitigated impacts from the spill by moving oil offshore. Only one survey site was sufficiently oiled such

Table 9-2. Studies with documented or estimated impacts to and recovery of rocky shore communities from spills of crude oil, diesel, and bunker fuel (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Oil Cleanup	Indicator Species	Documented Effect/Impacts	Recovery (years)
2001 T/V *Jessica*, Galapagos Islands, Ecuador[a]	2,800 bbl diesel and 2,160 bbl bunker fuel	No	Macrofauna community composition and abundance, macroalgae percent cover	Little impact due to wave and tidal energy limiting shoreline oiling. Several studies in the year after the spill showed no measurable impacts to rocky shore flora or fauna.	0
1974 T/V *Universe Leader*, Bantry Bay, Ireland[b]	15,500 bbl Kuwait crude	Yes	Macroalgae and macrofauna abundance	Sheltered shorelines kept oil onshore up to 3 months post spill, and oiled invertebrates were observed detached from their shells. Greatest impacts to macroalgae were due to treatment with detergents that caused bleaching. Some macroalgae species had recovered 2 months post spill.	>0.2
1988 T/V *El Omar*, Milford Haven, Wales[c]	670 bbl light Iranian crude	Yes	Macroalgae and macrofauna abundance and community composition	Loss of grazers at some sites caused green flush of macroalgae; prolonged oiling deteriorated macroalgae mucoid coating but began to recover six months post spill.	>0.5
1994 Tebar V Refinery, Sao Paulo, Brazil[d]	19,000 bbl crude	Yes	Macrofauna percent cover	Wave energy and evaporation of aromatic hydrocarbons limited impact from oil. 1-year post-spill percent cover of mussels and barnacles were similar to control sites and pre-spill levels.	<1
1973 Lamma Island, Hong Kong[e]	14,000–21,000 bbl heavy marine diesel	No	Macrofauna abundance	Heavy contamination in tide pools caused acute mortality in several gastropods. Lipped periwinkle and blotched nerite were absent from monitoring sites for at least 13 months post spill.	>1
2015 Refugio Beach, CA[f]	500 bbl Monterey crude	Yes	Macrofauna abundance	Measurable decrease in percent cover of owl limpets, mussels, and barnacles due to oiling. Little recovery of species observed 1 year post spill.	>1
1987 T/V *Nella Dan*, Macquarie Island, Australia[g]	1,690 bbl diesel	No	Macrofauna community composition	Greatest impact in the upper shore and higher portions of subtidal zone. Depletion of isopod community that typically dominates kelp holdfast community allowed opportunistic polychaetes to dominate. Little evidence of recovery 6 years post spill.	>6

Oil Spill	Oil Volume and Type	Oil Cleanup	Indicator Species	Documented Effect/Impacts	Recovery (years)
1997 Torch/Platform Irene, Santa Barbara, CA[h]	163 bbl crude*	No	Macrofauna abundance	Estimated 10–15% decline in black abalone abundance. no recovery evident 1 year post spill. Impacts from the oil spill expected to exacerbate the decline of black abalone and limit chances for recovery in the foreseeable future (~10 years).	>1
1971 Texaco March Point Refinery, Guemes Island, WA[i]	4,700 bbl diesel	Yes	Macrofauna mortality	Mortality of invertebrates was greatest at higher intertidal regions and ranged from 30–100%.	Unknown

[a]Edgar et al. (2003a); Gelin et al. (2003); Lougheed et al. (2002); [b]Cullinane et al. (1975); [c]Little et al. (1990); Little and Little (1991); [d]Lopes et al. (1997); [e]Stirling (1977); [f]Raimondi et al. (2019); [g]Simpson et al. (1995); Smith and Simpson (1995); [h]Torch/Platform Irene Oil Spill Natural Resource Trustees (2005); [i]Chia (1971)

*Although this spill does not meet the 500–20,000 bbl threshold, it is included here due to the limited number of relevant studies.

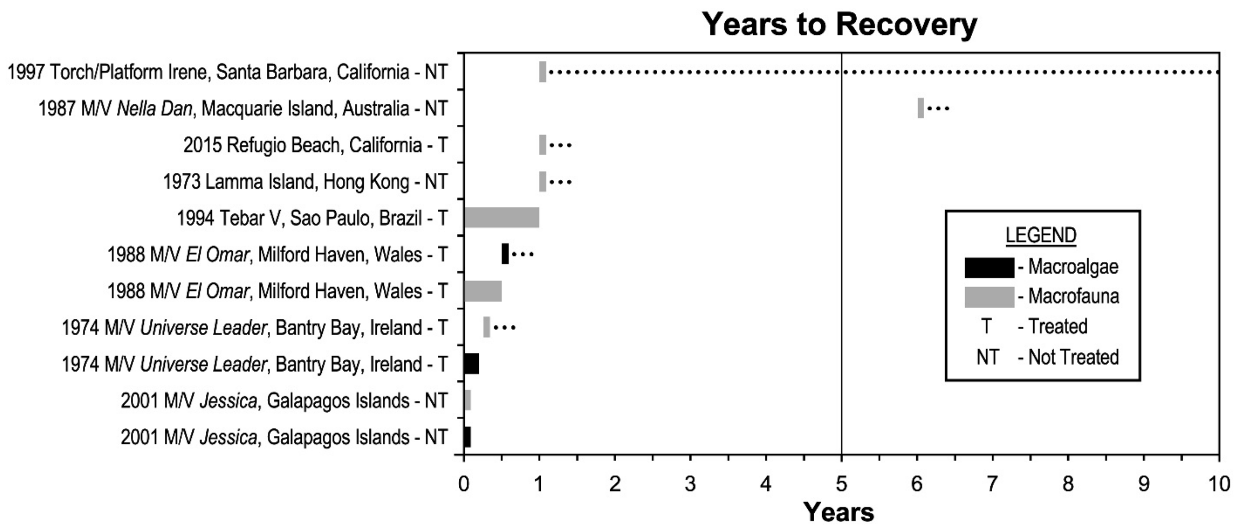

Figure 9-3. Recovery of rocky shore communities following the oil spills listed in Table 9-2
Dotted lines indicate incomplete recovery at the time of the most recent study.

that it was considered moderately oiled, while all other locations were either classified as light or very light (Lougheed et al. 2002). Further, weather patterns played a major role in minimizing the impacts from the spill, as warm weather surrounding the spill evaporated much of the oil (Edgar et al. 2003a). One month after the spill, there was very little change in macroalgae percent cover and macroinvertebrate abundance (Edgar et al. 2003a). In another study conducted 4–11 months after the spill, impacts were not observed on macroalgae or intertidal invertebrate communities were observed (Gelin et al. 2003). Similarly, warm weather in the days following the Tebar V refinery spill also increased the rate at which aromatic hydrocarbons in oil were evaporated, ultimately limiting the oil's toxicological effects (Lopes et al. 1997).

As described in the previous section, weather patterns surrounding the *El Omar* spill moved oil from contaminated shores to those initially unaffected by the spill, resulting in a more expansive impact (Little et al. 1990). Impacts from the spill were further exacerbated when oil was resuspended during high spring tides, reaching the oiled upper shore. Oil pooled in crevices, gullies, and tide pools, where gastropods were subjected to the non-specific narcotic effects of oil (Little et al. 1990; Little and Little 1991), which include reduced feeding efficiency and non-responsiveness. At some sites where oil persisted, a "green flush" of ephemeral algae was present due to the loss of grazing invertebrates (Little and Little 1991). Six months after the spill, most communities had recovered; however, the invertebrate community had visibly changed at some sites due to reductions in barnacle abundance (Little et al. 1990; Little and Little 1991).

Microhabitats such as tide pools and crevices that often characterize rocky shores may experience considerable impacts from oil spills due to their proclivity to retain oil. For example, during the Lamma Island spill in Hong Kong, tide pools located in a relatively clean rocky shore remained contaminated, resulting in acute gastropod mortality (Stirling 1977).

Ocean currents, tidal range, and wave action were influential in the spread of oil along the southern California coast during the Refugio Beach spill, reaching shorelines over 100 miles from the release site (Raimondi et al. 2019). Immediately after the release, approximately 20% of biota on oiled shorelines were coated in oil. Owl limpets were heavily impacted, demonstrated by a reduction in their percent cover to nearly 0%; mussels and barnacles were also severely impacted, with respective percent covers at oiled sites approximately 30% and 50% of levels at control (not oiled) sites (Raimondi et al. 2019). One year later, little recovery of species cover was observed, and 10% of biota were still oiled (Raimondi et al. 2019).

In addition to wave energy, the extent and duration of oil impacts from the *Jessica* spill were largely dictated by tidal zonation, where higher tidal zones were the most contaminated portions of the shore (Gelin et al. 2003). In one area, invertebrate abundance in the high intertidal zone of an impacted site was half that of a reference site, while abundance at the mid-tide zone was greater at the impacted site than at the reference site (Gelin et al. 2003). Similarly, invertebrates located at higher intertidal regions during the Texaco Refinery March Point diesel spill were more severely damaged than those in lower zones, likely because the upper shore invertebrates were coated with oil for a longer period (Chia 1971). The species most impacted by extended oil contact included brittle stars, polychaetes, hermit crabs, and limpets. For these species, mortality ranged from 30% to 100% (Chia 1971).

Following the *Nella Dan* spill in Australia, Simpson et al. (1995) observed the most evidence of impacts in the lower intertidal and upper portions of the subtidal zones. In this case, kelp holdfast communities were dramatically impacted, likely due to substantial PAH residues in holdfast sediments. As a result, opportunistic polychaetes took over holdfasts in oiled sites usually occupied by isopod crustaceans (Simpson et al. 1995). Six years later, there was little evidence of recovery, with traces of diesel remaining in kelp holdfasts at oiled sites and the community still dominated by polychaete worms (Smith and Simpson 1995). Some recovery was evident in terms of holdfast community structure, however, as abundances of sensitive species increased and those of polychaetes decreased (Smith and Simpson 1995). Because of the limited reproductive season, low species fecundities, and fewer larval stages present for many species in polar regions, rocky shore community recovery from oil spills may be slower at high latitudes than in more temperate regions. In polar regions, recovery relies more on immigration of fauna from sites unaffected by oil than on recruitment of pelagic larvae (Smith and Simpson 1995).

Species' sensitivity to oiling is also a major determinant of the degree of impact to the rocky shore community. Limpets, littorinid snails, and other grazing mollusks are often sensitive to oil. These organisms may be impacted by oil's toxicity or narcotic effect, which causes a loss of grip to the rock. As these organisms fall off the substrate, they become more susceptible to predation or may die from desiccation (Little and Kitching 1996). Following the Lamma Island spill, Stirling (1977) found that bivalve mollusks, dog whelk, and the lipped periwinkle were most sensitive to diesel, while other gastropods such as the tropical periwinkle were nearly unaffected. In particular, the lipped periwinkle and blotched nerite demonstrated the largest declines in abundance and were absent from monitoring sites for at least 13 months after the spill (Stirling 1977).

Most species of macroalgae are relatively resistant to the adverse effects of oiling due to a mucoid coating that protects the algae from oil's toxic effects (Cimberg et al. 1973). Following the *El Omar* spill, however, prolonged oiling of macroalgae deteriorated their mucoid coating. Six months post spill, several macroalgae species had lost their mucoid coating and were covered by epiphytic green algae, which are usually unable to attach due to the coating's slippery nature (Little et al. 1990; Little and Little 1991). However, new growth of healthy mucoid tissue was observed on algae fronds, indicating the onset of recovery (Little et al. 1990).

Species already in decline due to factors such as disease are particularly susceptible to impacts from oil spills. At the time of the Torch/Platform Irene oil spill, black abalone populations were already stressed due to withering foot syndrome (Torch/Platform Irene Oil Spill Natural Resource Trustees 2005). Oil from the spill coated black abalone in several sites, causing an estimated 10–15% loss to the population. One year after the spill, no juvenile recruitment was observed, and Torch/Platform Irene Oil Spill Natural

Resource Trustees (2005) anticipated the spill would exacerbate the decline of black abalone and hinder the species' chances for recovery.

Particularly on exposed shorelines, wave and tidal action often rapidly remove oil. When oil is not quickly removed by natural processes, anthropogenic treatment methods may be applied. Cleaning methods, however, can cause more harm to rocky shore communities than the oil itself (Lopes et al. 1997). The cleanup methods applied in the days after the *El Omar* spill were responsible for many of the effects on rocky shore habitats and their communities (Little et al. 1990). In areas with heavily oiled cobbles and boulders, bulldozers were used to remove the substrate, also removing the attached macroalgae. Six months later, these impacts were still evident based on the lack of macroalgae and other life on the shore (Little et al. 1990). Along these bulldozed areas, as with the lower shore, a green flush of algae was evident due to the associated removal of grazing invertebrates (Little et al. 1990). Similarly, detergents applied to oil at sea and brought to shore during the *Universe Leader* oil spill were responsible for much of the impacts to macroalgae in Bantry Bay, including bleached tips of several algae species (Cullinane et al. 1975). Because detergents were not applied directly on the shore, impacts were most evident about 3–4 weeks post spill. About 2 months after the spill, many macroalgae species appeared healthy, but toothed wrack and some other heavily impacted species remained damaged (Cullinane et al. 1975).

9.4 Summary and Information Needs for Assessing Impacts to Rocky Shores

The behavior and subsequent biological impacts of oil on rocky shores are dependent on the unique circumstances of any spill. However, the literature describes clear trends and patterns of how oil interacts with rocky shores. Oil may persist on rocky shore habitats and associated fauna from days to years depending on several key factors, including wave and tidal energy, weather patterns, ocean currents, and substrate morphology. Ultimately, the degree of biological impact is determined by these abiotic factors as well as by the sensitivities of exposed species. The persistence and degree of impact from an oil release may vary along a single shoreline, as oil accumulates in the cracks, crevices, and tide pools of rocky shores. This phenomenon was observed during the Lamma Island spill, as tide pools on relatively clean shorelines were heavily contaminated. Further, the vastly different degrees of oiling along the tidal gradient cause varying levels of impact to rocky shore communities, as in the *Nella Dan* and Texaco Refinery March spills.

Despite trends in oil persistence and resulting impacts, the timeline of rocky shore community recovery is not as well understood. Definitions of recovery differ across the literature, and few studies document full community recovery. In some cases, recovery may be defined as recolonization to reference levels of abundance, while other studies may declare recovery a factor of community composition. Many of the studies reviewed for this synthesis documented findings qualitatively rather than quantitatively, posing a challenge to fully understanding impacts and recovery. Conversely, some oil spill studies (e.g., Lopes et al. 1997) have been aided by pre-existing habitat monitoring at the oiled sites that allowed for comparison to pre-spill conditions. These baseline studies are critical for understanding the extent of impacts from an oil spill.

A better understanding of the impacts of oil spills on rocky shores would be supported by: (1) additional baseline studies of rocky shore communities; (2) studies that report quantitative results; and (3) studies that are continued until baseline or reference conditions (recovery) are achieved.

10 Kelp

10.1 Habitat Description, Communities, and Ecological Functions and Services

Kelp are structurally and functionally diverse marine algae that provide physical substrate and habitat for kelp forest communities and act as “physical ecosystem engineers” (Jones et al. 1997). The basic structural units of kelp are: the holdfast, a root-like mass that anchors the thallus to the sea floor; the stipe, extending vertically from the holdfast and providing a support framework; and the fronds, leaf- or blade-like attachments extending from the stipe which are the sites of nutrient uptake and photosynthetic activity (**Figure 10-1 Left**). Many kelp species have pneumatocysts, or gas-filled bladders, usually located at the base of fronds near the stipe which provide the necessary buoyancy for kelp to maintain an upright position in the water column (**Figure 10-1 Right**).

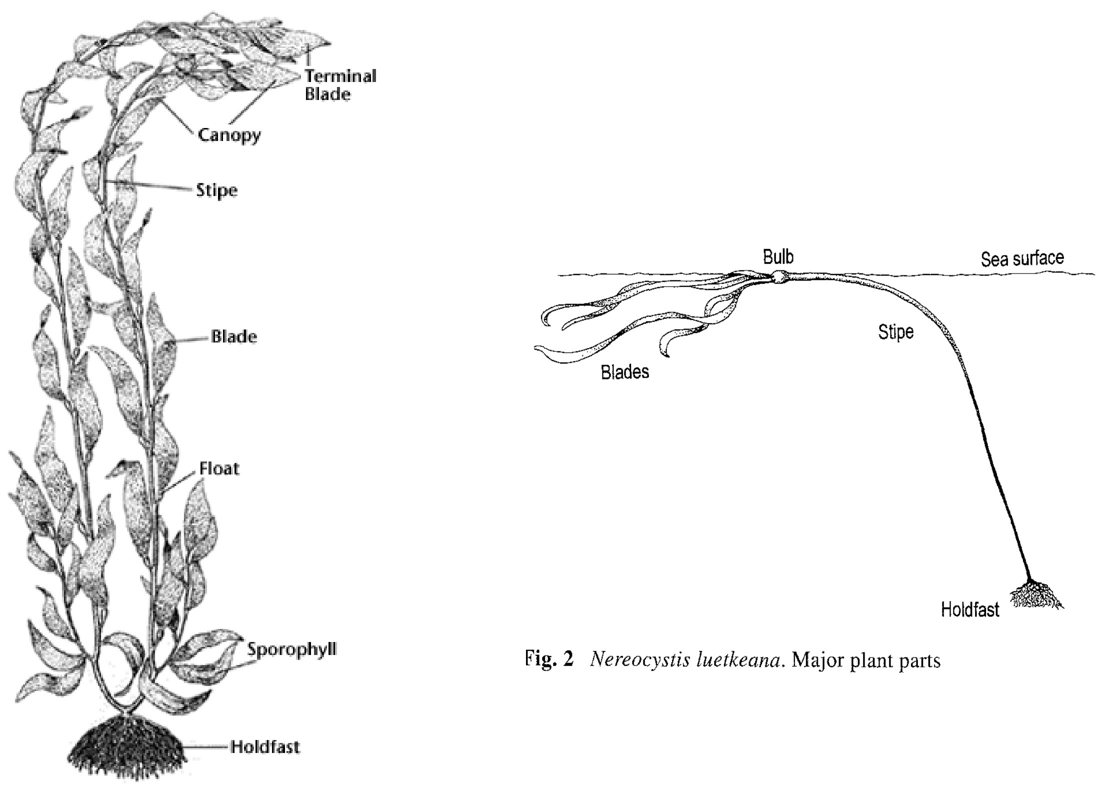

Figure 10-1. (Left) Giant kelp (*Macrocystis pyrifera*) from Monterey Bay Aquarium (2020). (Right) Bull kelp (*Nereocystis luetkeana*) diagram from Antrim et al. (1995)

Kelp forests occur along 25% of the world’s coastlines (**Figure 10-2**). They are characterized by extremely high rates of primary productivity, comprise the base of a complex food web, and form a three-dimensional structure that provides physical shelter for many inhabitants (Graham et al. 2003). Kelp thrive in cool to cold (5°–20° C), shallow, nutrient-rich, medium-energy waters worldwide in temperate to polar latitudes. In the U.S., kelp forests grow predominantly along rocky coastlines in depths of 2–30 m on the Pacific Coast, from Alaska to Baja California and on the Atlantic Coast in New England.

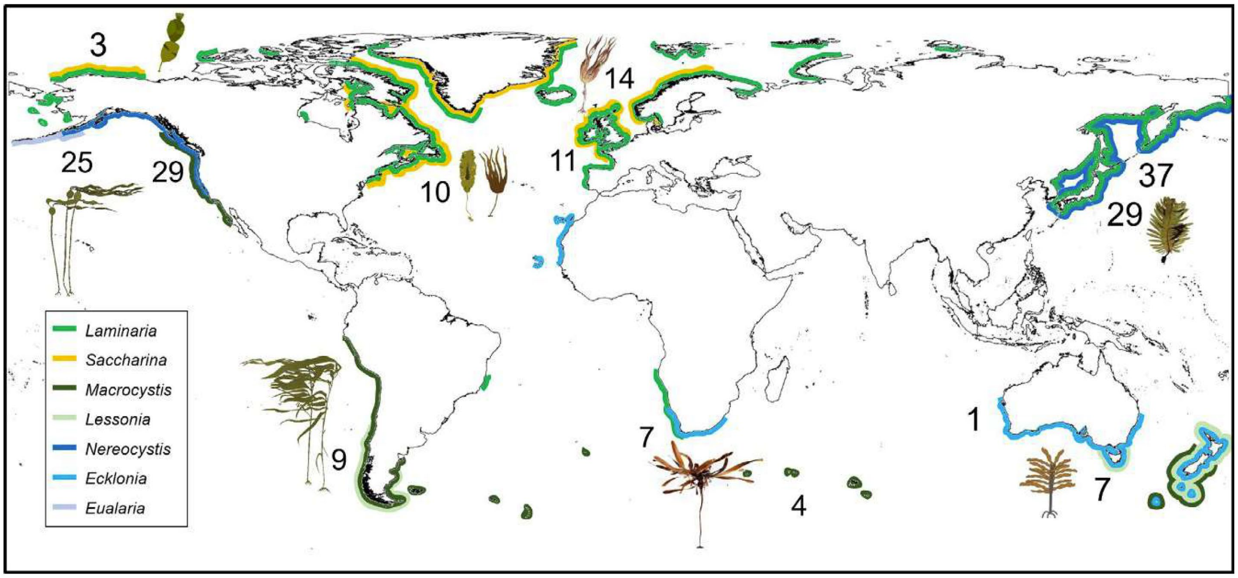

Figure 10-2. Distribution of laminarian kelps globally
Colored lines and numbers indicate the main distribution of major kelp genera and numbers of kelp species, respectively. From Wernberg et al. (2019).

Because kelp attaches to the seafloor, but relies on sunlight to generate food and energy, kelp forests are always coastal and require shallow, relatively clear water. Tiered like a terrestrial rainforest with a canopy and several layers below, kelp forests are major primary producers in the ecosystem (Dean et al. 1996). Kelp recruits most successfully in regions of upwelling, regions where the ocean layers overturn, bringing cool, nutrient-rich bottom waters to the surface.

Kelp forests provide valuable resources, habitats, and services for coastal communities. Kelp supplies food and critical habitat for many commercially and ecologically important animals, including fish (e.g., salmon, herring, white seabass, rockfish) and invertebrates (e.g., crabs, sea stars, and abalone), (Dean et al. 1996; Peckol et al. 1990). Marine mammals (sea lions, seals, sea otters, river otters, whales) and marine birds may feed in or seek refuge from predators in kelp forests. Marine life may retreat to kelp during storms or high-energy regimes as kelp slows currents and reduces wave energy (Jones et al. 1997). Sea otters prey on urchins and thus play a critical role in controlling the numbers of kelp grazers. The array of habitats on the kelp itself, including holdfasts and the surface mats of kelp fronds, supports thousands of invertebrate individuals, including polychaetes, amphipods, decapods, and ophiuroids. In locations where kelp is a keystone species, such as in the inshore marine environment of California, they influence the species abundance and diversity of all marine animals in the kelp community (CDFG and USFWS 1999).

10.2 Oil Behavior and Impacts in Kelp

The five median-range spills of crude oil or diesel-type oils identified for this literature review with any documented observations of oiling of kelp communities, independent of any determined effects, are summarized in **Table 10-1**.

Table 10-1. Studies with information on oil behavior and persistence of crude oil or diesel spills (500–20,000 bbl) in kelp communities

Oil Spill (Reference)	Oil Volume and Type	Kelp and Invertebrate Communities	Documented Effects/Impacts	Persistence / Recovery (years)
1992 UNOCAL tank farm, Avila Beach, CA[a]	600 bbl San Joaquin Valley crude	Giant kelp (*Macrocystis pyrifera*)	None observed.	0
1989 T/V *World Prodigy*, Narragansett Bay, RI[b]	6,900 bbl home heating oil	Sugar kelp (*Saccharina latissma*) Oarweed (*Laminaria digitata*)	There was no evidence of any detrimental effects to oil-exposed kelp; there was no observed necrotic or bleached tissue on any kelps in an oiled cove. Growth rates of both species were within the range previous years' data.	0
1981 BIOS Field Experiment, Baffin Island, Canada (0.4 km)[c]	94 bbl aged Venezuelan Lagomedio crude	*Laminaria spp.*	There was no evidence of reproductive failure in *Laminaria* spp. 2 years after the oil releases, but any such effects in 1981 or 1982 would not have been detected in the study.	Unknown
1985 T/V *ARCO Anchorage*, Port Angeles, WA[d]	5,690 bbl Alaska North Slope crude	Bull (*Nereocystis luetkeana*) and giant kelp Sea urchins (*Strongylocentro-tus* spp.)	Bull and giant kelp beds were impacted. Sheens and patches of emulsified oil stranded in kelp bed canopies, accumulations of weathered oil on stipes. Visible, strongly adherent oil in offshore kelp. 50–60% of kelp stipes were oiled at most sampled sites from the shoreline out to 30-foot depth. Sea urchins observed ingesting clean kelp fragments but not oiled kelp fragments and were not contaminated.	Emulsified oil persisted in kelp beds for 25 days. <1 year to recovery
1991 F/V *Tenyo Maru*, Cape Flattery, WA[e]	6,500 bbl intermediate fuel oil, 2,166 bbl diesel, and unknown amounts of lube oil and fish oil	Bull kelp	Oil patties were found in the dense bull kelp forests that fringed the coastline, and kelp stipes appeared to be bleached and dying in the areas most heavily impacted.	Unknown; no long-term data
2015 Refugio Beach, Santa Barbara, CA[f]	500 bbl Monterey crude	Giant kelp	Submerged oil observed at or on the sea floor in kelp beds during the first 7 days post release. 'Tumble tar' (tar balls mixed with loose algae) between kelp areas in sandy bottom depressions between reefs and caught in low energy sand channels and blotches of sheen trapped among the kelp at the surface. No oil was observed in kelp beds after 11 days. Results of tPAH analyses showed that no samples of kelp exceeded the level of concern.	7–11 days persistence No data on recovery

[a]CDFG and USFWS (1999); [b]Peckol et al. (1990); [c]Cross et al. (1987b); [d]Kittle Jr. et al. (1987); Mancini et al. (1989); [e]Antrim et al. (1995); [f]Klasing et al. (2015); Valentine (2019)

10.2.1 UNOCAL, Avila Beach

During the 1992 UNOCAL pipeline rupture near Avila Beach, California, marine surface waters transported San Joaquin Valley crude oil from the release site to the south where it contacted kelp beds and offshore rocks were subsequently exposed to the oil. Direct impacts to kelp were not reported (CDFG and USFWS 1999).

10.2.2 T/V World Prodigy

The June 1989 *World Prodigy* released 6,900 bbl of home heating oil into Narragansett Bay, Rhode Island under light winds and resulting calm seas. Peckol et al. (1990) compared the condition of sugar kelp (*Saccharina latissma*) and oarweed (*L. digitata*) growth rates with depth, and pigment acclimation were compared with pre-spill measurements of kelp at the same site from 1984 to 1987. There was no evidence of any detrimental effects to oil-exposed kelp; there was no observed necrotic or bleached tissue on any kelps in an oiled cove (Peckol et al. 1990). Growth rates of both species were within the range previous years' data. Peckol et al. (1990) suggested that Narragansett Bay kelp was potentially spared because little fuel oil mixed into the water column and contacted subtidal organisms.

10.2.3 BIOS Field Experiment

An experimental subsurface release of chemically dispersed oil at Cape Hatt, northern Baffin Island, resulted in short-term relatively high oil concentrations in the waters of two adjacent bays. *Laminaria* spp., which reproduce sexually, were rare and patchily distributed at the depth studied. There was no evidence of reproductive failure in this genus in 1983, 2 years after the oil releases, but any such effects in 1981 or 1982 would not have been detected.

10.2.4 T/V ARCO Anchorage

The 1985 *ARCO Anchorage* spill of 5,690 bbl Alaska North Slope crude oil at Port Angeles, Washington impacted bull and giant kelp beds (Mancini et al. 1989). An extensive 82-station visual observation survey was conducted by the state of Washington to document the nature and extent of oiling. The study documented the presence of sheens or patches of emulsified oil at 18.5% and 7% of the stations, respectively (Mancini et al. 1989). Oil was documented in kelp beds within an area of approximately 24 km east and west of Port Angeles where oil was noted during the initial spill response and monitoring activities. Kittle Jr. et al. (1987) observed visible, strongly adherent oil in offshore kelp. At stations where oiling was documented, divers observed oil on 50-60% of stipes, in some locations extending from the shoreline out to 10 m depth (Kittle Jr. et al. 1987). The oil had a viscous, sticky consistency and formed a band up to approximately 2.5 cm wide and 0.3 cm thick extending along the stipe from the pneumatocyst as far as 1.2 m. Accumulations of weathered oil were found on individual stipes, particularly where dense growths of epiphyton served as a sorption surface on the exposed stipes at the water surface.

The observations indicated oiled kelp transported oil to the bottom. Urchins were observed ingesting clean kelp fragments but not oiled kelp fragments and were not contaminated. The density of sea urchins and sea cucumbers and average size of sea urchins did not change following the spill (Kittle Jr. et al. 1987). Kelp crabs were impacted by the spill as some demonstrated narcotic effects of oiling, resulting in increased predation (Kittle Jr. et al. 1987). Emulsified oil persisted in kelp beds for 25 days (Kittle Jr. et al. 1987). Oil impacts likely persisted for 1 year or less in affected kelp beds due to the annual replacement of stipes (Mancini et al. 1989).

10.2.5 F/V *Tenyo Maru*

In 1991, the *Tenyo Maru*, reportedly carrying 6,500 bbl of intermediate fuel oil, 2,166 bbl of diesel fuel, and approximately 535 bbl of lube, bilge, and fish oil, collided with the freighter *Tuo Hai* 32 km northwest of Cape Flattery, Washington. The *Tenyo Maru* began leaking and sank within minutes of the collision. Oil patties were found floating in the dense bull kelp (*N. luetkeana*) canopies that fringed the coastline, and kelp stipes and bulbs appeared to be bleached (intense, synchronous color loss) and dying in the areas most heavily impacted (Antrim et al. 1995).

Following the incident, the effects of three petroleum products were tested on bull kelp due to the lack of published field or laboratory data on oil spill effects on this species. Moderate to severe damage to kelp tissue was observed following whole-plant exposure to weathered and unweathered diesel, intermediate fuel oil, and crude oil (Antrim et al. 1995). The 4-hour and 24-hour exposures to diesel and intermediate fuel oil resulted in moderate to severe damage to kelp tissue (i.e., clearly delineated bleached line accompanied by tissue necrosis). The most severe damage to bull kelp was concentrated at the meristematic zone (junction of stipe and bulb). When heavily bleached, portions of the plant decayed in 3–4 days and then broke off, while disruption of the mucus layer and subsequent drying lead to splitting and microbial decay of the tissue. Based on both photosynthetic-rate studies and oiled whole-plant effects, Antrim et al. (1995) found the relative ranking of the damaging effects of petroleum treatment on bull kelp to be weathered diesel > unweathered intermediate fuel oil > unweathered diesel > weathered intermediate fuel oil > unweathered crude > weathered crude.

10.2.6 Refugio Beach

Following the 2015 Refugio Beach spill of 500 bbl Monterey crude oil from the All American Pipeline, the oil that entered the ocean posed risks of exposure to marine plants and wildlife in the nearshore zone, which includes kelp forests and rocky reef. Surface oil was observed within the 0–10 m bathymetric zone where kelp occurs and farther offshore (Donohoe et al. 2019). Species included brown feather boa kelp, bladder chain kelp, and giant kelp. In Santa Barbara, the giant kelp is the foundation species of the subtidal rocky reef ecosystem (Miller et al. 2015). Early life stages of many fish and invertebrate species live in the kelp canopy near the water surface as the fronds dampen currents and provide protection; common species include polychaetes, sea urchins, sea stars, spiny lobsters, kelp bass, rockfish, sheepshead and algae (Schiel and Foster 2015). Seabirds and marine mammals also frequently forage in the kelp forest.

Submerged oil was observed to be most abundant at or on the sea floor in kelp beds during the first 7 days after the spill (Valentine 2019). Eleven days post spill, oil was likely diluted, presumably via transport processes and through a combination of flushing to the continental shelf, surfacing to form slicks, and incorporation into sediment. The transport of oil to the open shelf was likely slowed in the kelp forests and the occurrence of oil slicks within the kelp was consistent with resurfacing of benthic oils (Valentine 2019). Tumble tar was observed in kelp areas in sandy bottom depressions between reefs and caught in low energy sand channels between ridges. Patches of sheen trapped among the kelp were observed at the surface. Visible oiling was largely absent from the kelp beds 11 days after the spill.

Kelp was collected from Refugio Beach fisheries closure areas. Results of tPAH analyses showed that no samples of kelp exceeded the level of concern (27 ppb total benzo[a]pyrene equivalents), and concentrations of total benzo[a]pyrene equivalents in all kelp samples were very low (0.2 to 0.3 ppb) (Klasing et al. 2015). Farther offshore (3–10 m depth interval), giant kelp attached to rocky reefs were exposed to oil in the water column, and the surface of the kelp forest canopy was oiled. Fish and invertebrate species that inhabited kelp were found dead on beaches, primarily during the first week after the spill, and elevated PAH concentrations were detected in drift kelp consumers (urchins and sea cucumbers) (Valentine 2019). Because the kelp canopy can trap oil, this may have increased the exposure duration for kelp and the fish and invertebrates associated with the canopy, potentially resulting in increased mortality for the exposed organisms, particularly the more sensitive fish and invertebrate early life stages. In addition, direct contact of the kelp canopy with oil may have reduced primary productivity.

10.3 Recovery of Kelp Communities

For most spills, there was some documentation on the oil persistence but only anecdotal observations about the degree and extent of impacts to the kelp or the invertebrate and fish communities, and no information on which to estimate recovery. Kelp most often occurs in relatively high-energy environments where the oil persisted in the surface canopy for days to weeks. Bleaching of stipes and bulbs was observed only for the one spill that included light refined oils (*Tenyo Maru*).

10.4 Summary and Information Needs for Assessing Impacts to Kelp

The existing literature on the impacts to kelp and kelp communities from median-range spills of crude oil or diesel is very limited and only provides anecdotal information on impacts. For most spills, the oil that was trapped in the kelp canopy persisted for days to weeks. Light refined oils are likely to have more acute effects, resulting in bleaching of stipes and bulbs, primarily when the light oil is dispersed into the water column.

Following the 2015 Refugio Beach oil spill, benthic dive surveys, frequent tracking of surface oiling, and analyses of kelp samples provided a more detailed picture of potential impacts to and the duration of oil exposure in kelp communities. Any future spills impacting kelp communities will likely require a varied and intensive program of sampling and observation protocols, both at the surface and in subtidal benthic habitats, to improve the limited knowledge base of kelp community impacts following median-range spills.

Further laboratory studies could include experiments on mucus production, tissue uptake, pigment degradation, cellular damage, reproductive cycle disruption, and differing exposure durations. Modeling the degree of damage to the population in terms of loss of individuals, reproductive output, and recruitment and could be used to estimate the long-term damage of a spill to an impacted kelp forest (Antrim et al. 1995).

11 Submerged Aquatic Vegetation / Seagrass

11.1 Habitat Description, Communities, and Ecological Functions and Services

This chapter addresses impacts of spills of crude oil, condensate, or diesel (500–20,000 bbl) on submerged aquatic vegetation (SAV), defined as rooted vascular plants occurring mostly below the water surface. For this synthesis, the focus is on seagrass–the SAV habitat most studied after marine and coastal oil spills. SAV provides numerous ecosystem functions including habitat that is important to fish, shellfish, and wildlife resources, and value to humans in the form of coastal protection, water quality maintenance, fisheries production, and recreational opportunities. Seagrass represents one of the most productive ecosystems in the world (Zieman et al. 1984; Barbier et al. 2011). The distribution of seagrass is limited to shallow coastal waters due to light requirements for photosynthesis; seagrass requires sediments for rooting/attachment and nutrient supply (Thayer et al. 1984; Zieman et al. 1984). Seagrass grows throughout OCS regions with most species having their entire life cycle, including flowering and pollination, occurring underwater (Zieman et al. 1984).

Seagrass meadows perform biological and physical functions within their community with important linkages to other habitats and ecosystems, including:

- Leaves decrease current velocities, stabilizing sediments and increasing sedimentation.
- Roots and rhizomes create dense mats that bind sediments reducing erosion potential.
- Plants provide organic matter in the form of detritus, maintaining an active sulfur cycle and contributing to food web support.
- Leaves support large numbers of epiphytes that also contribute to food web support.
- Epiphytic loads can match or come close to the biomass of the plants themselves.
- Plants and epiphytes contribute to nutrient cycling and biogeochemical processes that involve transformations of carbon, nitrogen, phosphorus, and other key elements.
- Habitat is important for larval, juvenile, and adult life stages of fish, shellfish, and other macroinvertebrates, providing nursery functions, cover, and feeding areas.
- Habitat is important for wildlife, including provision of food resources for manatees and sea turtles through direct grazing and consumption of macroinvertebrate prey.

Marine organisms are found in great abundance in seagrass when compared to a seafloor devoid of vegetation. Many fish species use seagrass at some point in their life cycle, including larvae and juveniles of species spawned elsewhere that recruit to seagrasses as nursery habitat. Some species may stay within this habitat or move elsewhere as larger juveniles, sub-adults, or adults, spawning in inlets or offshore. Seagrass is vitally important to recreational and commercial fisheries. Federal regulations prohibit damage, destruction, or injury of this habitat type, often requiring mitigation or restoration when impacts do occur.

Seagrass habitats cover a wide geographic range and depths, ranging from beds that can be exposed at low tides to those at depths >10 m, and occurring primarily in coastal environments with unconsolidated substrates (Kenworthy et al. 2018). Belowground components of seagrass account for large portions of biomass including roots and rhizomes that anchor the plants, stabilize sediments, absorb nutrients, and enrich the substrate with organic matter (Kenworthy et al. 2017). Aboveground biomass, including the leaves and shoots, facilitate photosynthesis and is largely affected by water clarity that limits light penetration, thus dictating the depth at which seagrass can grow (Kenworthy et al. 2017). The combination of below- and above-ground biomass components of these ecosystems make them especially vulnerable to physical, chemical, and biological disturbances (Orth et al. 2006; Zieman et al. 1984; Kenworthy et al. 2018).

11.2 Oil Behavior and Persistence in Seagrass Communities

All studies identified for this review of oil impacts to seagrass communities were for crude oil or diesel spills; no studies were identified for condensate spills. While oil typically strands on shorelines in the upper intertidal or supratidal zones (during storms), oil can coat the lower intertidal zone inhabited by seagrass during low water levels and can directly coat shallow subtidal seagrass leaf blades that extend to or above the water surface. Subtidal seagrasses can also be exposed to physically dispersed oil droplets and dissolved oil constituents, settle on or be deposited within subtidal seagrass below the water surface when heavy or weathered oils sink, become neutrally buoyant, or mix with sediments or other particulate material that then deposit on the seafloor. Suspended sediments and particulates in the water column can adsorb oil, providing a mechanism for transport of oil to the substrate where seagrass communities are rooted. Seagrass belowground rhizome and root structures bind and stabilize substrates, and in the case of an oil spill, can promote retention of oil particles or oily detritus. Oil on the sediment surface may be worked deeper into sediments by seepage, burial, bioturbation, or other disturbance (Zieman et al. 1984). Remnant oil on the surface will be degraded by microbes; however, microbial activity is limited in aerobic sediments and even more limited in anaerobic sediments, potentially leaving oiled material buried for years (Lee 1977; Zieman et al. 1984).

Oil persistence in buried sediments in a seagrass meadow can allow reoiling of adjacent areas through periodic or slow release from the sediments or through animals that disturb or ingest the oil. Ingestion of oiled detritus or sediment particles can impact benthic deposit and filter feeding invertebrates (Zieman et al. 1984). Biological, chemical, mechanical (disturbance), and climatological factors in a specific location will determine oil persistence in a seagrass habitat. Oil is more likely to persist in seagrass at colder, higher latitudes for years or even decades, while tropical latitudes with warmer temperatures may facilitate more rapid degradation of oil in the community, unless oil is trapped or buried under sediments.

11.3 Impacts of Oil Exposure and Response on Seagrass

There are few studies that have quantified the impacts and recovery of seagrass from marine oil spills of crude oil or diesel in the median-size range examined here. Impacts to seagrass habitats may not occur as frequently as for other habitats, may at times go unrecognized, or determination of impacts may be more difficult to document as assessment methods often require divers, snorkelers, or remote sensing techniques (such as sonar or aerial photography), many of which are subject to sea and cloud conditions and water clarity.

Direct impacts to seagrass from oil range from complete mortality to leaf exfoliation, sublethal stress, and chronic impairment of plants and sediment metabolism and function. During the *Amoco Cadiz* spill (a larger spill than those examined in detail in this report, but useful as an example of impacts on seagrass), *Zostera marina* (eelgrass) was directly oiled and displayed dieback and “burnt” leaves in heavily oiled locations followed by normal new leaf tissue growth within a few weeks (Jacobs 1980). Secondary impacts include biophysical and chemical disturbance to the sediment, microfauna, and microflora (Short and Burdick 1995). Seagrasses in the Chandeleur Islands, Louisiana, exposed to oil during the *Deepwater Horizon* spill (another larger spill), showed persistent and delayed losses of seagrass cover over 2 years, documented using aerial image analysis, although gains in cover in nearby seagrasses were also observed (NOAA et al. 2016; Kenworthy et al. 2017).

A recent review of the impacts of oil on seagrass found that effects were dependent on many factors (Fonseca et al. 2016). The primary factors included proximity of the site to the point of oil release; oil type; tidal stage and range, and circulation patterns; and the location of the seagrass beds in the tidal frame. Lighter oils are generally more acutely toxic, and heavier oils can result in fouling and smothering effects. Longer-term impacts may be expected where sediments are contaminated, and roots and rhizomes are exposed to heavy or chronic oiling.

Direct and indirect impacts to seagrass habitats during oil spills can also occur during spill response. Potential impacts during response operations can include physical impacts and increased turbidity from vessel groundings, boat propeller (prop) scarring, prop-wash blowouts, anchoring of vessels and booms, stranding of boom in shallow water, and boom washing back and forth over seagrass beds during each tidal cycle. These actions can cause physical damage to leaves, shoots, and rhizomes; displace sediments directly or cause scour; and cause dieback and loss of seagrass (NOAA et al. 2016). Meehan (2015) found that response vessel operations during the *Deepwater Horizon* spill caused prop-scar impacts to seagrasses during boom deployment in shallow areas. Prop scars are well known impacts to seagrass beds when boats operate in shallow seagrass areas. Prop scars can displace sediment and cut through the rhizomes, exposing the bed to potential scouring and erosion that can expand beyond the original scar, leading to further loss of seagrasses. Prop scars and other physical vessel impacts can take years to decades to fully recover and, in some cases, damaged seagrass beds will not recover without active restoration (Kenworthy et al. 2006). Finally, shading by response vessels, especially barges, moored over seagrasses for long periods, can prevent or reduce the amount of light reaching the plants, reducing photosynthesis and leading to plant stress or death.

From a thorough review of the literature on the impacts of oil spills and response, two oil spills were identified as having data from field-based studies from spills of 500–20,000 bbl of crude oil (**Table 11-1**). The *Cosco Busan* spill of a heavy fuel oil is also included due to the limited number of detailed field studies available for spills in the median range and evidence of response-related damage was documented.

It is important to note that the specific definition of recovery varies from case-to-case and by the species or taxonomic groups and metrics that were studied. In this report, we summarize and synthesize the limited information available and acknowledge that recovery in this context does not necessarily equate to full ecological or ecosystem recovery. This point applies to all habitats. Years to recovery in **Table 11-1** is based on information available in the literature. However, given the heterogeneous nature of oil spills within nearshore waters, different areas within seagrass beds may exhibit varying effects of exposure, thus recovery can begin within 1 year and extend years or decades depending on the severity of the impact.

Based on the available literature, impacts to seagrass communities can be described in three phases: 1) an impact phase, with a measurable reduction in abundance and species occurrence caused by direct mortality from the oil's toxicity and/or disturbances from response activities; 2) an initial recovery phase, where there is an increase in vegetative growth, density, and species occurrence, particularly for colonizing species; and 3) the return of normal species composition, growth, and density indicating the start of full recovery, once the toxicity of any residual oil is reduced to below effects levels. Full recovery is achieved when the seagrass community reaches the species composition, density, and age structure comparable with unoiled reference sites.

The research conducted to evaluate impacts to seagrasses from oil ranges extensively from scientifically rigorous evaluations (i.e., *Cosco Busan,* Refugio Beach, TROPICS field oiling experiment) to opportunistic observations without documented data (i.e., UNOCAL and *Garbis*).

Table 11-1. Studies with documented or estimated impacts to and recovery of seagrass from spills of crude oil and one heavy fuel oil spill (500–20,000 bbl)

Oil Spill	Degree of Oiling (oil type and volume)	Oil Cleanup	Seagrass Species	Documented Effect/Impacts	Recovery (years)
2007 *Cosco Busan*, CA[a]	Very Light to Heavy (1,285 bunker fuel oil)	Yes, shoreline cleanup methods were used including cold spraying and manual removal	Eelgrass (*Zostera marina*)	Though many eelgrass beds were exposed to oil, there was little evidence to suggest serious injuries to them. Response vessels impacted shallow subtidal eelgrass beds, documented through side scan sonar surveys.	<1
2017 Refugio Beach, Santa Barbara, CA (214 km)[b]	Very Light to Heavy (500 bbl Monterey crude)	Yes, manual	Surfgrass (*Phyllospadix* sp.), *Zostera marina*	Two months post spill, discolored/ dead surfgrass (was observed in areas of heavy oiling. Almost 1 year post spill, oil-related injuries such as bleaching, necrosis, loss of biomass, cellular death, and loss of surfgrass leaf tensile strength were observed throughout the range of surfgrass habitat in areas of heavy oiling.	>1
1984 TROPICS, Panama[c]	Moderate (6 bbl Prudhoe Bay crude)	No, field experimental oiling study	Turtle grass (*Thalassia testudinum*)	Initial short-term effects in the first 2 years indicated reduced seagrass growth rates within the oiled site and included effects on leaf blade area and densities that continued for 10 years. 20 years post spill: an increasing trend of turtle grass growth rate at the oil treatment sites.	10–20

[a]Fonseca et al. (2016); Cosco Busan Oil Spill Trustees (2012) [b]Refugio Beach Oil Spill Trustees (2020); Cosentino-Manning (2018); Tenera Environmental (2020); [c]Ward et al. (2003); Baca et al. (2014)

The 2007 *Cosco Busan* spill of heavy fuel oil caused heavy to very light shoreline oiling within San Francisco Bay that included adjacent eelgrass beds. Approximately 380 hectares of eelgrass beds were exposed to oil; however, only 12.1 hectares were within moderately or heavily oiled shorelines (Fonseca et al. 2016). Response workers deploying sorbent pompoms behind boats (for trapping oil) in open water and within eelgrass beds did not find submerged oil, and it was concluded that negligible amounts of oil were in the water column or on the bottom of the bay (Cosco Busan Oil Spill Trustees 2012). Side scan sonar surveys were conducted at several sites to measure the density of eelgrass beds and identify anomalies that may have occurred during the response. The sonar surveys did not provide conclusive differences between oiled vs. unoiled eelgrass beds; however, they did document water craft injuries resulting from response actions at Keil Cove (Cosco Busan Oil Spill Trustees 2012). The surveys revealed four prop scars that appeared within the time of the response. Researchers compared pre-spill to post-spill data of eelgrass beds throughout the bay by evaluating photosynthetic activity, rhizome node production, and phenolic compound analysis. Those results were also inconclusive for impacts specific to oiled beds vs. unoiled beds. The authors suggested that natural variation among eelgrass beds, bed elevation, proximity to oiling, location within the tidal range, and seasonal variability as reasons for much of the differences among survey locations (Fonseca et al. 2016). Furthermore, the study was conducted 3 weeks post spill, possibly allowing the eelgrass to recovery from minimal exposure to the oil (Fonseca et al. 2016).

The 2015 Refugio Beach spill in California released Monterey crude oil from a ruptured pipeline into the Pacific Ocean, oiling over 200 km of shoreline and extending more than 13 km offshore (Refugio Beach Oil Spill Trustees 2020). Nearshore and subtidal habitats were impacted by wave action that actively mixed the oil throughout the water column within the surf zone causing the exposure of *Phyllospadix* sp. (surfgrass) and eelgrass beds to oil. Within 2 weeks of the spill, the response utilized multi-beam sonar surveys, side scan sonar surveys, videos, photographs from a remotely operated vehicle, and diver inspections to investigate reports of submerged oil. Priority sites were surveyed from near shoreline to depths of 10 m where divers collected sediment, vegetation, and invertebrate samples from eelgrass and surfgrass habitats. Oil was detected (as PAHs) in the vegetation, and many invertebrate species in the surfgrass had detectable oil consistent with the spilled oil (Refugio Beach Oil Spill Trustees 2020). Two months post spill, discolored and dead surfgrass was observed in areas of heavy oiling; after almost 1 year post spill, oil-related injuries such as bleaching, necrosis, loss of biomass, cellular death, and loss of surfgrass leaf tensile strength were observed throughout the range of surfgrass habitat in areas of heavy oiling (Cosentino-Manning 2018; Tenera Environmental 2020). In the first year, these impacts to surfgrass ranged from 35% to 85% in the heavy oiling locations, compared to 2% at the unoiled reference site.

The 1983 TROPICS field experiment in Panama was designed to determine the intertidal/subtidal tradeoffs involved with chemical dispersant use during oil spills in shallow tropical waters (Ballou et al. 1987). The treatments included no oil, oil, and dispersed oil at sites characterized by *Thalassia testudinum* (turtle grass), coral reefs, and mangrove forests. Only the oiled treatment effects on seagrasses are discussed here. Controlled releases over 48 hours of Prudhoe Bay crude oil (dosed at $1L/m^2$) were used in the oiled treatment (Baca et al. 2014). Both short- and long-term effects and recovery rates have been analyzed over the past 4 decades. Initial short-term effects within the first 2 years indicated reduced seagrass growth rates within the oiled site, including effects on leaf blade area and densities. Long-term effects, 10 years after oiling, showed reduced seagrass shoot densities from 841 shoots/m^2 to 440 shoots/m^2 at the oiled treatment sites (Ward et al. 2003). Studies conducted 20 years post spill demonstrated an increasing trend of turtle grass growth rates at the oil treatment sites but a decreasing trend in the reference sites. It appeared that the seagrass beds had stabilized from the effects and no measurable site differences were found (Ward et al. 2003). Seagrasses recovered to pre-spill conditions between 10–20 years after exposure, and localized changes could be a result of competition with corals for cover or nutrient enhancement.

Several median-range oil spills examined here have documented impacts to seagrass but do not include recovery estimates. The 1992 UNOCAL spill in Avila Beach, California, occurred in the vicinity of surfgrass; however, impacts were not documented. The restoration action report for the spill described heavy oiling impacts within Boulder Cove, an intertidal area composed of rocky outcrops, bedrock platforms, large boulders, cobbles, and sand that supports a diversity of plant and animal life with extensive surfgrass in sand-lined tidepools (CDFG and USFWS 1999). Additionally, studies of the 1975 *Garbis* spill in the Florida Keys reported that floating seagrass blades were oiled and deposited on the beach as wrack, thus oiling the sand beaches (Chan 1977). No attached seagrass was observed to be oiled during this spill response. Observations such as these do not provide enough detail to accurately document impacts or estimate recovery to the resource.

11.4 Summary of Data Gaps for Assessing Impacts to Seagrass Communities

Impacts to seagrass communities from an oil spill can have relatively minor to major impacts depending on the oil type, amount, and degree of weathering; the proximity of the oil release to seagrass communities; environmental factors including tides, water depth, waves, turbidity, water temperature, and salinity (among others); and response actions that can reduce and/or increase impacts, depending on the circumstances. Though there are limited documented impacts to seagrass from median-range oil spills, the literature is consistent in that seagrass vegetation is usually not damaged extensively unless heavily coated with oil, physically impacted, or deprived of light for prolonged periods. Acute exposure may cause minor fouling of leaf blades and flowering parts of seagrass; however, full recovery is often relatively rapid, within 1–2 years. Oil entrained in sediments within seagrass meadows may persist for many years before degradation occurs. This long persistence may decrease seagrass abundance, cover, and diversity, and have similar effects on associated infauna communities, but surrounding areas are likely to recover. There was a notable paucity of, and need for, oil-impact studies on faunal communities associated with seagrass habitats.

In some cases, response operations can have greater impacts and require longer recovery times than the direct oil impacts. Response impacts can include physical damage through vessel groundings, prop scarring, and prop wash blowouts, as well as increased turbidity, sedimentation, and even oil exposure caused by shoreline cleanup activities occurring in proximity to nearshore shallow seagrass habitats. Shading due to increased turbidity or the presence of response vessels and equipment on the water over seagrass beds could also be a factor in some cases (e.g., barges staged over seagrass beds). Identifying the presence and location of seagrass during oil spill planning and response can facilitate appropriate measures for their protection during oil spills and response operations.

To better quantify the impact and recovery of oil on seagrass communities (not just the vegetation), it is recommended that rigorous field study designs for seagrass habitats be prepared, ready for implementation in the event of a spill. Field study designs could be developed based on insights from prior studies such as the TROPICS experiment, natural resource damage assessment studies, and other well-designed studies of oil exposure, impacts, and recovery in field and mesocosm settings. Such field designs and future studies would better define the responses of different seagrass species and associated organisms to oil and response impacts.

12 Fish and Motile Invertebrates

12.1 Resource Description

This chapter addresses the impacts of spills of 500–20,000 bbl of crude oil, diesel, or condensate on fish and larger motile invertebrates (crustaceans, cephalopods, etc.). Invertebrates associated with an intertidal or subtidal habitat or community type (e.g., beach infauna, bivalves, rocky intertidal species) are discussed in other chapters.

Fish and invertebrates occupy a wide variety of habitats and have an amazing diversity of life-history strategies. Pelagic species live primarily in the water column and may remain within a relatively narrow depth range or move up and down within the water column in search of food. Demersal species live near the bottom of the ocean and can be categorized as benthopelagic (occurring near the bottom), or benthic (living in or on the sea floor). These species can be associated with specific benthic habitats or more broadly distributed. Estuarine and coastal species live in nearshore waters. Many of these species show ontogenetic shifts in habitat, commonly spawning in specific habitats or depth zones (nearshore and or offshore), using shallow estuarine and nearshore habitats as juveniles, then transitioning to deeper nearshore of offshore habitats as they mature. Some may move seasonally in search of food and/or spawning habitats. Anadromous species move between marine and freshwater habitats, spawning in freshwater rivers and lakes, typically depositing eggs on coarse substrate. Juveniles hatch and use riverine and estuarine habitats as nurseries before returning downstream to marine habitats, where they mature into adults. Catadromous species have opposite life histories, being born in the ocean and using inland habitats as nursery habitats. A subset of estuarine species, wetland resident species, live in very close association with coastal wetlands such as marshes and mangroves, often using flooded wetlands for foraging on the high tide and retreating to pools, tidal creeks, and wetland shoreline edges during low tide.

12.2 Routes of Exposure and Species Sensitivity to Oil

The extent to which fish and invertebrate species are exposed to spills of crude, condensate, or diesel depends on many factors, including the fate and extent of oiling, habitats oiled, and species biology. Oil in the water column can physically foul gills and other tissues, be absorbed during respiration, and ingested while feeding. Oil at the surface of the water column can become trapped within floating *Sargassum*, increasing the exposure of *Sargassum*-associated communities (Powers et al. 2013). Oil that sinks or becomes bound to sediment can affect benthic species that forage in those habitats or burrow for refuge from predation.

Risk of exposure may be mitigated or enhanced by aspects of individual species' biology, including habitat preferences, behavioral traits, or life-history strategies. Habitat preference is especially important because it determines the route and magnitude of a species' exposure. Pelagic species are highly mobile and may have less risk of exposure from oil spills because juveniles and adults are better able to avoid heavily oiled areas. Demersal species are often associated with specific benthic habitat types, such as coral reefs or seagrass beds, and use these features for foraging or as refuge. As a result, they presumably move less than pelagic species and are less likely to avoid oil exposure.

Biological and behavioral differences among taxa determine the sensitivity of a species and/or population to oiling impacts. Feeding mode may affect the potential that a species is exposed to oil contamination. Filter-feeding and deposit-feeding organisms have increased exposure to oil (both oil droplets and oil attached to particulates) while feeding, compared to organisms that selectively target their prey. Biological differences lead to different rates of accumulation and depuration of contaminants among invertebrates, leading to differential impacts. Species may exhibit more than one behavior or characteristic that exacerbates or minimizes their risk to oiling.

Some behaviors can decrease an organism's exposure to oiling. Several species of fish and invertebrates have the ability to detect oil (e.g., Pearson et al. 1981), leading to the hypothesis that, to some extent, they can avoid contaminated waters and/or sediments. However, there is limited evidence this occurs in environmentally relevant exposures. Substrate preference may also have a role in exposure, as certain sediments may bind to oil more readily; therefore, they are associated with higher hydrocarbon levels.

The timing of a spill with respect to important life-history events (i.e., spawning, migration) also determines the extent to which a species is impacted. Larvae are especially sensitive to impacts from oil because oil may be easily absorbed into the body (Dupuis and Ucan-Marin 2015). Research has shown that even transient exposures to oil can cause delayed toxicity to larval fish, even to those that subsequently develop in clean water (Hicken et al. 2011). Spills that occur outside of seasons important for reproduction and growth of larvae or juveniles are likely to have less impact than spills that occur in these seasons.

The duration and extent of life-history events also determines the severity of impacts to a population. Species that have protracted spawning seasons and high fecundity may be able to compensate for spill-related impacts at the population level, as mortality to part of the year-class can be offset by higher survival of offspring that spawned earlier or later in the year. Similarly, populations that spawn across a large area could be more resilient than populations that spawn in a limited geographic region. Lastly, species that have longer lifespans are less likely to show population-level effects because impacts to one annual cohort can be compensated for by enhanced survival of other cohorts. However, when long-lived species are heavily impacted, recovery timelines can be lengthy because they are dependent on the generational timelines of these species.

12.3 Effects of Oil Exposure for Fish and Motile Invertebrates

Oil exposure can cause lethal or sublethal effects. Lethal effects occur when fish and invertebrates come into contact with high concentrations of oil and can occur due to physical smothering by more viscous oils (i.e., fouling of gills leading to asphyxiation), exposure to the lethal concentrations of toxic components of oil, or through secondary effects of the spill such as hypoxic events resulting from biodegradation of oil or toxicity via ingestion of contaminated prey. Direct mortality of fish and invertebrates is difficult to observe because dead animals may sink or be consumed by other organisms.

Sublethal effects from exposure to the toxic components of oil range in duration and severity of impact. Early life history stages are especially vulnerable. Embryonic exposure has been shown to cause cardiac defects in fish (Incardona et al. 2011) and crabs (Suchanek 1993) and reduced reproductive success in adult fish (Rice et al. 2001). Larval exposure has been shown to cause elevated mortality and stunted growth in many species (e.g., Brown et al. 1996).

Exposure to oil can be detected by measuring PAHs and/or metabolites of PAHs, or by measuring biochemical changes (e.g., biomarkers) elicited by oil exposure. PAH concentrations and associated biomarkers are generally indicators of recent oil exposure (days to weeks). Common metrics for assessing oil exposure in fish and motile invertebrates include:

- PAH concentration: M\measured PAH concentrations present in different body tissues, commonly muscle, reproductive tissue, and/or liver (or in the case of many invertebrates, hepatopancreas);
- Induction of cytochrome p450 1A (CYP1A), which is an enzyme activated to metabolize organic compounds, including aromatic hydrocarbons; and
- Biliary metabolites: PAH metabolites that are found in bile, generally collected from an animal's gall bladder. This measure can only be made in fish and other vertebrates with gall bladders, thereby excluding its application in invertebrate species.

Exposure to oil can lead to sublethal impacts (**Table 12-1**). These impacts can indirectly affect an organism's health and can have delayed impacts, such as mortality or declines in abundance or viability of offspring.

Table 12-1. Sublethal impacts and effects of oil exposure to fish and motile invertebrates

Impact	Detection Method(s)	Potential Effects
Genetic damage	Oxidative DNA damage, formation of DNA adducts	Cancer, abnormal growth, impaired reproduction
Morphological changes (heart defects, developmental abnormalities, fin rot)	Histological examination; visual examination	Impaired movement, leading to increased mortality
Behavioral changes	Lab observations; tagging studies	Decreased fitness (e.g., reduced foraging efficiency, decreased oil avoidance behavior)
Reproductive impairment	Histological examination, detection of PAHs in tissues, gonadosomatic index (GSI)	Reduced fecundity or viability of offspring
Immune dysfunction	Lab studies	Increased susceptibility to disease and mortality
Physiological impairments	Lab observations	Reduced foraging ability; reduced predator avoidance
Lowered growth rates	Lab observations, assessing growth by microanalysis of otoliths	Decreased growth
Decreases in general health	Condition indices (somatic index, hepatosomatic index, gonadosomatic index)	Decreased fitness

12.4 Oil Impacts to Fish and Motile Invertebrates

Information on fish and invertebrate impacts was available for twenty-three of the sixty-two spills included for this review (**Table 12-2** and **Table 12-3**). Spills reviewed were primarily crude oil and light refined products. Only one spill involved condensate. The heavy fuel oil spill from the *Cosco Busan* was included because of the quality of the post-spill monitoring and the paucity of such data that quantified impacts. This discussion is limited to wild animals. Impacts to caged or reared animals are not discussed as part of this chapter.

For all the spills considered, impacts were localized and limited to populations exposed to large amounts of the spilled product(s). Three studies did not find evidence of impact from spills. Six spills only measured PAH contamination in affected species without assessing or observing sublethal or lethal impacts to the species. Eleven spills had information on mortality estimates of affected species. Information on sublethal impacts was available for seven spills. All the impacts reported occurred in nearshore waters. Most of the impacts documented occurred in confined nearshore environments or intertidal areas.

Table 12-2. Studies of spills where impacts to fish and motile invertebrates were not detected or only documented PAH contamination without assessing impacts to the population

Oil Spill, Location, Volume and Oil Type	Species Group (Species)	Documented Impacts/Exposure
1972 Long Island Sound Spill New Haven, CT 1,905 bbl No. 2 fuel oil[a]	Invertebrates (hermit crabs, whelk, quahog) Fish (flounder)	Evidence of PAH contamination to hermit crabs; no contamination in other species; no impacts assessed.
1977 T/B *Bouchard 65* Buzzards Bay, MA 2,000 bbl No. 2 fuel oil[b]	Invertebrates (bivalves, lobster, crabs)	Some contamination found but impacts were not observed; ice kept oil out of sensitive shoreline habitats.
1979 T/V *Esso Bayway*, TX 6,000 bbl Light Arabian crude[c]	Invertebrates (shrimp)	Impacts were not detected; salinities were lower than normal and shrimp habitats had shifted away from the area of impact as a result.
1981 T/V *Sefir* Baltic Sea 2,800 bbl No. 1 fuel oil and diesel mixture[d]	Invertebrates (clams, mussels)	PAH exposure detected (levels 50 times higher than background) but impacts to communities were not detectable.
1984 Uniacke G-72 blowout Nova Scotia, Canada 1,500 bbl gas condensate[e]	Demersal fish (haddock, cod, American plaice, redfish)	No evidence of tainting in fish caught near well; likelihood of adverse effects to fish stocks 'minimal'.
1989 T/V *Bahia Paraiso* Arthur Harbor, Antarctica 3,760 bbl diesel fuel[f]	Fish (icefish)	Evidence of contamination in areas near spill found 2 years post spill; no assessment of impacts to species health or population.
1996 T/V *Julie N* Portland, ME 2,058 bbl home heating fuel and 2,219 bbl IFO 380[g]	Invertebrates (softshell clams, blue mussel)	PAH concentrations in invertebrates 8-30 times higher than normal; no other impacts assessed.
1999 T/V *Estrella Pampeana* Rio de la Plata, Argentina 15,700 bbl crude oil[h]	Fish	Impacts were not detected; high abundance of larval fishes found after the spill.
2000 T/V *Eurobulker* Aegean Sea, Greece 4,850 bbl crude[i]	Invertebrates (clams)	Low levels of PAH contamination in clams; no other impacts assessed.

[a]EPA (1973); [b]Schrier (1978); [c]Neff et al. (1981); [d]Linden et al. (1983); [e]Carter et al. (1985); Gill et al. (1985); [f]McDonald et al. (1992); [g]Mauseth and Csulak (2003); Reilly (1998); [h]Moreno et al. (2004); [i] Zenetos et al. (2004)

Table 12-3. Studies with documented or estimated impacts to and recovery of fish and motile invertebrate species from spills of crude oil, condensate, or diesel-like oils (500–20,000 bbl)

Oil Spill, Location, Volume and Oil Type	Species Group (Species)	Documented Impacts/Exposure	Impact Period (years)
1975 T/V *Garbis* Florida Keys 1,500–3,000 bbl crude emulsion[a]	Nearshore fish and shellfish	Dead and distressed animals noted in proximity to oiled marsh.	Not provided
1979 T/V *Ryoyu Maru No. 2* St. Paul Island, Bering Sea, AK 6,190 bbl No. 2 fuel oil[b]	Invertebrates	Spill killed 50% of microorganisms and 'higher invertebrates' in impacted lagoon.	Not provided
1980 T/V *American Trader* Huntington Beach, CA 9,919 bbl Alaska North Slope crude[c]	Fish (White seabass, grunion)	Mortality of animals observed following contact with oil; Eggs collected from oiled beaches had reduced viability compared to reference beaches.	Not provided
1989 T/V *World Prodigy* Narragansett Bay, RI 6,900 bbl home heating oil[d]	Invertebrates (lobsters and crabs)	Documented mortality of 807 lobsters and crabs killed in impacted coves (mostly rock crabs); true impact likely higher.	Not provided
	Pelagic fish and lobster larvae and eggs	Large portion of year class impacted.	Not provided
1998 Equinox well blowout Lake Grand Ecaille, LA 1,535 bbl South Louisiana crude[e]	Fish and invertebrates	Injury assessment indicated lost production of less than 1,707 kg of finfish and shellfish.	Not provided
2017 Refugio Beach, Santa Barbara, CA 500 bbl Monterey crude[f]	Surf zone fish species (grunion, surf perch)	Exposure to oil documented; increased mortality of grunion embryos from oiled areas compared to reference beaches.	1 month
	Subtidal fish and invertebrates	Over 30 species collected dead on the beach. Repeat surveys 1 year later recorded lower abundance and diversity of dead species on the beach.	1 week
2001 T/V *Jessica* Galapagos Islands, Ecuador 2,800 bbl diesel and 2,160 bunker fuel[g]	Reef fish, invertebrates	Anecdotal reports of mortality but impacts were not observed in formal assessment. Three months after spill, fish densities and species richness were higher closer to wreckage.	<0.25
2005 Barataria Bay Barataria Bay, LA 600 bbl South Louisiana crude[h]	Marsh edge fish and crustacean communities	Immediate effects on abundance. Mobile species emigrated. Species composition remained constant.	<1 (hurricanes prevented additional sampling)

Oil Spill, Location, Volume and Oil Type	Species Group (Species)	Documented Impacts/Exposure	Impact Period (years)
2007 M/V *Cosco Busan** San Francisco, CA 1,285 bbl bunker fuel oil[i]	Fish (Pacific herring)	Spill impacted spawning beaches in central San Francisco Bay. As much as 25% of winter spawn impacted; near total mortality of spawn in oiled sites. Sublethal effects found 2 years after the spill but without elevated mortality; no long-term impacts.	1
1987 M/V *Antonio Gramsci* Gulf of Finland, Baltic Sea 4,200 bbl crude[j]	Fish (perch, flounder, herring)	Elevated hydrocarbons in liver and negative impacts to health (lower GSI/liver somatic index, increase in parasite occurrence and disease).	1
1987 T/V *Arco Anchorage* Port Angeles, WA 5,690 bbl Alaska North Slope crude[k]	Invertebrates (hardshell clams, crabs, shellfish)	Mortality and reduced fitness of affected animals.	1.5
1969 T/B *Florida* West Falmouth, MA 4,385 bbl No. 2 fuel oil[l]	Fish, Invertebrates	95% mortality of fish and invertebrates in nearshore waters. Shellfish more contaminated 2 years post spill than immediately post spill due to oil being released by sediment. Fish contamination noted.	Shellfish: >2 Fish: 1
1993 Rose Atoll American Samoa 2,380 bbl diesel and lube oil[m]	Invertebrates (giant clams and echinoderms)	Mortality of giant clams and echinoderms observed after grounding/spill occurred, population effects minimal. Urchin densities still reduced 5 years later.	Clams: <2 Urchins: 5
1996 T/B *North Cape*, South Kingstown, RI 19,700 bbl home heating oil[n]	Surf clams	19 million estimated killed; extremely high densities of recruits documented in the 2 years after spill.	3–5
	Lobsters	Loss estimated at 9 million animals. Density in impacted areas was reduced by 85% compared to nonimpacted areas.	4–5
	Fish (multiple)	4.2 million killed in offshore area; 533,400 in salt ponds. Oil exposure led to 51% decrease in embryonic survival of winter flounder, but overall impacts 'slight'.	Not provided

[a]Chan (1977); [b]Reiter (1981); [c]American Trader Trustee Council (2001); [d]Catena (1996); Pilson (1990); [e]State of Louisiana (2005); [f]Witting et al. (2018), Refugio Beach Oil Spill Trustees (2020); [g]Edgar et al. (2002); [h]Roth and Baltz (2009); [i]Cosco Busan Oil Spill Trustees (2012); [j]Hirvi (1990); [k]Kittle Jr. et al. (1987); [l]Blumer et al. (1970b); Teal et al. (1992); [m]USFWS (1997); [n]DeAlteris et al. (1999), NOAA (1999), Hughes (1999)

**Cosco Busan* spill was a heavy fuel oil but included here because of the detailed study conducted.

Nearshore impacts are more commonly documented for many reasons, including:

- There is potential for greater exposure to oil in these areas compared to offshore environments because oil aggregates along the shoreline, where waters are shallower and more confined.
- Nearshore impacts are easier to detect, due to confined geographies and ease of observation.
- Nearshore resources are more often studied and more likely to be exploited, because they are more accessible.
- Many of these spills occur in nearshore areas because they are the result of groundings or allisions, which are more likely to occur in shallow waters and/or confined waters with more obstacles and vessel traffic.

Mortality was the most commonly documented impact (**Table 12-3**) and could be quite high in cases where a lighter oil was spilled in a confined area. Of those impacts, the *North Cape* spill had the most severe losses due to a spill; model results provided estimates of more than 30 million fish and invertebrates killed due to the spill (NOAA 1999). Impacts also varied based on the chemical composition of the spilled product.

Multiple spills showed impacts to species due to oiling of spawning or nursery habitat. Early life history stages are especially vulnerable to oiling due to increased biological susceptibility, limited mobility, and their occurrence in confined habitats, which tend to aggregate oil. Following the *North Cape* spill, winter flounder showed a 51% reduction in embryonic survival (Hughes 1999). The impact was exacerbated because pelagic embryos were concentrated by the same lagoonal hydrodynamics that concentrated the oil. Despite this drastic reduction, population losses were minimal because later-season spawning compensated for early-season losses, and impacts were limited to a few bays. However, although not evaluated, losing a major input from early or mid-season spawners will have the potential to reduce life history diversity. The *World Prodigy* spill of home heating oil also caused mortality of spawned pelagic juveniles that came into contact with floating oil (NOAA 1996).

Oiling of shoreline spawning habitats is documented to cause embryonic mortality. Three spills along the Pacific shoreline of the U.S. had documented impacts to intertidal spawning species. The *Cosco Busan* spill of a heavy fuel oil caused heavy oiling to intertidal areas used for spawning by Pacific herring, impacting 25% of spawning beaches. Embryos from oiled areas showed evidence of sublethal cardiac toxicity, higher mortality, and increased tissue necrosis compared to unoiled areas. Sublethal effects to embryos exposed to oiled areas persisted a year after the spill (Incardona et al. 2012a). Despite the loss of an estimated 14-29% of the winter spawn in central San Francisco Bay, population levels were not impacted by the spill because later spawning events compensated for losses caused by the spill. Elevated embryonic mortality was also documented for grunions spawning in oiled beach sediments after the *Refugio* (Witting et al. 2018) and the *American Trader* spills (American Trader Trustee Council 2001).

Many of the spills reviewed were caused by vessel accidents, which can impact fish and invertebrate communities in other ways in addition to oil exposure. The physical structure of the wreckage can cause changes in local fish and invertebrate communities separately from the oil spilled. Following the *Jessica* spill of diesel and a heavy fuel oil, the structure of reef fish communities changed such that fish were responding to the increase in structure provided by the ship (Edgar et al. 2003b). The resulting vessel debris can have secondary effects.

Environmental factors can mitigate or exacerbate spill effects. The *Esso Bayway* spill occurred during a year with abnormally high precipitation, which lowered salinities in the estuary and shifted the distribution of shrimp towards more offshore environments, preventing the spill from affecting these species (Neff et al. 1981). The presence of ice can also affect the impacts from a spill. However, following the *Antonio Gramsci* spill, oil became trapped in the sea ice until it melted, lengthening the duration of exposure. Subsequently released oil caused PAH contamination of liver tissue for 6 months and sublethal impacts to liver, immune system, and reproductive systems to perch, flounder, and herring (Hirvi 1990).

Spills that sink or become mixed into sediments can lead to longer-term effects on species. Following the *Florida* spill, oil became incorporated into intertidal and subtidal sediments, causing contamination of shellfish from the area for at least 2 years (Blumer et al. 1970a). Marsh fish (*Fundulus* sp.) showed evidence of PAH contamination and increased levels of hydrocarbon-metabolizing enzymes in the year following the spill, indicating continued exposure (Burns and Teal 1971). These compounds were detected as long as 20 years after the spill, though in very small concentrations that may be have resulted from other sources of oiling (Teal et al. 1992).

Recovery times for fish and invertebrates following eighteen spills are shown in **Figure 12-1**. Much of the literature was focused on evaluating initial impacts; therefore, information on recovery times is limited. In some cases, recovery times shown in **Figure 12-1** are based on estimated time to recovery given the life-history of a species. In most cases, recovery began once the oil was removed or dissipated and was estimated to occur within a year. The longest predicted recovery time was 4–5 years for lobsters following the *North Cape* spill; however, no studies were done to confirm the recovery of this species.

12.5 Summary and Information Needs for Assessing Impacts to Fish and Motile Invertebrates

Studies reviewed generally found that the impacts to fish and invertebrate populations after spills of 500–20,000 bbl were relatively minor and populations recovered quickly once the oil was removed. The most severe impacts occurred when essential habitats were heavily oiled; however, they did not result in long-lasting impacts to populations. In all spills examined here, natural recruitment was observed or expected to lead to population recovery within a generation, if not sooner. In some cases, recruitment was unusually high within a year after the spill as new animals repopulated impacted areas.

Literature on documented impacts was relatively sparse, outside of studies completed in support of damage assessments. Very little information was available on long-term impacts; in fact, many studies did not assess any effects beyond the initial impact. While this is sufficient for damage assessment purposes, additional monitoring to document recovery in these cases would inform responses to future spills. As an example, the work documenting impacts to Pacific herring from the *Cosco Busan* built upon studies of toxicity that had been conducted after the *Exxon Valdez* spill, which allowed for a greater understanding of the effects of oil on the early life stages of Pacific herring, and led to the discovery that ultraviolet (UV) radiation enhanced the toxicity of weathered oil to those animals (Incardona et al. 2012b).

Linking individual impacts to population-level impacts is challenging for fish and invertebrate species. Direct mortality can be difficult to observe, especially in certain habitats, such as the open ocean. Fresh oil fouls nets and other sampling gear, preventing sampling in the immediate aftermath of a spill. Short-term effects to individual fish and invertebrates often do not lead to detectable long-term impacts to the population (Fodrie et al. 2014; McIntyre 1982). Many fish and invertebrate populations are naturally variable in space and time, making it difficult to discern the influence on fluctuations in abundance and/or density.

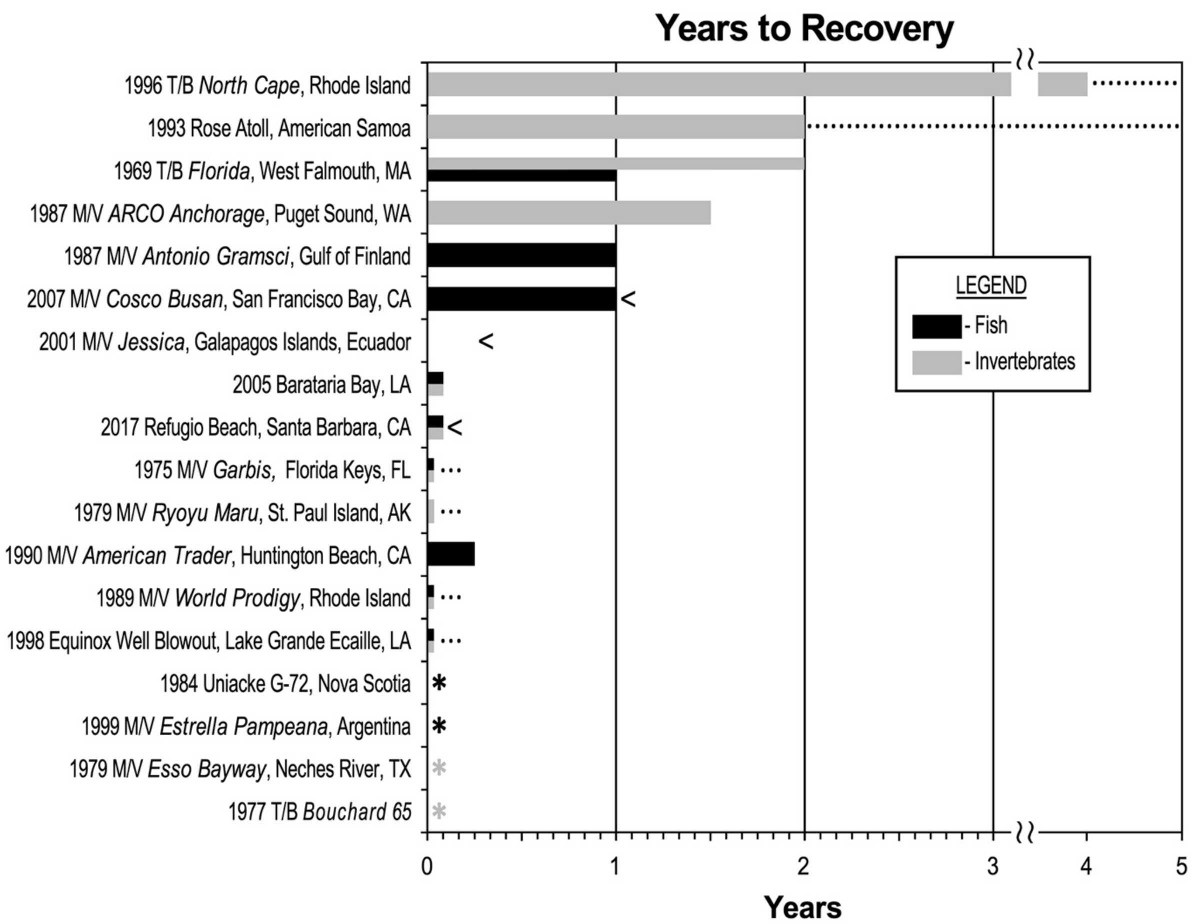

Figure 12-1. Recovery of fish and invertebrates for spills (500–20,000 bbl) of crude oil, diesel, or condensate (including the *Cosco Busan* spill of a heavy fuel oil)

Black lines show recovery rates for fish; grey lines show recovery rates for invertebrates. Dotted lines indicate incomplete recovery at the time of the most recent study.

Asterisk indicates that impacts were not detected. < symbol indicates that recovery occurred prior to the date of the most recent study.

13 Marine and Coastal Birds

13.1 Resource Description and Habitat Use

This chapter addresses potential impacts of spills of 500–20,000 bbl of crude oil, condensate, and diesel to marine and coastal birds. **Table 13-1** bins species into bird groups, based predominantly on habitat preference (e.g., beaches, open water, marshes), foraging behavior (e.g., plunge diving for fish; intertidal and/or shallow subtidal feeding on invertebrate prey; tidal flat, rocky shoreline, and/or tide pool mollusk and/or infauna foraging).

Table 13-1. Avian species commonly inhabiting coastal and marine environments

Bird Groups	Common Habitats	Behavior and Diet	Species Examples
Pelagic Birds (seabirds)	Nearshore to offshore, open waters, large bays	On-water almost exclusively/rarely on-land, nest in colonies often remote from feeding sites, often on rocky shores, islands, islets; plunge-dive nearshore benthic foraging	Common murres, auklets, marbled murrelets, shearwaters, petrels, storm-petrels, pigeon guillemots, alcids, penguins, fulmars, boobies
Gulls and Terns	Ubiquitous: nearshore to offshore, open waters, large bays, rivers, beaches, marsh edge, islands	Diving (terns, skimmers), on-water, nesting and foraging on islands, beaches, loafing in marshes on beaches	Gulls, south polar skuas, shags, jaegers, terns, skimmers, phalaropes
Diving Birds	Nearshore to offshore, bays, rivers, marsh edge, islands	Plunge-diving, on-water, dense flock aggregations	Brown pelicans, grebes, cormorants, loons, gannets
Waterfowl (sea ducks, diving ducks, dabbling ducks)	Nearshore to offshore, bays, rivers, marshes, creeks	On-water, diving (some species), limited time on land (marshes, beaches); shallow subtidal and intertidal plunge feeding (sea ducks)	Sea ducks, scoters, scaup, common eiders, coots, mergansers, long-tailed ducks, canvasbacks, buffleheads, shovelers, teals, wigeons, shelducks, dabbling ducks
Shorebirds	Shorelines of beaches (sand or gravel), rocky shores, flats, marsh edge, nearshore waters	Foraging along waterline, on flats, in marshes, rocky shorelines, nesting in dunes, tundra; migratory	Piping plovers, western snowy plovers, sanderlings, dunlins, oystercatchers, curlews, godwits, sandpipers, avocets
Wading Birds	Beaches, marshes, creeks, estuaries, islands	Foraging and nesting in marshes, estuaries	Herons, egrets
Raptors	Ubiquitous, shorelines, beaches, marshes	Foraging along shorelines, on beaches, in marshes, along rivers	Peregrine falcons, hawks, harriers, bald eagles
Marsh Birds	Interior marshes, uplands, riverine and estuarine	Marshes, sometimes secretive	Rails, gallinules

The species composition of oiled and dead birds is often different than the overall composition of avian species in an oiled environment due to differences in habitat and foraging behavior. Foraging behavior and spill location and timing play key roles in the likelihood of certain species being exposed to oil, often regardless of spill size (Burger 1993). Relatively small spills in densely occupied areas can result in high bird mortality. Seabirds, diving birds, sea ducks, and penguins obtain much, or all, of their food from beneath the surface of the water and must frequently pass through the water's surface via repeated diving and surfacing (Lougheed et al. 2002). Therefore, when floating oil is present, they become fouled. Pelagic seabirds are especially vulnerable because they spend most of their life at sea, only returning to land to breed (O'Hara and Morandin 2010). Diving birds (e.g., double-crested cormorants, grebes, brown pelicans, loons) are common, piscivorous, and inhabit a wide array of habitats, including pelagic, coastal, and inland waterways (Cunningham et al. 2017). Diving birds are particularly vulnerable to spills when they are concentrated in coastal waters during nesting and wintering seasons.

13.2 Exposure Routes and Reponses to Oiling in Birds

13.2.1 Exposure Routes

Birds are exposed to oil through several potential routes, including adsorption, ingestion, inhalation, fouling, and aspiration (**Figure 13-1**). External contamination/fouling of feathers is the most common, and typically most devastating, form of exposure to birds and is the main cause of immediate mortalities of marine birds following oil spills (Leighton 1993). When feathers absorb oil, the plumage becomes matted and compressed. Oiling disrupts feather microstructure, causing collapse of the “zipping mechanism” of feathers (hooks, barbs, and barbules), which results in the loss of the feathers’ capacity to repel water (Paruk et al. 2020). When water penetrates to the skin of oiled birds, it results in heat loss; thus birds in cold water environments are known to be highly susceptible to hypothermia when their insulation is compromised due to feather oiling (Jenssen and Ekker 1991; O'Hara and Morandin 2010). Oiled feathers reduce buoyancy and flight capability (Leighton 1993). Oiled birds often die rapidly from hypothermia (regardless of water and/or air temperatures), starvation, and/or drowning (Lougheed et al. 2002; Paruk et al. 2020).

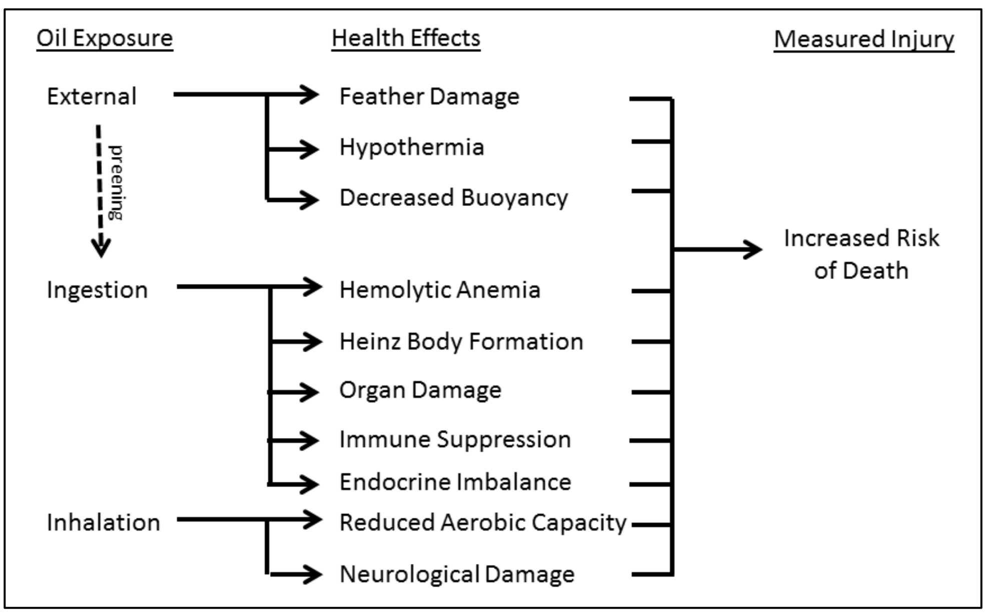

Figure 13-1. Potential routes of oil exposure and health effects for birds
From U.S. Department of the Interior in *Deepwater Horizon* NRDA Trustees (2014).

In addition to direct fouling, birds also may ingest oil when preening, consuming oil-contaminated food, water, or sediments, and potentially inhaling volatile compounds (Leighton 1993; NRC 2003). Burger (1997) found that the amount of time spent preening increased with the degree of oiling. Over 4,000 shorebirds (including 3,324 sanderlings) were exposed to oil as they foraged among stranded tar balls in the upper intertidal zone of the bayshore and coastal beaches following the *Anitra* spill of 952 bbl of Nigerian light crude (Anitra Oil Spill Natural Resource Trustees 2004). Oiling resulted in reduced weight gain during the period the birds spent in the Delaware Bay region, which was particularly problematic, as shorebirds need to feed heavily to be able to reach the Arctic and nest successfully (Burger 1997). Sanderlings are quite vulnerable to the effects of reduced weight gain caused by oiling (Burger and Tsipoura 1998). An estimated 4,000 waterbirds were oiled and died as a result of the *ARCO Anchorage* spill of 5,690 bbl Alaska North Slope crude oil in Port Angeles, Washington. The majority of impacted birds were estimated to be oiled within 1 to 10 km from shore, and most of the birds were incapacitated and sought shelter in the drift line of the beach while attempting to preen (Kittle Jr. et al. 1987).

The Texaco Oil Spills Natural Resources Trustees (2004) reported that birds were indirectly affected via ingestion of contaminated prey species in the intertidal and shallow subtidal habitats of Fidalgo Bay, Washington, as a result of the release of 714 bbl of Alaska North Slope crude oil. Recovered dead and oiled birds included diving ducks (bufflehead, mergansers) grebes, gulls, and shorebirds–all species that forage on shoreline invertebrates, nearshore subtidal invertebrates, and shallow nearshore fishes. Consumption of contaminated prey can lead to accumulation of oil in birds, and potential toxic effects of ingested oil are wide ranging and include hemolytic anemia, gastrointestinal irritation and hemorrhage, as well as liver and kidney disorders (Alonso-Alvarez et al. 2007; Balseiro et al. 2005; Leighton et al. 1983; NRC 2003; Paruk et al. 2016; Seegar et al. 2015; Yamato et al. 1996).

Though less is published about oil inhalation as an exposure pathway, Hughes et al. (1998) found that pulmonary congestion and pneumonia, both potential manifestations of oil inhalation resulting in severe inflammation of the respiratory tract, were observed in 43% of sampled birds during the *Sea Empress* spill (which released 490,000 bbl of Forties blend crude oil; thus not included in this synthesis).

13.2.2 Bird Mortality Following Spills of 500 to 20,000 bbl Crude Oil, Condensate, or Diesel

Impacts to birds are well documented from coastal and oceanic oil spills (**Table 13-2**). Median-range spills with acute losses to pelagic birds include *Apex Houston*, Terra Nova blowout, *American Trader*, *ARCO Anchorage*, and *Julie N*. Median-range spills with high losses, often hundreds to thousands, of diving birds and/or sea ducks, include *American Trader*, *ARCO Anchorage*, and *North Cape*. Of the 29 median-range spills reviewed for this chapter, species most impacted included gulls (13 spills), diving birds (12 spills), shorebirds (11 spills), pelagic birds (10 spills), and waterfowl (9 spills). Note that literature with detailed counts and species information was not available for 11 of the 29 spills reviewed.

Table 13-2. Spills of crude oil, condensate, or diesel-like oils (500–20,000 bbl) with information on impacts to marine and coastal birds

Oil Spill	Oil Volume and Type	Summary of Documented Bird Impacts (bird groups impacted and estimated mortality when available*)
1990 T/V *American Trader*, Huntington Beach, CA[a]	9,919 bbl Alaska North Slope crude	Estimated 3,400 birds killed and reproductive impacts likely. 95% of oiled birds were sea ducks, surf scoters, pelicans, grebes, gulls, cormorants, loons, murres, auklets, murrelets, shearwaters, and petrels.
1996 T/V *Anitra,* Delaware River[b]	952 bbl Nigerian light crude	At least 53 ESA-listed piping plovers oiled during nesting season; 4,000 lightly to moderately oiled shorebirds.
1986 T/B *Apex Houston*, offshore Marin, San Francisco, San Mateo, Santa Cruz, and Monterey counties, CA[c]	615 bbl crude	Estimated over 9,900 birds killed. At least 26 species were impacted, including thousands of common murres and rhinoceros auklets, hundreds of loons, grebes, Cassin's auklets, and scoters, and 12 ESA-listed marbled murrelets. Losses and

Oil Spill	Oil Volume and Type	Summary of Documented Bird Impacts (bird groups impacted and estimated mortality when available*)
		declines recorded at common murre breeding colonies.
1985 T/V *ARCO Anchorage,* Port Angeles, WA[d]	5,690 bbl Alaska North Slope crude	Nearly 2,000 birds known to have been oiled, with an estimated 4,000 likely oiled, including over 1,100 grebes, 475 ducks, 225 alcids, cormorants, gulls, loons, and shorebirds.
1992 UNOCAL tank farm, Avila Beach, CA[e]	600 bbl San Joaquin Valley crude	At least 77 marine birds killed, including 28 alcids, 20 shearwaters, cormorants, shorebirds/gulls, and 4 California brown pelicans (ESA-listed at the time).
1989 T/V *Bahia Paraiso*, Arthur Harbor, Antarctica[f]	3,760 bbl diesel fuel arctic	Population-wide mortality of south polar skua chicks hypothesized to be due to sublethal oiling of adults thus temporarily disrupting parental guarding of chicks. Loss of shag nestlings to toxicity and abandonment. Adelie penguins and seabirds exposed to the spill while feeding on krill and fish.
2004 BP MP-80 Delta 20 and Nakika MP-69 pipeline Hurricane Ivan oil spills, LA[g]	7,058 bbl South Louisiana crude	Estimated 25–500 birds injured from Ivan-related spills.
1988 T/V *El Omar,* Milford Haven, Wales[h]	670 bbl light Iranian crude	326 birds oiled, including 132 shorebirds, 119 gulls, and 75 waterfowl.
1991 El Segundo offshore Chevron pipeline, Santa Monica Bay, CA[i]	500 bbl diesel-like oil	15 dead oiled birds; 6 live oiled birds.
1998 Equinox well blowout, Lake Grande Ecaille, LA[j]	1,535 bbl South Louisiana crude	95 birds estimated killed.
1990 Exxon Bayway Refinery, Arthur Kill, NY[k]	13,715 bbl No. 2 fuel oil	700 birds killed, including 300 waterfowl and 300 gulls.
1991 Texaco Refinery, Fidalgo Bay, WA[l]	714 bbl Alaska North Slope crude	145 dead oiled birds recovered, including waterfowl, shorebirds, grebes, and gulls. Estimated 300 birds killed. Indirect effects occurred via ingestion of contaminated of prey species in the intertidal and shallow subtidal habitats.
1969 T/B *Florida*, West Falmouth, MA[m]	4,385 bbl No. 2 fuel oil	Birds suffered high mortality immediately after the spill.
2001 T/V *Jessica*, Galapagos Islands, Ecuador[n]	2,800 bbl diesel and 2,160 bbl bunker fuel	138 oiled birds, including pelicans (100s), shearwaters, boobies, gulls, petrels, herons, and owls.
1996 *T/V Julie N,* Portland, ME[o]	2,058 bbl home heating fuel and 2,219 bbl IFO 380	1,679 oiled birds, including 1,345 gulls, 155 double-crested cormorants, 70 shorebirds (mostly black-bellied and semipalmated plovers), 52 black ducks, and 23 great blue herons.
1997 Texaco pipeline, Lake Barre, LA[p]	6,561 bbl South Louisiana crude	Trustee model estimated 333 birds lost; Texaco estimated 100 birds lost. 58 live oiled birds observed: clapper and king rails, gulls, great and snowy egrets, plovers, sandpipers, and herons.
1999 T/V *Laura d'Amato*, Sydney Harbor, Australia[q]	1,750 bbl Murban light crude	16 oiled birds caught and cleaned; 2 cormorants died. Other oiled birds sighted.
1972 Long Island Sound[r]	1,905 bbl No. 2 fuel oil	28 dead oiled birds reported.

Oil Spill	Oil Volume and Type	Summary of Documented Bird Impacts (bird groups impacted and estimated mortality when available*)
1993 McGrath Beach/Berry Petroleum, CA[s]	2,075 bbl crude	166 dead birds collected; actual mortality was estimated to be at least 20% higher. Apparent loss of one breeding season; least tern, snowy plover, avocet, and other shorebirds nest on the beach.
1996 T/B *North Cape*, Moonstone Beach, South Kingstown, RI[t]	19,700 bbl home heating oil	405 oiled birds recovered on beaches (392 died), estimated over 2,100 marine birds killed including loons, eiders, gulls, mergansers, goldeneyes, grebes, cormorants, dabblers, murres, herons/egrets, and gannets. Piping plover (ESA) impacted by food base reductions.
2017 Refugio Beach, CA[u]	500 bbl Monterey crude oil	At least 269 birds were oiled, including brown pelicans, common murres, loons, shearwaters, cormorants, gulls, grebes, northern fulmars, scoters, and auklets. Western snowy plovers: spike in percentage of infertile eggs in year following the spill likely due to oil ingestion via preening and feeding.
1997 Rose Atoll (*Jin Shiang Fa*), American Samoa[v]	2,380 bbl diesel and lube oil	The effects on seabirds were unknown and assumed to be minimal. No birds exhibited obvious signs of fuel contamination, but none were captured and examined.
1992 T/B *RTC-380*, Long Island Sound, NY/CT[w]	524 bbl diesel	4 dead and 12 oiled birds found.
2004 Terra Nova blowout, Grand Banks, Canada[x]	1,000 bbl crude	Estimated 10,000 seabirds (murres and dovekies) were killed.
1971 Texaco March Point Refinery, Guemes Island, WA[y]	4,700 bbl diesel oil	Estimated 1,000 dead birds.
1997 Torch platform, Offshore Santa Barbara, CA[z]	163 bbl crude	Estimated 635 to 815 birds impacted. 140 oiled birds recovered, including: common murres, cormorants, grebes, sanderlings, loons, shearwaters, phalaropes, and brown pelicans. Additional reports of oiled birds included brown pelicans and snowy plovers.
1984 TROPICS field oiling study, Panama[aa]	6 bbl Prudhoe Bay crude	Death of birds from 0 to 15 days.
Uniacke G-72, blowout off Sable Island, Nova Scotia, 1984[bb]	1,500 bbl gas condensate	Some oiled birds were sighted.
1989 T/V *World Prodigy*, Narragansett Bay, RI[cc]	6,900 bbl home heating oil	13 oiled birds recovered, including gulls, loons, storm petrels, and cormorants.

[a]American Trader Trustee Council (2001); [b]Anitra Oil Spill Natural Resource Trustees (2004); [c]Apex Houston Trustee Council (2011); Carter et al. (2003); [d]Kittle Jr. et al. (1987); [e]CDFG and USFWS (1999); [f]Eppley (1992); [g]RPI (2005); [h]Little et al. (1990); [i]NOAA (1991); [j]LOSC et al. (2005); [k]Louis Berger & Associates Inc. (1991); [l]Texaco Oil Spills Natural Resources Trustees (2004); [m]Sanders et al. (1980); [n]Lougheed et al. (2002); [o]Mauseth and Csulak (2003); [p]Lorentz et al. (2001); [q]Incident Analysis Team 2000; [r]EPA (1973); [s]CDRP, CDFG and USFWS (2005); [t]NOAA (1999); [u]Nielsen et al. (2018); [v]USFWS (1997); [w]NOAA (1994); [x]Wilhelm et al. (2007); [y]Chia (1971); [z]Ford and Bonnell (1998); [aa]Baca et al. (2014); [bb]Gill et al. (1985); [cc]Pilson (1990)

*Note: Individual species counts of impacted birds are examples of key species oiled and are not intended to sum to total (or estimated) numbers of birds impacted per spill.

13.2.3 Acute Oiling Phase and Likely Impacts to Birds

A marine bird's ability to thermoregulate is essential to survival. Fouled birds typically have behavioral changes such as increased time spent preening and changes in foraging patterns, including either increased or decreased food consumption. The consequences of increased energetic demands to maintain body temperature while birds are under distress have the potential to cause weight loss, interfere with reproduction, affect the immune response, and prevent optimal body condition for migration (Dorr et al. 2019; Harr et al. 2017; Mathewson 2018; Cunningham et al. 2017). Inhalation of volatile compounds can cause alterations in neurological and respiratory function that may result in behavioral modifications and constraints on birds' abilities to fly, swim, and dive (Helm et al. 2015). Physiological effects also can result from absorption of toxic components of oil through the skin (Ziccardi 2015).

A key finding following the *Sea Empress* spill (a large crude oil spill of 490,000 bbl of Forties blend crude oil; thus not included in this synthesis), which impacted over 3,300 scoters, was that most of the heavily oiled birds died quickly from hypothermia, drowning, or being smothered and/or encased in oil (Hughes et al. 1998). In contrast, most partially oiled birds survived up to a week and then died from oil ingestion, damage to gut lining, or secondary infections due to immunosuppression. It is common to observe a lag in the deposition of dead, oiled birds on beaches. Oiled birds, although hypothermic and unable to forage, may take days to die, as many species do not come ashore or are difficult to capture until they are near death.

Of the twenty-nine median-range spills for which there was literature available on bird impacts, all reported acute impacts within days to weeks of the initial event. There were eight spills for which there was detailed literature on larger bird impacts, defined for the purposes of this review as hundreds to thousands of injured birds and/or impacts to any number of federal ESA species. The impacts are outlined in **Table 13-3**.

California brown pelicans (ESA-listed at the time) were severely impacted by the *American Trader* spill of 9,919 bbl Alaska North Slope crude off of Huntington Beach, California, with an estimated 185 dead birds. The Trustees estimated that half of the 750 to 1,000 pelicans roosting at the breakwater, the principal pelican roost in the area, were oiled (American Trader Trustee Council 2001).

California brown pelicans (73 oiled and collected), murres, shearwaters, fulmars, loons, and cormorants were impacted by the Refugio Beach spill of 500 bbl of Monterey crude oil onto Santa Barbara beaches and waters (Refugio Beach Oil Spill Trustees 2020). Brown pelicans typically forage in relatively shallow coastal waters (within 30 km of shore), feeding almost entirely on surface-schooling fish caught by plunge diving. During the non-breeding season, brown pelicans roost communally on offshore rocks and structures such as piers and wharfs. These two spills show that species that gather at roosts before moving to breeding islands bird are vulnerable to oiling in large numbers.

The *Apex Houston* spill of 615 bbl of crude oil that extended offshore from San Francisco to Monterey killed over 9,000 birds, with greatest mortality to common murres (6,300), rhinoceros auklets (1,300), and 1,400 other alcids, loons, grebes, scoters, cormorants, shorebirds, and gulls (Carter et al. 2003). Most carcasses and live oiled birds beached in the first 5 days (Page et al. 1990). Murres are a highly social species and forage in coastal waters on schooling fish and krill. Twelve ESA-listed marbled murrelets, a seabird that dives for small fish in nearshore waters within 5 km of the coastline, were recovered oiled and dead.

Table 13-3. Notable spills of crude oil, condensate, or diesel (500–20,000 bbl) with acute or reproductive impacts to birds

Bird Groups	Notable Median Spills with Acute Impacts (Mostly Lethal)	Notable Median Spills with Reproductive Impacts
Pelagic Birds	*Apex Houston*: Common murres (1000s), rhinoceros auklets (1000s), Cassin's auklets (100s), 3–11 ESA-listed marbled murrelets[a] *American Trader*: 1000s of sea birds (murres, auklets, murrelets, shearwaters, and petrels)[b] *ARCO Anchorage*: 225 alcids[c]	*Apex Houston*: Losses and declines at common murre breeding colonies[d].
Gulls and Terns	Julie N: 1,345 gulls[e]	
Diving Birds	*American Trader*: 185 California brown pelicans (ESA listed at the time of the spill) died, 100s oiled, western grebes (100s), cormorants (100s)[b] *Apex Houston*: Loons (100s), grebes (100s)[a] *ARCO Anchorage*: 1,100 grebes, 40 cormorants, 14 loons[c] *Jessica*: Pelicans (100s)[f] *Julie N*: 155 double-crested cormorants[e] Refugio: 73 brown pelicans, 36 loons, 16 cormorants, 11 grebes[g] *North Cape*: estimated 414 loons[h]	*American Trader*: Reproductive impacts likely to brown pelicans[i]. North Cape: Recovery time for loons and grebes was determined to continue beyond the first breeding season[j].
Waterfowl (sea ducks, diving ducks)	*American Trader*: Sea ducks, surf scoters (1000s)[b] *Apex Houston*: Scoters (100s)[a] *ARCO Anchorage*: 475 ducks[c] Exxon Bayway: 300 waterfowl[k]	North Cape: Recovery time for sea ducks was determined to continue beyond the first breeding season[j].
Shorebirds	*Anitra*: Piping plovers (ESA-listed; 51 adult and 2 chicks) oiled during the nesting season[l]	*Anitra*: Piping plover oiling contributed to reduced nesting success and lowered productivity on affected beaches[l]. Refugio: Western snowy plover reproductive effects – high rate of infertility the year following the spill[g]. *North Cape*: Piping plover productivity dropped the year of the spill[j]. McGrath Beach/Berry Petroleum: Loss of one nesting season for beach nesting terns and shorebirds[m].

[a]Carter et al. (2003); [b]American Trader Trustee Council (2001);[c]Kittle Jr. et al. (1987); [d]Carter et al. (2003); [e]Mauseth and Csulak (2003); [f]Lougheed et al. (2002); [g]Refugio Beach Oil Spill Trustees (2020); [h]Evers et al. (2019); [i]Anderson et al. (1996); [j]NOAA (1999); Evers et al. (2019); [k]Louis Berger & Associates, Inc. (1991); [l]Anitra Oil Spill Natural Resource Trustees (2004); [m]CDP&R, CDF&G, USFWS (2005)

High mortality of grebes (at least 1,118) and ducks (at least 475) occurred within a short time following the *ARCO Anchorage* spill of 5,690 bbl Alaska North Slope crude into Port Angeles, Washington, likely due to these species of diving birds and ducks being exposed to oil slicks on the water surface (Kittle Jr. et al. 1987).

13.2.4 Post-acute Oiling Phase and Likely Impacts to Birds

Oil exposure at less than acutely lethal doses can lead to a wide array of adverse impacts that slow population recovery. Physiological impairments and health effects include: feather damage, reduced flight capability, inflammation, immune system suppression/ increased rates of disease, cellular damage, altered organ function, liver damage, gastrointestinal damage, changes in the salt gland, reduced reproductive success, hemolytic anemia (which compromises the ability of the blood to carry oxygen), decreased nutrient absorption, weight loss, impaired osmoregulation, endocrine disruption, and altered stress response (Alonso-Alvarez et al. 2007; Briggs et al. 1996; Burger and Tsipoura 1998; Fry et al. 1987; Golet et al. 2002; Leighton 1993; Troisi et al. 2007).

13.2.5 Reproductive Impacts on Oiled Birds

Reproductive effects on oiled birds include adverse hormone changes, delayed egg laying, impaired egg formation, decreased eggshell thickness, lowered fertility, reduced hatchability, and abandonment of reproductive effort (Haycock et al. 1998; Sharp et al. 1996; Golet et al. 2002). Oil brought back to nests can reduce hatching and fledging success. Avian embryos, especially very young ones, are highly sensitive to oil that contaminates the eggshell; amounts as little as 1–10 microliters may result in eggs failing to develop (Leighton 1993; NRC 2003). Oil can be transferred to the surface of eggs when adults build nests with oil-contaminated materials or when adults carry the oil back to the nest on contaminated feathers and incubate. Reduced prey availability can lead to declines in breeding success.

California brown pelicans (ESA-listed at the time of the spill) were exposed to oil from the *American Trader* spill just prior to the 1990 breeding season in the Southern California Bight. Anderson et al. (1996) found that oiled and rehabilitated pelicans showed no breeding activity (or even presence or association with breeding colonies) in 1990 or 1991. In contrast, unoiled pelicans were active at breeding colonies in 1990 and 1991. Rehabilitated pelicans tended to remain sedentary and farther away from the breeding colonies than unoiled pelicans in their second post-rehabilitation breeding season. Anderson et al. (1996) suggested that rehabilitated pelicans that survived for 6 months were no longer contributing members of the breeding population.

ESA-listed piping plovers nest on the upper beach on bayshore and coastal barrier beaches oiled during the *Anitra* spill of 952 bbl Nigerian light crude in Delaware Bay. Plovers were exposed to oil as they foraged among stranded tar balls in the upper intertidal zone, and at least 51 adult piping plovers and two chicks were oiled during the 1996 nesting season. The Trustees predicted that the oiling directly or indirectly contributed to reduced nesting success and ultimately lowered productivity on affected beaches (Anitra Oil Spill Natural Resource Trustees 2004). Burger (1997) suggested a decreased subsequent breeding success in the Arctic for sanderlings and semipalmated plovers heavily oiled by the *Anitra* spill.

Common murres are a highly social species that breed in large, dense colonies on offshore rocks, islands, and cliffs. Murres have high fidelity to their breeding sites and typically return to breed at the natal colony. During the time of the *Apex Houston* spill, the central California common murre population was in sharp decline from a variety of factors. The *Apex Houston* spill contributed to the decline of the colonies at Castle Rocks and Mainland and Hurricane Point Rocks, as an estimated 6,300 murres were killed by the oil spill. After the oil spill, the Devil's Slide Rock breeding colony, which held close to 3,000 breeding murres in the early 1980s, was extirpated (Carter et al. 2003).

Nesting marine birds were censused following the *ARCO Anchorage* spill. Double-crested cormorant, pelagic cormorant, and pigeon guillemot reproduction appeared normal at Protection Island, and marbled murrelet numbers reproduction appeared normal in survey areas as well (Kittle Jr. et al. 1987).

A Berry Petroleum Company pipeline ruptured and discharged 2,075 bbl of crude oil impacting McGrath Lake, the Pacific Ocean, and 11 km of sand beach. There was an apparent loss of one breeding season for least tern, snowy plover, avocet, and other shorebirds (CDPR, CDF&G, USFWS 2005).

Following the *North Cape* spill of 19,700 bbl of home heating oil onto Moonstone Beach, Rhode Island, piping plover productivity dropped 37% in 1996, after steadily increasing for the previous 5 years, while productivity at Rhode Island reference area sites increased 6% (Casey 1996). Moonstone Beach piping plovers exhibited unusual behavior consistent with reduced food supply and reproductive impairment. The Trustees concluded that the local piping plover population was unlikely to naturally recover from this loss (NOAA 1999). Piping plovers were also impacted by food base reductions following the spill. In addition, recovery time for sea ducks, loons, and grebes was determined to continue beyond the first breeding season following the spill.

The 2015 Refugio Beach spill occurred during the western snowy plover breeding season, with many chicks recently hatched and foraging on sandy beaches. The year following the spill, the rate of egg infertility was 9.3%, more than four times the long-term average (2001–2015). In 2017, infertility was within the range of natural variation observed prior to the spill. Over half the plovers observed had oil on their body or beak within the first 6 weeks following the spill; the oil was likely removed due to preening, with the birds ingesting 50% of the oil (Hartung 1963). Nielsen et al. (2018) found a plausible connection between ingestion of oil via preening and diet and the increase in the rate of infertile eggs in 2016 at Coal Oil Point. It is unlikely that oil ingestion affected infertility in 2015 because over 70% of the season's nests were already established by the time oil reached the beach. The effects manifested a year later, the first time most birds had laid eggs since the spill. A delayed effect of oiling on infertility rates has been documented in other spills and for other species (Fry et al. 1987).

Eppley (1992) documented population-wide mortality of south polar skua chicks following the *Bahia Paraiso* spill of 3,760 bbl of diesel fuel arctic in Antarctica, hypothesized to be due to sublethal oiling of adults thus temporarily disrupting parental guarding of chicks. Shag nestlings were also lost to toxicity and abandonment.

13.3 Time to Recovery for Birds Following Spills of 500–20,000 bbl of Crude Oil, Condensate, or Diesel

For the majority of median-range spills examined for this literature review, the only documented data were estimates of oiled, mostly deceased, birds that were presumably observed or collected within days to weeks, or at most months, following the incidents. Short-term impacts, or those with recovery within 1 year are driven primarily by acute oil exposure and death due to fouling (e.g., cormorant and guillemot colonies following the *ARCO Anchorage* spill).

There are also several examples of breeding populations that were depressed the year of or following a spill and rebounded by year 2 post spill, as discussed in the above section on reproductive impacts, including south polar skuas (*Bahia Paraiso*), piping plovers (*Anitra*), dabbling ducks and grebes (*North Cape*) (**Table 13-2, Figure 13-2**). Examples of breeding populations with estimated recovery of >3 years include the *North Cape* spill of home heating oil where sea ducks (e.g., scoters, mergansers, goldeneye, bufflehead, eiders) recovered in 1–3 years, piping plovers recovered > 3 years, and loons recovered in 6 years. The two median-range spills with the longest estimated recovery times were the *American Trader*, after which California brown pelicans that were oiled, rehabilitated, and released were considered no longer viable members of the breeding population 11 years post spill, and the *Apex Houston*, where already tenuous common murre breeding colonies were extirpated following the spill. Restoration efforts from the *Apex Houston* settlement resulted in common murre colonies re-establishing and re-colonizing to numbers of breeding pairs consistent with recovery goals within 10–14 years following the spill.

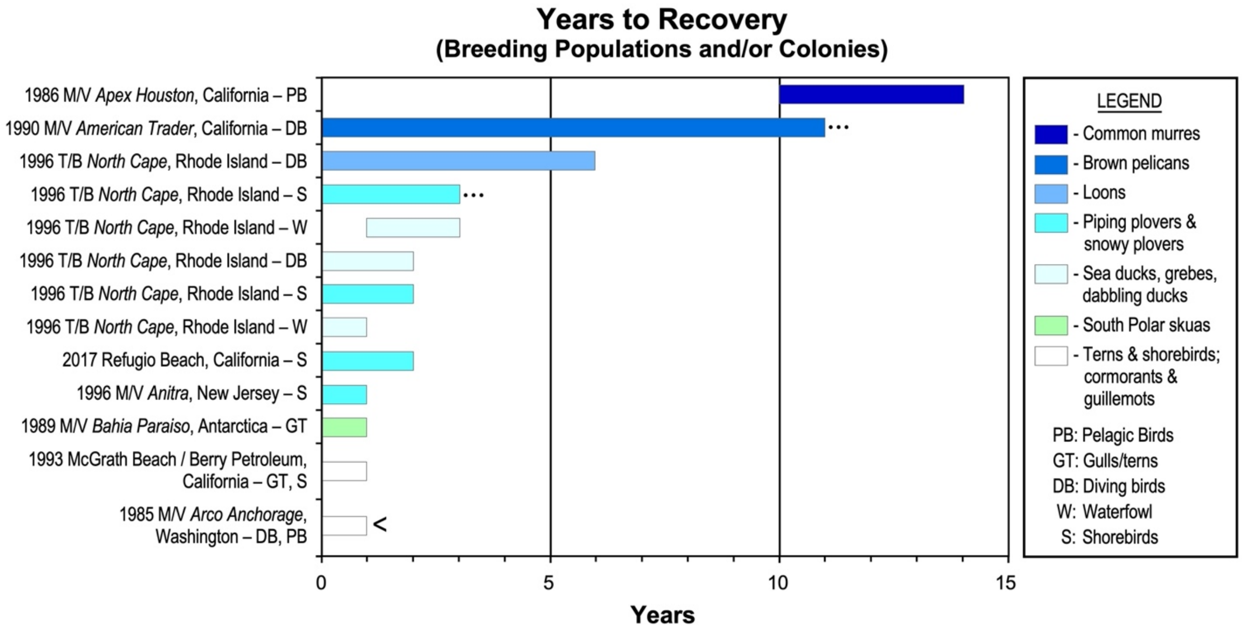

Figure 13-2. Years to recovery for breeding bird populations and/or colonies for spills of crude oil and diesel-like oils (500–20,000 bbl)

Short-term recovery of bird populations following marine spills are based on a number of factors, including initial mortality, fecundity rate, breeding success, prey availability, and foraging habitat impacts. Estimating the long-term significance of sublethal exposure to oiling is difficult without knowing the recovery rates of individual species.

13.4 Summary and Information Needs for Assessing Impacts to Birds

Marine oil spills often affect a wide range of habitats and thus a vast number of bird species that use these habitats for feeding, nesting, migration, and overwintering. Marine and coastal birds are often oiled and experience high acute mortality because their habitat use (e.g., rafting, foraging, nesting, plunge-diving, chick-rearing, scavenging) overlaps nearly completely with habitats most likely to be oiled and where oil tends to persist. As was clear in this literature synthesis, spatial and temporal use of potentially oiled habitats, and particularly behavior, all directly influence vulnerability of different species groups of birds to oiling and to the likelihood of birds suffering short- and long-term impacts. Seabirds (e.g., murres, murrelets), diving birds (e.g., pelicans, cormorants), sea ducks (e.g., scoters, scaup, eiders), and shorebirds (e.g., piping plovers, western snowy plovers) were the predominant casualties of oil spills examined in this review. Mortality was primarily acute and due to fouling and ingestion of oil by preening.

Spills that occurred immediately prior or during nesting season for beach nesting and colonial nesting birds tended to have both acute and longer-term (1–2 nesting seasons or more) impacts. Reasons for this included the tendency for diving birds and seabirds, such as pelicans and common murres, to congregate in large numbers, nearshore, either on-water or at roost sites, prior to active breeding at colonies. Shorebirds with immediate or delayed impacts (1–2 seasons) on nesting beaches tended to be impacted while foraging on impacted beaches. Timing was also a major factor for the *Bahia Paraiso* spill in

Antarctica, as sublethal oiling of adults disrupted parental guarding of polar skua chicks, similar to the loss of shag nestlings to abandonment.

The *Apex Houston* spill is an interesting case, as the amount of spilled oil was relatively small (615 bbl) and was spread over a large area off the coast of California. Large numbers of birds were oiled and died, undoubtedly due to large numbers of very vulnerable species of seabirds (common murres and rhinoceros auklets) and diving birds (loons and grebes) congregating on-water where the slicks also occurred. The *American Trader* and *ARCO Anchorage* both also had heavy losses of marine birds (e.g., sea ducks, seabirds, grebes, pelicans), but the volume of oil released for those two spills was an order of magnitude higher than for the *Apex Houston* and impacted smaller areas.

Of the spills discussed in detail above, most were crude oil spills. In contrast, the *Bahia Paraiso* (loss of skua chicks) was arctic diesel, and the *North Cape* was home heating oil. The *North Cape* was a unique case where the barge grounded on Moonstone Beach, releasing 19,700 bbl of home heating oil in cold water during the winter. Birds impacted by the *North Cape* included species on the beach, in Block Island Sound, and within coastal salt ponds, which are extremely important habitats for birds as well as their prey items (e.g., invertebrates and fish).

Despite a large body of evidence with very detailed discussions on exposure pathways and how oil negatively impacts individual birds, few species have been studied extensively enough to predict the likelihood of population level and/or long-term effects. Pre-spill surveys are essential for post-spill impact comparisons and were carried out for some species in some locations, such as western snowy plovers at Coal Oil Point, for the Refugio Beach spill. Most median-range spills were not studied as intensively or for as long as some large spills, such as *Exxon Valdez* or *Deepwater Horizon*, so the long-term understanding of recovery for most species in most habitats is somewhat limited. However, surface oiling where birds concentrate and raft during critical wintering or migratory periods, extensive oiling of intertidal or subtidal prey items (particularly if species foraging in the spill zone have very limited diets), and persistent surface slicks where nearshore plunge-divers inhabit coastal waters may be expected to have greater impacts (Michel 2021). Alternatively, it appears that many colonial-nesting seabirds, diving birds, terns, and shorebirds are resilient at the population level, and counts at impacted breeding locations tended to rebound within 3 years, with a few exceptions. Continued surveying of oiled populations is critical to understanding of oil spill effects on birds. However, environmental factors such as climate change, weather, annual variation, and various other stressor makes it very difficult to relate oil spill impacts from past spills to current population assessments.

14 Sea Turtles

14.1 Resource Description and Habitats

This chapter addresses the impacts from spills of 500–20,000 bbl of crude oil, condensate, or diesel and spill response activities on sea turtles and their habitats throughout their life stages. These species utilize terrestrial, estuarine, nearshore, and offshore marine environments during their life and are found throughout most of the world's temperate and tropical waters. Five of the seven sea turtle species in the world can be found in waters of the U.S. OCS–loggerhead (*Caretta caretta*), green (*Chelonia mydas*), Kemp's ridley (*Lepidochelys kempii*), hawksbill (*Eretmochelys imbricata*), and leatherback (*Dermochelys coriacea*). The olive ridley sea turtle (*Lepidochelys olivacea*) resides solely in the Pacific Ocean and inhabits waters of U.S. territories. All sea turtles that inhabit U.S. waters are listed as endangered or threatened under the ESA.

Though most of sea turtles' lives are spent at sea, nesting occurs on sand beaches around the tropical and sub-tropical waters of the world. **Figure 14-1** shows the general life history of sea turtles. Every 2–4 years, adult female sea turtles will return to their natal beach, choose a location to dig a nest, and deposit a clutch of eggs, potentially repeating these steps several times over the course of a season, and then return to the neritic zone (continental shelf) (Bolten 2003). The eggs are left unattended to incubate in the nest. Moisture capture, oxygen exchange, and heat exchange occurs across the eggshell of the turtle and between the nest and the interstitial pores in the sand, providing water, oxygen, nutrients, and warmth necessary for the embryos to develop into hatchlings. Once hatchlings emerge from the nest, they cross the beach and swim to the open ocean, often seeking refuge in *Sargassum* habitats on the way to oceanic waters. As a surface-pelagic juvenile, the sea turtle may remain in the oceanic and convergence zones for years before moving to the neritic zone, where they will continue to feed, live, and grow for the remainder of their life. The time they emerge from the nest to the time that females return to nest was called the "lost years" because little was known about where they went and how they survived (Skwarecki 2014). Only recently, through improvements and expansion of satellite tracking, has it been possible to more closely track the movements of sea turtles (Hays and Hawkes 2018).

Pelagic *Sargassum* is a floating macroalgal community with the majority of drifts occurring within the western North Atlantic, Caribbean Sea, and Gulf of Mexico between latitudes of 20 and 40º N. *Sargassum* habitats are known to be highly productive patches and associated with thousands of animals (Dooley 1972; Butler et al. 1983). Witherington et al. (2012) examined *Sargassum* habitat to determine how important this habitat is to juvenile sea turtles. From 1992 to 2004, they observed 1,884 turtles of four species in the Atlantic Ocean and Gulf of Mexico. Data from observing three juvenile Kemp's ridleys showed that they spend 97% of the day and 87% of the night within 1 m of the surface (Witherington et al. 2012). Adults spend less time at the surface and more time diving deep into open-water habitats. Convergence zones can serve as natural collection points not only for *Sargassum* but also oil and surface debris. Surface-pelagic juvenile turtles prefer these collection points, which provide a food source and protection from predation; however, their presence in this zone can increase their exposure to oil during a spill. During the *Deepwater Horizon* spill, most reports of oiled surface-pelagic juvenile turtles were from *Sargassum* in convergence zones (NOAA 2016a).

Adult sea turtles can inhabit bays, sounds, rivers, estuaries, and deep oceanic waters. Resurfacing intervals are typically every 15–20 minutes; however, they may remain underwater for up to an hour. Adult leatherback turtles have been known to dive to depths up to 1,000 m.

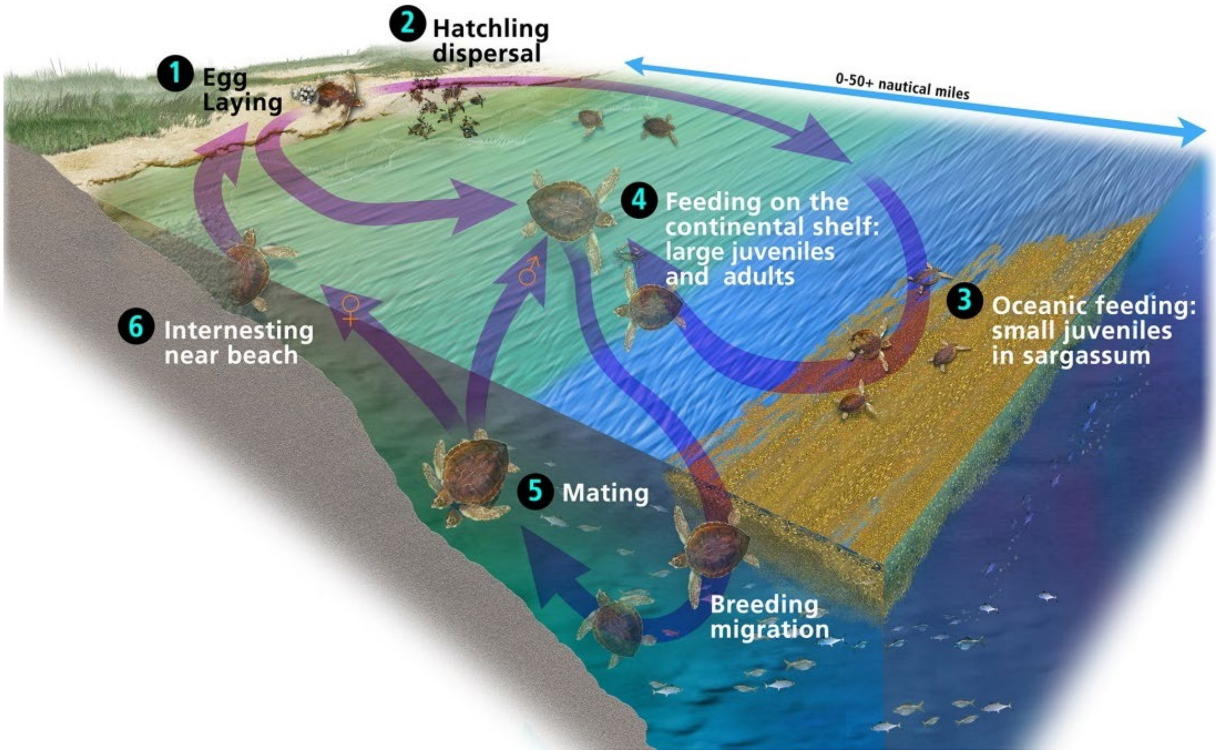

Figure 14-1. Generalized sea turtle life cycle (NOAA 2016a)

Federal and international regulations are important for the protection of sea turtles due to the large threat humans have been to sea turtles around the world. The collection and sale of sea turtle eggs remains a threat primarily in Central America and the Caribbean; their skins have been exploited to make articles of clothing such as boots and belts; and their shells, especially the hawksbill, have been exploited for commercial products such as combs, brushes, jewelry, musical instrument picks, and furniture inlays (USFWS 2007). The direct killing of sea turtles and their eggs has contributed heavily to the decrease in sea turtle populations. Sea turtles within U.S. waters are managed by USFWS when on land and National Marine Fisheries Service (NMFS) when at sea.

Humans have been indirectly harming sea turtles by coastal development, artificial lighting on beachfront communities (disorients nesting females and hatchlings), bycatch in recreational and commercial fisheries, and marine debris that can entangle sea turtles causing them to drown, become immobilized and stuck at the surface where they could be hit by vessels or predators, or unable to forage (USFWS 2007). The Tafira Wildlife Rehabilitation Center (TWRC) in Gran Canaria Island, Spain analyzed results of 1,860 admitted loggerhead turtles to TWRC from 1998 to 2014 and found that 71% of the animals were stranded due to anthropogenic causes including entanglement in fishing gear and/or plastics and ingestion of hooks and monofilament (Orós et al. 2016). Other factors such as climate change, prey availability, algal blooms, disease, and reproductive failures influence sea turtle populations.

14.2 Pathways of Oil Exposure to Sea Turtles

Sea turtle exposure to oil depends on the oil spill characteristics and varies by life stage and habitat. **Figure 14-2** shows where sea turtles can be exposed to oil and potential impacts of oil on sea turtles.

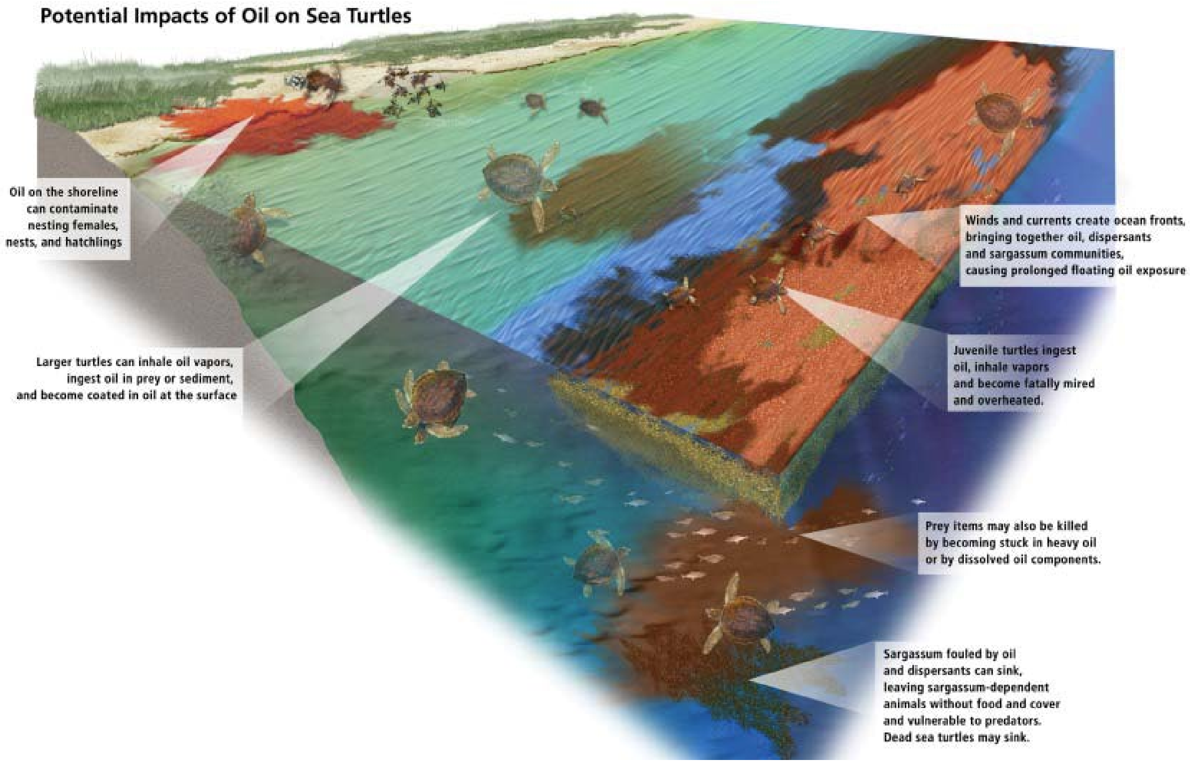

Figure 14-2. Potential impacts of oil from the *Deepwater Horizon* oil spill on sea turtles in the northern Gulf of Mexico (NOAA 2016a)

Physical fouling of eggs, hatchlings, and nesting females can occur on oiled beaches. Inhalation of oil may impair the olfactory gland affecting a sea turtle's sense of smell (Frazier 2005). The sense of smell plays a key role in sea turtle navigation and orientation, and interfering with that sense can reduce the ability of individual turtles and populations to successfully navigate along migration routes or to natal nest beaches (Frazier 2005). Frazier (2005) also noted that oil on a nesting beach could potentially affect the locational imprinting on hatchlings of their natal beach, impairing their ability to return to breed and nest in the future. Unknowingly ingesting tar balls or contaminated food is a direct effect of an oil spill; however, reduced food availability is an indirect effect that can lead to a decline in local sea turtle populations. Seagrasses damaged due to an oil spill, or die-off of prey within the seagrass beds, can cause reductions in food availability.

14.3 Impacts of Oil Exposure and Response Actions to Sea Turtles

Field and laboratory studies on sea turtles following oils spills are limited due to the inability to track live injured or oiled animals, and legal and ethical considerations regarding exposing living animals to oil in a laboratory experiment.

Physical fouling by oil is the most frequently reported effect of oil exposure on sea turtles (Wallace et al. 2020; Shigenaka et al. 2010). Oil covering sea turtles at any life stage can cause smothering, clogging the mouth and nose, or creating an inability to manuever. Oil contact can cause acute toxicity in hatchlings and impair their movements and normal bodily functions if coated (Shigenaka et al. 2010). At sea, juvenile and adult sea tutles can be weighed down by oil, obstructing their ability to surface for air, reducing their ability to dive for food, avoid predators or vessel strikes, and regulate body temperature. Ingesting oil either directly (i.e., ingesting tar balls) or indirectly (i.e., consuming contaminated foods) can cause acute toxicity or, in terms of tar balls, can lead to blockage of their mouths or esophageal pathways.

Loehefener et al. (1989) found tar balls in the mouths, esophagi, and stomachs of 65 out of 103 post hatchling loggerhead turtles off the east of Florida in a convergence zone. Witherington (1994) studied post hatchlings in convergence zones, or weed lines, off the coast of Florida and found that 34% of individuals had tar in their mouths or esophagi, with over half with tar in their jaws. Hatchlings, juveniles, and adults can experience difficulty eating if their beak or esophagus is blocked, which could lead to starvation. Ingested oil can cause lead to decreased absorption efficiency, increased absorption of toxins, general intestinal blockage (i.e., local necrosis or ulceration), interference with fat metabolism, and buoyancy control problems caused by buildup of fermentation gases (Shigenaka et al. 2010). Buoyancy control is important because it allows sea turtles to surface or dive to depth freely; without it, they are especially vulnerable to predators, vessel strikes, and disruption of normal feeding behavior.

Chemical or toxicological effects of oil exposure on sea turtles has been assessed in very few studies and experiments and even less so during an oil spill. In one study (Lutcavage et al. 1995), loggerhead sea turtles, aged 15–18 months old, were exposed to south Louisiana crude oil over a 96-hour period. Physical effects of oil exposure included skin decay that required up to 21 days to recover. Oil was observed in the nares (nostrils), eyes, upper esophagus, and feces, indicating that even though the sea turtles were not fed during the exposure, ingestion was occurring. The internal effects of exposure included a decrease of nearly 50% in their hematocrits (red blood cell: total blood cell volume) and an increase in white blood cells to a level four times higher than in control animals. This response lasted throughout the recovery period before returning to normal blood chemistry (Lutcavage et al. 1995; Shigenaka et al. 2010).

Stacy et al. (2017) found that loggerhead and Kemp's ridley sea turtles exposed to oil during the *Deepwater Horizon* spill did not exhibit the same chemical effects as noted in the above laboratory study. Instead, oiled turtles had nonspecific blood abnormalities commonly associated with stress, dehydration, and exertion from oiling, capture, and transport (Stacy et al. 2017). Furthermore, no specific tissue toxicity, hemolytic anemia, or salt gland dysfunctions were observed based on blood analyses (Stacy 2012; Stacy et al. 2017).

Mitchelmore et al. (2017) estimated the mortality of oceanic sea turtles that were minimally to moderately oiled during the *Deepwater Horizon* spill by compiling estimates of oil ingestion by the sea turtles and comparing them to toxic endpoints following oil ingestion in turtles and vertebrate species. Oiled sea turtles were separated into oiling categories based on the extent of oiling, as assigned by the *Deepwater Horizon* sea turtle technical working group (STTWG). The five categories were non oiled (0), minimally oiled (1), lightly oiled (2), moderately oiled (3), and heavily oiled (4). The STTWG concluded that 100% of heavily oiled turtles died of physical effets of the oiling; therefore, that category was excluded from further comparison. Estimations of internal exposure and uptake of oil in sea turtles were used to calculate mortality in the known numbers of sea turtles documented by direct capture operations during the *Deepwater Horizon* spill. Mortality was calculated for each oiling category, resulting in an estimated overall mortality of 30% for all oceanic turtles within the footprint of the *Deepwate Horizon* spill, in addition to those already presumed dead from heavy oiling (Mitchelmore et al. 2017).

An oiled sand beach, or altered nesting substrate, can affect the incubation environment of the nest, changing natural temperature and moisture of the sand and the respiratory function of eggshells. Shifts in the temperature can alter natural sex ratio of sea turtles as embryos are temperature dependent (Hays et al. 2001; Phillott and Parmenter 2001).

Following the 1979 Ixtoc-1 blowout, Kemp's ridley eggs were used for a field experiment in which paired samples of eggs were incubated in non-oiled and oiled (aged) sands. Loggerhead eggs were used in a laboratory experiment where eggs were incubated with varying quantities of oil added on top of the sand throughout the incubation period (Fritts et al. 1982). The field experiments involving the Kemp's ridley eggs yielded no effects related to oil contamination. The loggerhead eggs that were oiled (during partially complete incubation) had increased embryonic mortality and differences in hatchling morphology. The results suggested that weathered oil is less toxic to turtle embryos than fresh oil. If an oiling event occurs near the end of a nesting season with hatchlings about to emerge, mortality could be high. However, if nests are built in older weathered oil, mortality and morphologic impacts may be reduced. It is unknown whether females would avoid an oiled beach or continue to nest even with the presence of oil.

According to Wallace et al. (2020), of all the thousands of spills worldwide in the last 60 years, only 22 spills reported impacts to sea turtles. The number of spills heavily favored toward North America (13 spills). Effects involving nesting females and/or hatchlings were reported more frequently than oceanic involvement of sea turtles, other than the occasional observational report of dead sea turtles in the waters near a spill. The potential for sea turtles to be impacted by oil spills is very high; however, these incidents are few in numbers either by chance occurrence or by limited reporting. By far, most studies of these impacts come from the *Deepwater Horizon* spill. The combination of being a recent large spill, occurring within U.S. waters, and completion of a NRDA case makes it one of the largest collection of resources for providing a summary of oil spill impacts to sea turtles.

Omissions or underreporting of sea turtle impacts limits our understanding of effects of spills on sea turtles during prior spills. Observations of nests and nesting females are reported in greater numbers, whereas ocean-going turtles are highly mobile and often difficult to observe, study, and monitor in remote areas (Bolten 2003; Plotkin et al. 1993; Wallace et al. 2020).

Only one of the sixty-two spills of 500–20,000 bbl of crude oil, condensate, or diesel included in this synthesis (**Table 14-1**) had documented impacts to sea turtles. Studies of the *Jessica* spill documented diesel and bunker fuel oil impacts (no studies of crude or condensate were identified), including details of the spill, life stage of impacted turtles, and details of the effects. No recovery information from this spill was identified.

Table 14-1. Studies with documented impacts to and recovery of sea turtles from diesel spills (500–20,000 bbl)

Oil Spill (Shoreline Oiling, km)	Degree of Oiling (oil type)	Oil Cleanup	Life Stage and Habitat Impacted	Documented Effect/Impacts	Recovery (years)
2001 *Jessica,* Galapagos Islands, Ecuador[a]	Very Light to Light (2,800 bbl diesel and 2,160 bbl bunker fuel oil)	No	Marine, sandy, rocky shoreline; adult sea turtles and marine iguanas.	8 adult green sea turtles and 24 marine iguanas were impacted by the spill.	Recovery was not assessed.

[a]Lougheed et al. (2002)

The *Jessica* spill discharged diesel and bunker fuel oil in the waters of the Galapagos Islands, Ecuador, impacting the shoreline and waters of seven islands. Sea turtles were impacted on all seven islands and included eight adult green sea turtles. Though researchers state that the impact on coastal vertebrates appeared negligible, they highlight the difficulty in accurately documenting ocean-going species that could be affected, die, and sink before they are found or reported. Furthermore, animals may not be immediately affected by exposure to oil and long-term survival may be compromised (Lougheed et al. 2002).

Reports for the fishing vessel *Jin Shiang Fa* that ran aground at Rose Atoll in 1993 suggested that the spill from the vessel may have affected sea turtles by exposing them to oil that persisted around the island for at least 6 weeks; however, there were no documented impacts to sea turtles. Adult sea turtles were observed at the wreck site, but the only potentially associated impacts to sea turtles comes from the report that two emaciated juvenile sea turtles were captured on the atoll approximately 1 year after the spill (USFWS 1997). Additional concerns were raised regarding potential entanglement hazards to sea turtles from debris and long lines that were still attached to the vessel. No further impacts were documented, and no recovery estimates were assessed.

During the MP-69 spills as a result of Hurricane Ivan in the Mississippi River delta, Louisiana, anecdotal reports from overflights indicate that two sea turtles were swimming through the surface oil. Limited shoreline cleanup occurred, and no further documentation of impacts to sea turtle was reported (RPI 2005).

Oil spill response is typically focused on cleaning up the oil quickly and efficiently. In haste, responses can do more harm to sea turtles than originally expected. Even with established best management practices, response activities that can affect nesting beaches include: using artificial lighting at night, increasing human presence, and using mechanical cleanup methods that physically remove contaminated sand and debris (Michel et al. 2015). Without precautions, these activities can destroy nests, disorient hatchlings, disturb or deter adult females from nesting, or create obstacles (i.e., boom) that prevent or hinder access to the beach or sea (Lauritsen et al. 2017).

Rehabilitation of oiled sea turtles is a response activity that can be considered if trained personnel are involved. In offshore spills, locating and capturing live, oiled animals are difficult without an intensive rescue effort. Furthermore, long-term success of rehabilitation is extremely difficult to determine. Stranded, oiled turtles that are captured and cleaned have a high probability of being released back into the wild; however, their long-term survivorship is difficult to determine (Stacy et al. 2017). Pilcher (2000) noted that several oiled adult turtles treated during the 1991 Gulf War spill nested later that year on the Gulf Islands. Studies of oiled sea turtles during the *Deepwater Horizon* spill detailed adverse physiological effects as a consequence of stress, exertion, physical exhaustion, and dehydration coincident with oiling, capture, and transport (Stacy et al. 2017). All factors of rehabilitation should be considered prior to embarking on this response strategy.

Based on the available literature, impacts to sea turtles from oil exposure can be described in four phases: 1) an initial impact phase, where sea turtles experiences a measurable reduction in abundance caused by direct mortality from the oil's toxicity, fouling effects, and disturbances from treatment methods, as well as indirect effects from habitat degradation; 2) a secondary impact phase, where sea turtles return to oiled habitats unknowingly or re-oiling events have covered nesting beaches, causing additional toxicity to the sea turtles and their food source; 3) short-term decrease in populations caused by direct mortality or lack of reproduction; and 4) long-term recovery of the species population, which in some cases could take decades for the species to return to pre-spill population numbers or the return of normal diversity and density indicating the start of full recovery (NOAA 2016a). Slow recovery is possible for sea turtles as they have long lifespans, overlapping generations, and wide distributions (NOAA et al. 2016). Full recovery is achieved when the sea turtle population reaches diversity, density, and age structures comparable with unoiled reference sites (though studies seldom are conducted long enough or rigorously enough to document such full recovery).

14.4 Summary and Information Needs for Assessing Impacts to Sea Turtles

There was one median-range spill that documented impacts to sea turtles. Even considering larger spills not included in this analysis, there are very few oil spills that provide more than opportunistic observations of injured or dead sea turtles caused by a spill (Michel 2021). Physical fouling by oil is the most frequently reported effect. Heavily oiled sea turtles can become weighed down, reducing their ability to surface to breathe, dive for feeding, avoid predators or vessel strikes, and regulate body temperature. Feeding on tar balls that are confused with prey items can lead to blockage of their mouths or esophageal pathways. Oil ingestion can lead to decreased absorption efficiency, absorption of toxins, general intestinal blackage, interference with fat metabolism, and buoyancy control problems caused by buildup of fermentation gases. Many of these effects of ingestion are inferred; however, studies of oiled sea turtles during the *Deepwater Horizon* spill showed nonspecific blood abnormalities commonly associated with stress, dehydration, and exertion from oiling, capture, and transport.

Oil stranded on beaches can result in reduced nesting success due to direct oiling of eggs and indirect effects to temperature, moisture, and oxygen exchange, though these effects have not been documented during actual spills. Studies following the Ixtoc-1 spill and laboratory studies suggest that weathered oil is less toxic than fresh oil.

Physical impacts to sea turtle nests during a response are a serious concern during nesting season. In the U.S., best management practices to protect nests and hatchlings are enforced, and compliance is monitored to prevent such impacts.

No studies have documented population-level impacts to sea turtles following an oil spill, mostly because their population is poorly understood, and data on where juveniles go and how they survive, their extensive geographic range, and other stressors that reduce their populations, are limited.

To better quantify the impact and recovery of oil exposure to sea turtles, it is recommended that rigorous field study designs be prepared, ready for implementation in the event of a spill. Utilizing existing protocols, such as those available from the Sea Turtle Stranding and Salvage Network, will facilitate documentation of injured, harassed, or dead sea turtles during a spill. Furthermore, if live sea turtles are recovered, documented rehabilitation will facilitate future response personnel on how to handle such a situation. Effective response to oil spills requires early identification of species and life stages at risk, timely deployment of knowledgeable personnel and other assets, efficient preparation of emergency responders, and judicious collection of information that will only be available during and shortly following a spill. Tools available to understand the magnitude of potential injuries to sea turtles as a result of an oil spill and inform recovery include: surveys of nesting beaches and marine habitats (e.g., vessel-based and aerial) to document sea turtle presence, abundance, and oil exposure; evaluation of oiled turtles and nests encountered during rescue efforts, stranding response, and other activities to document exposure and effects (including those caused by response activities); and for larger spills, integration of field observations, remote sensing data, and other information over time to evaluate the magnitude and persistence of injuries to sea turtles and their habitats.

More information can only help to provide safe, reliable methods on handling and rehabilitating sea turtles. Wallace et al. (2020) provide general recommendations to resource managers, wildlife researchers, and other stakeholders on how to properly collect and document relevant data on oiled sea turtles, provide guidance to response personnel, and conduct studies or data analyses that can further our understanding of oil exposure to sea turtles and how best to protect sea turtles in an oiled environment.

15 Marine Mammals

15.1 Resource Description and Habitats

This chapter addresses impacts of spills of 500–20,000 bbl of crude, condensate, or diesel on marine mammals–on land, on or in sea ice, or at sea. Marine mammals are found in marine ecosystems around the world and include a broad range of species uniquely adapted to live in marine environments throughout the world. Physical adaptations allow them to live in extreme marine environments that span temperatures, depths, pressures, levels of darkness, and availability of food sources. Marine mammals feed at trophic levels ranging from zooplankton and macroalgae to other marine mammals (Bossart 2011; Moore 2008). Though primarily inhabitants of the marine environment, specific requirements of each group, and ultimately each individual species, require the utilization of many other habitats including beaches, rocky shores, tidal flats, sea ice, and mid-depth and deep water (Sullivan et al. 2019). Marine mammals are classified into four taxonomic groups: cetaceans, pinnipeds, sirenians, and marine fissipeds.

Cetaceans, including whales, dolphins, and porpoises, represent over 70 different species and are completely aquatic: feeding, mating, calving, and raising young in the water. They are an extremely diverse group and inhabit all of the world's oceans. Cetaceans occur in waters ranging from shallow estuarine and nearshore to deep offshore waters, including whale species that can dive to depths of 3,000 m. Cetaceans inhabit U.S. waters from the Arctic, Pacific, and Atlantic oceans to the Gulf of Mexico and there are approximately 70 species in U.S. waters, compared with 21 species in the Gulf of Mexico (NOAA 2016a). Cetaceans breathe through blowholes (their nostrils) and must surface periodically for air, though some species can stay submerged for 2 hours or more. Cetaceans are further divided into two subgroups of baleen whales (mysticetes) and toothed whales (odontocetes). Mysticetes are filter feeders that forage for zooplankton and small fish by skimming or gulping large amounts of prey and water. As the water is forced back out, baleen plates trap the prey inside the whale's mouth. Odontocetes have various numbers of conical or spade-shaped teeth that are used to capture prey, which is often found using echolocation.

Pinnipeds, including phocids (earless, or true, seals), otariids (eared seals, generally grouped as sea lions or fur seals), and odobenids (walruses), are fin footed and move on land, sea ice, and water. Most of their lives are spent swimming and eating in water but they come on land and sea ice to bear young, rest, and molt. There are 34 species of pinnipeds; 14 species inhabit North American waters and are distributed from Mexico in the Pacific Ocean, up the west coast to and across the Arctic Ocean, and then south to New England in the Atlantic Ocean. The 2008 declaration of the extinction of the Caribbean monk seal[2] confirmed there are no pinnipeds living in the Gulf of Mexico. Pinniped habitats range from beaches, rocky shores, fast and pack ice, to nearshore and oceanic offshore waters. The life histories of pinnipeds vary widely among species with all spending time on land, sea ice, and at sea. During breeding and rearing of pups, some species will spend most of their time feeding, nurturing pups on land, while others, such as the harbor seal and walrus, enter water almost immediately following birth (Bowen 2017). The timing and duration of haul-out behavior varies based on social interaction, mating, giving birth, and caring for young. For species such as the walrus, birth, mating, and molting occur in various segments of the population for more than half the year (Geraci and St Aubin 1990); others spend time on land

[2] The last confirmed sighting of a Caribbean monk seal occurred in 1952

throughout the year. Most species have large breeding colonies with high site fidelity (Bowen 2017; Geraci et al. 1988).

Sirenians (manatees and dugongs) spend their entire lives in tropical water and are primarily herbivorous, though they incidentally feed on invertebrates (Powell 1978). Two subspecies of West Indian manatee, the Antillean manatee (*Trichechus manatus manatus*) and the Florida manatee (*Trichechus manatus latirostris*), inhabit U.S. waters. At one time in danger of extinction, the manatee is protected by both the ESA and Marine Mammal Protection Act (MMPA). Their population in Florida has grown in the past 25 years from an estimated 1,200 manatees to about 6,300 in 2020 (USFWS 2020b). Their distribution is typically limited to inshore, low-energy habitats that support the growth of seagrasses. They feed on a suite of aquatic and semi-aquatic vegetation and, once underwater food sources are gone, they will feed on floating vegetation that includes algae, roots, or detritus (Alves-Stanley et al. 2010). Seagrasses are often encrusted in other organisms such as diatoms, mollusks, and crustaceans that also inadvertently make up part of the manatee diet. In some situations, manatees will go into very shallow waters to reach vegetation that is on the shoreline or exposed during low tide. Generally, manatees overwinter in Florida, but during warmer months they can be found as far west as Texas and as far north as Rhode Island.

Marine fissipeds are paw- or pad-footed mammals that include polar bears and sea otters. Polar bears depend on the ocean for a majority of their food and utilize both land and sea ice for bearing young, grooming, resting, or feeding. Polar bears are endemic to Arctic waters and adapted to a semi-marine life. Their ability to traverse over ice or through water for great distances makes them a unique Arctic animal that can thrive in such an environment. Sea otters are the smallest marine mammals, seldom come ashore, and live and feed in nearshore waters (generally within 1–2 km of the shoreline), typically near kelp beds (Geraci and St Aubin 1990). They feed primarily on shellfish and marine invertebrates and are found off the Pacific Coast of Alaska, Canada, Washington, and California. Sea otters have dense fur that provides thermal insulation, protecting them from the cold waters that they inhabit. They do not have the same subcutaneous fat (blubber) layer that other marine mammals have; thus, their energetic requirements are high, and they consume an amount of food equivalent to 20–25% of their body mass per day (Geraci and St Aubin 1990).

In the U.S., NMFS manages cetaceans and most pinnipeds including phocids (seals) and otariids (sea lions and fur seals). The USFWS manages odobenid pinnipeds (walruses), polar bears, otters, manatees, and dugong. Marine mammals within waters of the U.S. are protected under the MMPA, which prohibits humans from harassing, harming, or disturbing marine mammals. Some marine mammal species, such as the manatee, beluga whale, blue whale, bowhead whale, fin whale, North Atlantic right whale, North Pacific right whale, and sperm whale, are also protected under the ESA.

The wide range of traits and habitats utilized for each species makes marine mammals vulnerable to a suite of threats including, but not limited to, oil spills (**Figure 15-1**). Threats to marine mammals are largely caused by human impacts such as accidental capture in fishing gear, habitat destruction, poaching, pollution, harassment, and vessel strikes (NOAA 2016a). Natural stressors such as climate change, prey availability, harmful algal blooms, and disease can all have a short- or long-term effects on marine mammal populations.

15.2 Pathways of Oil Exposure to Marine Mammals

Either on land, sea ice, or at sea, marine mammals can be exposed to oil through dermal contact (e.g., skin, mucous membranes), grooming (e.g., preening of furred marine mammals), inhalation, water and prey ingestion, and/or aspiration (Sullivan et al. 2019; Godard-Codding and Collier 2018).

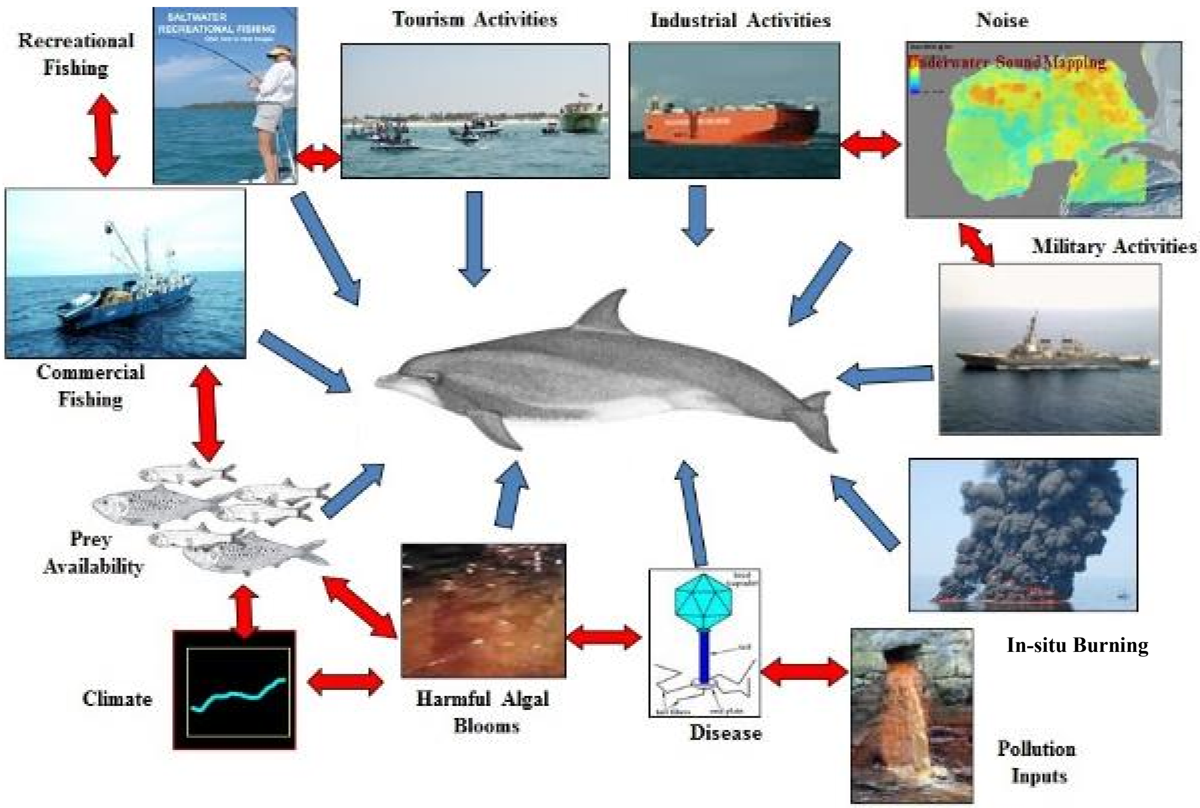

Figure 15-1. Factors affecting marine mammal population health and resiliency in the Gulf of Mexico (NOAA 2016a)

Due to the rapid metabolism of PAHs by vertebrates, determining exposure of marine mammals can be difficult. Matrices for determining exposure include external swabs of skin/fur, blowhole, and stomach contents (PAHs), bile (metabolites), urine (metabolites), and feces (PAHs and metabolites) (Sullivan et al. 2019). Lung tissue of a stranded dolphin suggested exposure to volatile petroleum components during the *Deepwater Horizon* spill (NOAA 2016a; Sullivan et al. 2019).

Early laboratory research and some observations during spills suggested that some marine mammals, especially dolphins, could detect and would avoid oil on the water. However, during the *Deepwater Horizon* spill, dolphins were observed in and around oily waters, clearly not avoiding oil (NOAA 2016a). Photographic evidence of marine mammals in and around visible oiling is used to provide qualitative evidence of exposure.

With no reports of oiled manatees from large oil spills, and no experimental studies on the subject, direct exposure can be assumed based on the nature of these animals to move into shallow, nearshore waters for foraging, reproduction, and refuge. Once in shallow waters, the manatee could easily be oiled as it surfaces in intervals at least once every 20–30 minutes, often more frequently. The feeding behaviors would suggest that ingestion is likely by either incidentally ingesting tar balls or feeding on contaminated vegetation (Geraci and St Aubin 1980). Three manatee carcasses recovered in Florida in 1981 and 1982 contained tarlike material in the lower digestive tract along with plastic in one of its stomachs (St Aubin and Lounsbury 1990). There is a small chance that a vegetation die-off and subsequent shortage of food could impact manatees; however, this type of impact has been determined to be too brief and localized to be a threat (Beck et al. 1978). Refer to Chapter 11 for potential impacts of oil spills to seagrass habitats, where recovery for an actual spill was documented as <1 year.

Pinnipeds that come ashore for foraging, resting, breeding, or avoiding predators could be exposed to oil stranded on the shoreline. Oiling events that result in oil accumulation on the seafloor can create situations in which periodic re-exposures or continued exposure can occur. Species that dig while foraging can disturb sediments enough that they could be exposed (Garshelis and Johnson 2013). Many pinnipeds are piscivorous, but they also feed on cephalopods, planktonic crustaceans, and epibenthic organisms. Some species are primarily benthivores (e.g., walrus and bearded seal).

Sea otters, fur seals, polar bears, or other furred marine mammals have a higher potential for impacts from direct oiling because oil adheres to the fur more so than bare skin, reducing their ability to maintain warmth and buoyancy (Geraci 1990). Fur traps a thin layer of air next to the animals' skin, which prevents skin contact with cold ocean waters. When the fur is oiled, the insulative layer becomes compromised, and the threat of hypothermia increases (Geraci 1990; Loughlin et al. 1996).

15.3 Impacts of Oil Exposure and Response to Marine Mammals

Field studies on marine mammals following oil spills are limited due to difficulty in tracking live injured or oiled animals. Additionally, there are very few laboratory studies on marine mammals due to ethical constraints; however, the laboratory studies conducted in the past are discussed as appropriate to inform a better understanding of potential impacts to marine mammals from oiling.

Cetaceans that are exposed to oil via direct contact, inhalation, ingestion, and/or aspiration of oil can experience severe damage to internal organs and disruption of reproductive processes, resulting in long-term population impacts (Engelhardt 1983; Geraci and St Aubin 1990). Laboratory studies on cetaceans have shown multiple effects from oil exposure, including liver damage in captive bottlenose dolphins that had crude oil added to their tank; skin lesions in a number of captive delphinid species from oil applied to their skin; and skin lesions after oil was applied to the skin of a live, stranded sperm whale (Geraci 1990). Studies focused on the health or survival of cetaceans following median-range oil spills are very rare; thus, most studies on the effects of oil on cetaceans come from large oil spills such as the *Exxon Valdez* and *Deepwater Horizon* spills.

Pinnipeds directly oiled through ingestion, inhalation, or coated with oil on their fur have shown many short- and long-term effects. Not only is thermoregulation affected, but physiological effects such as lung inflammation, increased respiratory rates, respiratory failure, abnormal nervous system functions, liver and kidney damage, and reproductive impairment can occur, as well as death (Engelhardt 1983). Three laboratory studies on two species of phocid seals examined the effects of crude oil ingestion (St Aubin 1990). Ringed seals (*Phoca hispida*) showed rapid absorption and clearing of hydrocarbons from body tissues and fluids when exposed to crude oil during experiments that included immersion and ingestion (Engelhardt et al. 1977). The researchers concluded that, though small quantities of oil could be tolerated, chronic ingestion of subtoxic quantities could have subtle effects, including increased hydrocarbon levels in tissue and blood, and plasma cortisol levels. Under natural conditions at sea, limited surface fouling with fresh oil would not appear to cause stress. However, pinnipeds trapped near the source of a spill or forced to inhabit an area of heavy oil accumulation (such as in ice) could exhibit severe negative effects: to the eyes, with conjunctivitis, swollen nictitating membranes, and corneal abrasions and ulcers; to the fur from fouling that can lead to hyperthermia and drowning; to the lungs from inhalation by animals that are already stressed by parasitism or other pre-existing metabolic disorders (St Aubin 1990).

No field or laboratory studies were identified that documented impacts to manatees during any actual median-range spill. Manatee exposure to oil would most likely irritate the eyes and sensitive mucous membranes with not much harm to the thick epidermis (St Aubin and Lounsbury 1990). Response workers responded to manatees in contaminated waters during the *Deepwater Horizon* spill; however, there was not enough information collected on exposure and injury to quantify the injury or estimate recovery (Wilkin et al. 2017). During the Iran-Iraq War in 1983, an anonymous communication noted that 53 dugong carcasses were recovered approximately 5 months after the spill in the Arabian Gulf (St Aubin and Lounsbury 1990). Researchers believe that this represented a major proportion of the local stock; however, no further analyses were conducted to confirm this claim.

Marine fissipeds exposed to oil can experience short- and long-term effects. Oil coating of fur decreases the ability to regulate body temperature and results in ingestion and inhalation of oil from preening. Oiled fur is matted, which can allow water to penetrate to the skin and the animal can cool rapidly and/or suffer secondary exposure and absorption of toxic compounds. Oil ingestion can greatly disrupt an individuals' organs and reproductive processes, with potential long-term population impacts. In polar bears, laboratory studies have shown the bears do not actively avoid oil and will eat oil-contaminated food (Hurst et al. 1991). Derocher and Stirling (1991) documented loss of hair by a polar bear heavily oiled with a type of lube oil at least three months after initial observation. Four years later, the same bear was recaptured and no damage was observed. Øritsland et al. (1981) experimentally oiled three polar bears with crude oil; two died one month after exposure. The oil was applied to a pool as a 1 cm-thick slick 36 hours before the bears were forced into the pool for 15–50 minutes. Fouling of the fur led to thermoregulatory and metabolic stresses. Oil ingestion during grooming caused behavioral abnormalities, including anorexia, acute anemia, and tissue damage. Renal changes were the most serious measured and were the cause of death of the two bears. Many of the systemic toxicity effects were manifested weeks after exposure.

In experimental studies, oiled sea otters would groom themselves for a long time and at higher frequency than unoiled sea otters. Coated sea otters will experience an almost self-destructive cycle that starts with oil fouling their fur, which reduces its insulative properties and increases heat loss; the otter then compensates by increasing its metabolic rate, which makes it consume more food, until eating gives way to grooming and all the energy is now spent ingesting the oil, causing internal damage, and ultimately never successfully regulating its body temperature (Siniff et al. 1982). Consumption of oiled prey, for example, bivalves, can lead to chronic exposure and internal damage in sea otters (Geraci and St Aubin 1990). Indirect exposure to oil includes inhaling the volatile vapors that injure lungs and other internal organs (Siniff et al. 1982).

From a thorough review of the literature from median-range oil spills, nine oil spills were identified as having data on impacts to marine mammals from field-based studies or reports of spills of 500–20,000 bbl of crude oil, condensate, or diesel (**Table 15-1**). However, only one spill reported a recovery timeframe. Even though the 1997 Torch platform spill of 163 bbl of crude oil was below the threshold of 500 bbl, it is included because there are so few spills for which there are any data on impacts to marine mammals.

Based on the available literature, impacts to marine mammals can be described in three phases: 1) an impact phase, where the marine mammal community experiences a measurable reduction in abundance caused by direct mortality from the oil's toxicity, fouling effects, disturbances from response methods, as well as indirect effects from habitat degradation; 2) a secondary impact phase, where marine mammals return to oiled habitats, potentially causing reduced reproduction and additional toxicity to the animal, its food source, and other species; and 3) long-term recovery of the species population, which in some cases could take decades. Full recovery is achieved when the marine mammal community reaches diversity, density, and age structures comparable with unoiled reference sites (though studies seldom are conducted long enough or rigorously enough to document such full recovery).

Table 15-1. Studies with documented or estimated impacts to and recovery of marine mammal populations from crude oil, condensate, or diesel spills (500–20,000 bbl)

Oil Spill	Oil Type and Volume	Marine Mammal Group	Documented Effect/Impacts	Recovery (years)
2001 T/V *Jessica,* Galapagos Islands, Ecuador[a]	2,800 bbl diesel and 2,160 bbl bunker fuel oil	Pinniped: sea lions	79 living oiled and one dead (juvenile) sea lion were recorded. Oil exposure caused conjunctivitis and burns; however, most were able to be cleaned. Population declines were observed for the first few months of the spill, but population numbers returned to pre-spill levels within 1 year.	1
1990 T/V *American Trader*, CA[b]	9,919 bbl Alaska North Slope crude	Pinnipeds: sea lions and seals; Cetacean: dolphins	21 dead sea lions, 3 dead harbor seals, and 2 Pacific white-sided dolphins were recovered during the response. Necropsies indicated that none died from causes related to the oil spill.	Recovery was not assessed
1987 T/V *Antonio Gramsci*, Baltic Sea[c]	4,200 bbl crude	Pinniped: seal	A single seal was reported as oiled, but a necropsy was not performed to confirm that oil exposure was the cause of death.	Recovery was not assessed
1992 UNOCAL, Avila Beach, CA[d]	600 bbl San Joaquin Valley crude	Fissiped: sea otters	Three dead and two living sea otters were oiled; Two of three dead sea otters died due to effects of acute oiling; One of the living sea otters was cleaned and released. Additional sea otters were observed swimming in the oil; however, the actual number of otters exposed to the spill is unknown.	Recovery was not assessed
1989 T/V *Bahia Pariaso*, Antarctica[e]	3,760 bbl diesel fuel arctic	Pinniped: seals	Several oiled seals were spotted, but no further documentation is provided.	Recovery was not assessed
2015 Refugio Beach, CA[f]	500 bbl Monterey crude	Pinnipeds: seals, sea lions; Cetaceans: dolphins, whales	Once adjusted for non-oil spill related strandings, the total strandings during the spill period was 104 pinnipeds and 19 cetaceans.	Recovery was not assessed
1979 M/V *Ryuyo Maro*, Bering Sea, AK[g]	6,190 bbl No. 2 fuel oil	Pinniped: seals	Dead seals were observed but no further documentation was recorded. Five seals were found trapped in fishing nets from the vessel with four of them dead and one alive, which was released.	Recovery was not assessed
1997 Torch platform, CA[h]	163 bbl crude	Pinniped: sea lion	One dead California sea lion was found oiled during the response.	Recovery was not assessed
1984 Uniacke G-72 blowout, Nova Scotia[i]	1,500 bbl gas condensate	Pinniped: seal; Cetacean: whale	Aerial and shipboard surveys for wildlife during the blowout observed one grey seal and one pilot whale within the condensate slick. Oiled seals were seen on nearby Sable Island, but no shoreline oiling or dead animals were reported.	Recovery was not assessed

[a]Lougheed et al. (2002); [b]Card and Meehan (1991); [c]Hirvi (1990), Geraci and St Aubin (1990); [d]CDFG and USFWS (1999); [e]Kennicutt et al. (1991a), Geraci and St Aubin (1990); [f]Refugio Beach Oil Spill Trustees (2020); [g]Reiter (1981); [h]Torch/Platform Irene Trustee Council (2007); [i]Carter et al. (1985)

It is important to note that the specific definition of recovery varies by spill and by the species or taxonomic group and the metrics studied. In this report, we summarize and synthesize the limited information available, and we acknowledge that recovery in this context may or may not equate to full species population recovery. Spills within the volume range examined here did not typically include estimates of recovery; they mostly documented impacted (dead or oiled) animals. However, necropsies to determine cause of death or fingerprinting of oil to verify source oil were not conducted on all spills. Where necropsies were conducted, several oiled animals were found to have a primary cause of death other than oil exposure. This further provides information that, without proper documentation and analysis, the link between marine mammals and oil exposure is unclear and cannot be confirmed.

Surveys of eight islands following the *Jessica* spill in the Galápagos Islands reported 79 oiled sea lions (including a dead juvenile) around four of the islands. Forty-two of the sea lions, all pups and juveniles, were cleaned with mild soap, mayonnaise, and water, and some were hydrated with a subcutaneous physiological solution of vitamin B (Lougheed et al. 2002). During monitoring, as many as 50% of pups and juveniles in affected colonies exhibited pathological effects such as conjunctivitis and burns.

Two of the three sites with oiled animals observed declines in population for the first few months of the spill. The short decline in population was unusual based on pre-spill data collected since the severe decline associated with a 1997/1998 El Nino event. Population numbers returned to pre-spill levels within 1 year (Lougheed et al. 2002).

For the other eight spills in **Table 15-1,** there was only documented oiling of marine mammals. Additional information is provided below, where available.

The 1992 UNOCAL spill in Avila Beach, California resulted in 600 bbl of crude oil that directly contacted kelp beds and offshore rocks in areas actively utilized by sea otters. Two of three dead sea otters examined during the spill died due to effects of acute oiling. Two live oiled sea otters were captured. One was released due to unrelated injuries making rehabilitation too risky, and the other was transported to Monterey Bay Aquarium for cleaning and rehabilitation. Ultimately, the live, cleaned sea otter was released and survived at least 8 weeks as determined by radio tracking (CDFG and USFWS 1999). Additional sea otters were observed swimming in the oil; however, the actual number of otters exposed to the spill was unknown.

Pinnipeds (seals and sea lions) and cetaceans (whales and dolphins) occur in waters affected by the 2015 Refugio Beach spill in California. Reports from the incident response and the California Marine Mammal Stranding Network documented 264 marine mammals stranded during the first 3 months of the spill. Many of the strandings were not related to the oil spill and more likely attributed to pre-spill injury or fishing-related entanglement. The NRDA Trustees relied on the mammal stranding reports during their assessment of marine mammal impacts due to the difficulty in studying the wide-ranging species (Refugio Beach Oil Spill Trustees 2020). Once adjusted for non-oil spill related strandings, the spill-related strandings were 104 pinnipeds and 19 cetaceans (Refugio Beach Oil Spill Trustees 2020). Further analysis of impacts to marine mammals, including recovery estimates, was not conducted.

In 1979, the *Ryuyo Maro* spill discharged over 6,000 bbl of No. 2 fuel oil in waters adjacent to known seal rookeries with 10,000 and 20,000 seals present. Seals were observed swimming in small slicks of oil and mousse offshore without being noticeably affected (Reiter 1981). Dead seals were observed but no further documentation was recorded regarding oiled animals. Five seals were found trapped in fishing nets from the vessel (the one alive otter was released). There was evidence that seals were killed during the vessel demolition (Reiter 1981).

Spills often require response operations with vessels and personnel on land or water assessing impacts, actively cleaning oil, removing oily debris, or conducting shoreline assessments. Response activities can disturb marine mammals by increasing vessel traffic and aircraft traffic (including manned and unmanned drones), physically excluding animals from areas with boom or other response equipment, and entrapment of animals in nearshore oiled zones during skimming, in-situ burning, or other response operations. These operations can increase stress to marine mammal populations by increasing respiratory and dermal exposure, disrupting normal reproductive behaviors or maternal care, or disturbing shallow feeding and resting habitat (Sullivan et al. 2019). Slow-moving marine mammals that inhabit shallow inshore waters, such as manatees, may be at the highest risk of strikes by response vessels. Of 2,940 necropsies performed by the Florida Fish and Wildlife Conservation Commission Marine Mammal Pathobiology Laboratory between 1993 and 2003, 713 fatalities (24%) were the direct result of vessel strikes. Though not related to an oil spill, these data can help to facilitate safe boating operations within the response community to protect manatees and other marine mammals.

15.4 Summary and Information Needs for Assessing Impacts to Marine Mammals

It is evident from this literature review that there have been few assessments of the impacts of median-range oil spills on marine mammals, and laboratory exposures have not been conducted for several decades for ethical reasons. Spills examined here do not provide comprehensive data on oil exposure of marine mammals or provide further guidance or understanding that can be used to estimate population recovery. Documentation of oiled animals can help in the development of restoration plans or NRDAs. However, without linking an oiled animal to the source oil through necropsy or oil fingerprinting, the connection between oil exposure and impact can be hypothesized, but not confirmed.

Larger-volume oil spills with extensive field studies should be looked to for population recovery estimates and guidance on oil exposure to marine mammals. Field studies following the *Exxon Valdez* and *Deepwater Horizon* spills (synthesized in Michel, 2021) have provided the majority of the current understanding on this topic. However, the advances occurring since 2010 have been only for cetaceans.

Until 2010, common assumptions about how oil spills in the median range affected marine mammals were: 1) cetaceans would mostly avoid spilled oil and experience, at most, transient impacts; 2) furred species would be at high risk of mortality from fouling and ingestion during preening, particularly sea otters, because of their close association with nearshore habitats, consumption of bivalves that can accumulate oil, and potential for long-term exposures to oil being released from shorelines; 3) pinnipeds at oiled haulouts would be at medium risk of fouling and inhalation impacts; and 4) consumption of oiled prey would not have measurable impacts to cetaceans and pinnipeds. One new paradigm can now replace the first assumption listed above. Based on studies following the *Deepwater Horizon* spill, we now know that cetaceans can experience substantial and long-lasting impacts, to include reduced reproduction and increased disease and mortality. Moreover, recovery of impacted populations may take decades. These impacts, not surprisingly, were documented in cetaceans that were resident in semi-enclosed waterbodies that were heavily oiled. These impacts were documented for inshore populations since this is where the studies were feasible to conduct. However, the impacts were estimated to have occurred over a wide range of species and geography.

The *Deepwater Horizon* spill provided an opportunity for researchers to develop a model that was parameterized for six populations of dolphins and compared population trajectories between oil-exposed populations and a no-oil exposure scenario. Time to recovery was determined by using a weight-of-evidence approach, and information was gathered from both stranded animals and health assessments conducted on live animals. Godard-Codding and Collier (2018) recommended that the model be refined as new observations of the affected populations are made, and that new models be designed for other species and other regions, ready to use in future spill situations.

To better quantify the impact and recovery of oil on marine mammals, rigorous field study designs could be prepared, ready for implementation in the event of an oil spill that is likely to affect marine mammals. Utilizing existing marine mammal stranding network protocols will facilitate additional information on how animals are injured or killed during a spill. Furthermore, documentation of rehabilitation of live animals will help future resource personnel on ways to help oiled animals. The response for the *Deepwater Horizon* spill was expansive and required many trained personnel to cover a large area. However, researchers still recommend that additional state, federal, and private personnel to be trained in responding to marine mammal stranding events (Wilkin et al. 2017). By training personnel in marine mammal stranding and oil spills, spill events can be used to further advance marine mammal research and better protect marine mammal populations from future spills.

16 Terrestrial Habitats and Wildlife

16.1 Habitat Description, Communities, and Ecological Functions and Services

This chapter includes terrestrial habitats and wildlife associated with, or geographically adjacent to, coastal and estuarine habitats that could be affected by median-range marine spills of 500–20,000 bbl of crude oil, condensate, or diesel and subsequent response impacts. Terrestrial habitats are defined by processes that affect their physical features including the availability of air and water, species diversity and composition, soil profiles, and topographic change. These differences lend themselves to a diverse assemblage of flora and fauna, much different from the adjacent marine habitats. Terrestrial habitats are not influenced by the transport of materials and organisms by way of ocean waves and currents, but rather by terrestrial pathways and climatologic patterns and events. These habitats also have the potential to be heavily influenced by human development. Terrestrial habitats and wildlife discussed in this chapter are ecologically connected to marine habitats providing ecosystem services including nesting, feeding, and roosting habitat for birds, mammals, amphibians, and reptiles; habitat for organisms that are critical to the food web; coastal protection; and aesthetic and recreation value to humans (NOAA 2016a).

Terrestrial habitats reviewed for this synthesis include dunes, coastal forests, coastal prairie, barrier islands, and arctic and sub-arctic tundra. These habitats are home to a wide array of coastal terrestrial wildlife. Terrestrial habitats provide shelter, food, and protection for terrestrial and semi-aquatic mammals (e.g., deer, bear, rodents, raccoons, river otter, etc.), terrestrial and marsh or beach dwelling birds (e.g., seaside sparrow), amphibians, and terrestrial and semi-aquatic reptiles (e.g., gopher tortoise, alligators, turtles). There are many terrestrial species that utilize beach or salt marsh habitats at some point of their lives for feeding, protection, or nesting. Examples include beach mice, seaside sparrow, and marsh rice rat. Some species using coastal terrestrial habitats are protected under the ESA or other federal or state regulations. For example, small mammals such as federally protected beach mice live solely in dunes and coastal scrub adjacent to beaches. Dunes adjacent to beaches in the northern Gulf of Mexico are used by 70% of the wintering population of the federally threatened piping plover (USFWS 2020a; NOAA 2009; NOAA 2016a).

16.2 Oil Behavior and Persistence on Terrestrial Habitats and Wildlife

All studies identified were for crude oil or diesel-like oil spills. Marine oil spills are carried to shore by winds and currents or sea ice and typically strand in the upper intertidal zone or supratidal zone of shorelines as oily swashes or strand lines, tar balls, patties, or oil mats (NOAA 2016a). For oil to strand within terrestrial habitats, the spill must originate within the upland environment via a pipeline, truck, rail, or storage facility accident; be transported to terrestrial habitats by response vehicles, equipment, or personnel crossing upland areas; or be transported by environmental factors such as strong storm events or through aeolian forces (i.e., oiled beach sand blown into upland habitats). Oiled wildlife can also transport smaller amounts of oil to terrestrial habitats.

Oil in terrestrial habitats can remain for days, months, or years, depending on the type and amount of oiling, geographic location, climatological factors (e.g., temperature, wind, precipitation), and response efforts. Heavy oiling in terrestrial habitats can penetrate soils, slowing rates of removal and weathering. Spills in arctic and subarctic regions may or may not penetrate the substrate depending on the season. For example, oil penetration would not be expected during frozen conditions in winter. However, colder temperatures may also slow the rate of oil degradation. Another unique situation that occurs in arctic regions is the interaction with permafrost, a perennially frozen layer of earth that may seasonally thaw but remains frozen at various depths throughout the year. Disrupting the permafrost layer during an oil spill or oil spill response may lead to increased long-term damages to the habitat (Linkins et al. 1984).

16.3 Impacts of Oil Exposure and Response on Terrestrial Habitats and Wildlife

There are very few studies that have documented, let alone quantified, the impacts and recovery of terrestrial habitats and wildlife from marine oil spills in the volume range examined here. Impacts to terrestrial habitats are often indirect and/or secondary impacts resulting from response and not direct oiling. Wildlife can be exposed to secondary impacts of an oil spill even if living inland of the shoreline. Degradation of habitat, alterations in the food web structure, and foraging within or near contaminated resources can all expose wildlife to an oiled environment (Bergeon Burns et al. 2014). Additionally, many species of terrestrial wildlife are generally mobile enough to avoid oiled shorelines.

Oil spills within terrestrial habitats have shown negative effects on terrestrial vegetation with recovery depending on the type of oil, amount spilled, and characteristics of the habitats and species involved. Experimental oil spills resulted in high levels of mortality in annual vegetation and minor to moderate levels of mortality to perennial species (Kinako 1981). A study of an oiled, tropical, grass-herbaceous community had a loss of almost 50% of vegetation in affected habitats. Over a period of 6 months, productivity was reduced by as much as 74% at affected sites vs. unaffected sites of herbaceous and perennial vegetation (Kinako 1981).

In Prudhoe Bay, Alaska, experimental oiling of wet and dry terrestrial habitats was conducted with crude oil and diesel to determine how tundra would recover from a pipeline spill. After 1 year, species of sedge and willow showed substantial recovery from crude oil spills while mosses, lichens, and most dicotyledons showed little to no recovery (Walker et al. 1978). Plots that were very wet or within standing water showed vegetation recovery within 1 year, while drier sites showed little to no recovery at all. Identical experiments using diesel resulted in complete plant mortality across all species except an aquatic moss. Lack of oil in the root zone was enough for the moss to survive under wet conditions, while in dry conditions the oil most likely killed the moss. Variation in recovery rates and remaining species diversity could be attributed to microhabitats within the test area that were dry, moderately wet, or very wet (Walker et al. 1978). Though not an example of recovery from a particular oil spill, this study showed that plant growth form, moisture availability, and differing species sensitivities are key attributes in determining vegetative recovery in the arctic (Walker et al. 1978).

Brendel (1985) compared the effectiveness of five revegetation methods for four years after a spill of crude oil in January 1981 that affected tundra along the Trans Alaska Pipeline. In soils where the TPH concentration was <100,000 ppm, seed and fertilize was effective; where TPH was >100,000 ppm, a combination of tilling and heavy fertilization was most effective. Behr-Andres et al. (2001) reviewed forty-nine spills of crude oil or diesel (mostly <12 bbl) in Alaska from 1995-2001 and concluded that diesel concentrations of 1,000–2,000 ppm were toxic, and crude oil concentrations <13,000 ppm were not toxic, whereas crude oil concentrations >100,000 ppm were toxic. They found that minimal disturbance and natural recolonization was generally recommended when possible in wet and moist tundra habitats, whereas in dry tundra habitats recovered faster with some treatment to encourage recolonization. Recovery rates varied from 1–12 years.

Jorgenson (1997) summarized the effects of oil spills on tundra ecosystems in the Prudhoe Bay area, noting that crude oil spill were relative uncommon and affected relatively small areas, whereas diesel spills were more common and affected larger areas, mostly within gravel pads.

Spills that contaminate upland dune habitats can result in oil being buried, weathered in place, or transported to other terrestrial areas; however, none of the spills included in this analysis documented these types of direct impacts. As discussed in Chapter 8 – Beaches and Tidal Flats, the supratidal zone, on the edge between upland and shoreline habitats, often includes high-tide wrack lines comprised of vegetation debris and high densities of invertebrates such as amphipods, isopods, and insects. Wrack-line invertebrates are an important prey source for terrestrial mammals, reptiles, and birds that could be affected by oil spills through direct contact with oiled wrack while scavenging, through ingestion of oiled prey, or through a decrease in suitable habitat due to the loss of wrack communities (NOAA 2016a; Hunt et al. 2002).

Spills of 500–20,000 bbl of crude oil or diesel-like oils that included documentation of impacts to terrestrial habitats or wildlife from oiling or response activities are limited to the Exxon Bayway Refinery, McGrath Beach/Berry Petroleum, and the *Universe Leader* spills (**Table 16-1**).

Table 16-1. Studies with documented or estimated impacts to and recovery of terrestrial habitats and wildlife from crude oil and diesel spills (500–20,000 bbl)

Oil Spill	Degree of Oiling (oil type)	Oil Cleanup	Terrestrial Habitats and Wildlife	Documented Effect/Impacts	Recovery (years)
1990 Exxon Bayway Refinery, Arthur Kill, NY[a]	Heavy (13,715 bbl of No. 2 fuel oil)	Yes, shoreline cleanup within marshes	Terrestrial birds, mammals, reptiles	Vertebrates and reptiles (muskrats, rabbit, domestic cat, and diamondback terrapin) were impacted by direct oiling, response disturbance, and loss of habitat.	Recovery was not assessed
1993 McGrath Beach / Berry Petroleum, CA[b] (10 km)	Heavy (2,075 bbl crude)	Yes, manual and mechanical removal of oiled sediments on the shoreline, lake, and in dunes	Dunes, Terrestrial vegetation, birds	Dunes were altered from oiled sediment removal, and manual and mechanical disturbances to habitat and vegetation.	Recovery was not assessed
1974 T/V *Universe Leader,* Ireland[c] (35 km)	Heavy (15,476 bbl Kuwait crude)	Yes, shoreline cleanup included natural sorbent (straw)	Terrestrial vegetation	Terrestrial habitats were impacted during the response as oiled straw and debris were transported from the shoreline to disposal sites.	Recovery was not assessed

[a]Desvousges et al. (1992); NOAA (1990); [b]CDPR et al. (2005); [c]Cullinane et al. (1975)

Whether terrestrial wildlife are exposed to oil can be linked directly to the geographic location of a spill and the life history of the wildlife species involved. Though little is known about the ecological and physiological effects of oil exposure on many vertebrate species, pathways of exposure for most species include: direct contact causing irritation in the skin, eyes, and other mucous membranes; oiling of fur or feathers than can affect thermoregulation and buoyancy; and internal exposure through ingestion and/or inhalation (Malcolm and Shore 2003). For terrestrial birds, secondary exposure (e.g., ingestion of contaminated prey) could be more common than exposure to the original source (e.g., oil on the substrate or vegetation) (Bergeon Burns et al. 2014). Mammals, including deer, bear, raccoon, and small rodents that feed in shoreline habitats, may be exposed to oil via inhalation, fur contact, or ingestion. Amphibians may be vulnerable via direct oil contact as water exchange through their skin is essential to their survival. Direct and indirect oiling exposure can also affect terrestrial arthropods (Bergeon Burns et al. 2014).

Oil spill response operations can cause secondary impacts to dunes and interior terrestrial habitat that are seldom directly oiled. Staging areas, access corridors, and decontamination sites are often established in or near dune areas or upland habitats adjacent to oiled shorelines (Michel et al. 2017). Due to the sensitivity of dunes to physical disturbance and the presence of wildlife in the dunes, best management practices are often prescribed to limit response impacts in these areas. For example, during the *Universe Leader* spill, responders used detergents and natural sorbents including straw, sawdust, and peat moss to soak up the onshore oil. The oil-soaked straw and other materials were collected and transported into nearby terrestrial habitat and burned. Though no quantitative data on damaged terrestrial habitat was documented, reports noted that damage caused to terrestrial vegetation consisted almost entirely of physical damage due to oil-soaked straw being drug across the cliff tops and trampling by responders (Cullinane et al. 1975).

The Exxon Bayway Refinery spill discharged No. 2 fuel oil that mostly oiled coastal marshes, though generalist terrestrial species that utilized the marsh were impacted. As shoreline cleanup operations expanded, wildlife units within the response recovered several hundred dead birds, 29 dead muskrats, one dead cottontail rabbit, one dead domestic cat, and nine live oiled turtles (diamond back terrapins) (NOAA 1990). No further documentation of impacted terrestrial wildlife was reported, and no estimates of recovery for these species were assessed.

The Berry Petroleum spill discharged crude oil from a ruptured pipeline in the vicinity of McGrath Lake in California, resulting in a contaminated slough that traversed dunes and beach before reaching the Pacific Ocean. Portions of the oiled sand dunes were disturbed by increased vehicle and foot traffic, altering dune structures and vegetation (CDPR et al. 2005). Cleanup operations for the beach, dune, and lake removed and disturbed oiled and un-oiled terrestrial and aquatic vegetation, debris, sand, soil, and sediments within the impacted area (CDPR et al. 2005). Vegetation impacts within the riparian zone included loss of >20 year old willows (*Salix* sp.). The impacted dune habitat is utilized by protected wildlife such as the silvery legless lizard (*Aniella pulchra* sp.), ESA-endangered California least tern (*Sterna antillarum browni*), and ESA-threatened western snowy plover (*Charadrius alexandrinus nivosus*) for foraging, nesting, cover, and thermal refuge. Disturbances to the native vegetation were expected to increase the potential for invasive exotic plant species to colonize the area, further disrupting the natural recovery of the impacted site. Recovery estimates were not assessed further for this spill.

One spill that was not included in **Table 16-1** was the 1999 *Erika* spill that discharged 146,000 bbl of heavy fuel oil off the coast of France. Oil spread across 400 km of shoreline and, due to weather conditions and high spring tides, the oil reached lichen communities and coastal plants in multiple habitats, including dunes. A 5-year monitoring program was initiated that included analysis of oiling and response impacts to dune vegetation. Quadrats were placed in oiled, oiled and cleaned, and incidental impact zones of white (shifting) and grey (stabilized) dunes. White dunes were generally oiled on the edges, and cleanup techniques consisted of cutting oiled vegetation and manual collection of tar balls. Impacts were limited and not noticeable within 2–3 years and incidental impacts such as heavy vehicle tracks were rapidly grown over by native grasses (Poncet et al. 2007). Grey dunes were not affected as much by direct oiling, but impacts were linked to response operations. Sites used for oil storage and vehicle access displayed decreased vegetation and had poorly recovered 5 years post impact (Poncet et al. 2007). Poncet et al (2007) suggested that, though impacts were localized, they should have been avoided because of the known low potential for recovery in grey dunes.

16.4 Summary of Data Gaps for Assessing Impacts to Terrestrial Habitats and Wildlife

There is very limited literature on the effects of marine spills of 500–20,000 bbl of crude oil, condensate, or diesel on terrestrial habitat and wildlife. From inland oil spills and experimental studies, it is known that oil residues can persist in terrestrial habitats for decades if not treated; however, there is little available literature focused on oil impacts to terrestrial habitats in the coastal setting.

To better quantify impact and recovery following oil spills for terrestrial habitats and wildlife, it is recommended that rigorous field study designs be prepared that focus solely on these features. Studying response activities, such as staging areas to determine their short- and long-term impacts to terrestrial habitats and wildlife, would benefit future responses and potentially further minimize impacts to sensitive species.

17 Commercial and Recreational Fisheries

17.1 Resource Description

This chapter focuses on the effects of spills of 500–20,000 bbl of crude oil, condensate, or diesel on commercial and recreational fishing activities. Commercial fishing includes all activity conducted for profit, while recreational fishing includes activity undertaken for pleasure or sport (subsistence activities are addressed in Chapter 21 – Vulnerable Coastal Communities). In many coastal communities, commercial fishing is an important economic sector. **Table 17-1** presents data on commercial fishery landings for all species in coastal states from NOAA Fisheries. In 2018, commercial fisheries landed more than 10 billion pounds across the coastal U.S., generating ex-vessel revenues totaling $5.8 billion.

Table 17-1. Commercial landings for all species by state (2018)

State	Million Pounds Landed	Total Landed Value (million 2019$)
Alabama	35.8	$69.2
Alaska	5,404.2	$1,880.0
California	180.4	$186.1
Connecticut	11.5	$17.0
Delaware	5.3	$10.7
Florida	108.8	$268.1
Georgia	7.7	$17.0
Hawaii	35.5	$121.2
Louisiana	1,889.1	$477.9
Maine	254.4	$667.7
Maryland	49.1	$73.9
Massachusetts	242.9	$665.2
Mississippi	628.9	$71.7
New Hampshire	10.1	$40.4
New Jersey	190.9	$175.5
New York	23.6	$49.9
North Carolina	57.1	$86.4
Oregon	313.1	$175.5
Rhode Island	392.2	$135.8
South Carolina	9.5	$23.9
Texas	84.4	$215.5
Virginia	674.7	$210.6
Washington	197.4	$244.1
Total	10,807	$5,883

Source: NOAA Fisheries Office of Science and Technology, Commercial Landings Query, available at: https://foss.nmfs.noaa.gov/apexfoss/f?p=215:200; Accessed June 22, 2020.
Note: Values converted to 2019 dollars using the Bureau of Economic Analysis Implicit Price Deflator for Gross Domestic Product.

Because recreational fishing is not conducted for profit, the scale of recreational fishing activity is best represented by the number of recreational fishing trips, as opposed to the quantity of fish landed. **Table 17-2** presents estimates of the number of ocean-based recreational fishing trips by state and fishing mode from the NOAA Fisheries Marine Recreational Information Program (MRIP). Overall, the MRIP data suggest that approximately 99 million ocean-based recreational fishing trips occur across all coastal states annually. In most states, the majority of recreational fishing occurs from shore.

Table 17-2. Ocean-based recreational angler trips by state[1]

State	Fishing Mode				
	Shore/ Shore-based structures	Charter Boat	Private and/or Rental Boat	Party Boat	Total
Alabama	3,813,951	133,491	970,346	-	4,917,788
Alaska[2]	-	-	-	-	559,577
California[3]	12,605,438	3,331,333	3,193,360	-	19,130,131
Connecticut	-	1,454	175,488	2,445	179,387
Delaware	544,455	4,007	131,672	693	680,827
Florida	20,383,131	985,955	12,208,464	-	33,577,550
Georgia	891,632	5,200	234,394	-	1,131,226
Hawaii	2,148,316	-	588,050	-	2,736,366
Maine	358,329	10,079	273,295	6,605	648,308
Maryland	120,833	8,692	249,543	10,510	389,578
Massachusetts	758,687	49,876	909,261	11,484	1,729,308
Mississippi	-	3,816	302,415	-	306,231
New Hampshire	83,050	12,010	188,422	28,960	312,442
New Jersey	3,521,027	146,364	1,814,343	76,516	5,558,250
New York	1,309,966	13,618	2,034,157	140,907	3,498,648
North Carolina	8,924,809	110,450	988,965	-	10,024,224
Oregon[3]	-	299,908	729,285	-	1,029,193
Rhode Island	1,007,968	4,328	835,210	2,551	1,850,057
South Carolina	6,663,702	54,911	749,226	-	7,467,839
Texas	-	-	-	-	1,121,586
Virginia	1,142,198	6,597	216,102	2,497	1,367,394
Washington[3]	32,158	234,775	595,574	-	862,506
Total	64,309,650	5,416,864	27,387,572	283,168	99,078,416

Notes:

1. Values for states on the Atlantic and Gulf of Mexico (excluding Texas) reflect recreational fishing activity in 2018. Values for the Pacific coast reflect average fishing activity over the 2013-2017 period. The values for Alaska and Texas reflect average activity from 2012 through 2016.
2. Data for Alaska and Texas are not disaggregated by category. Therefore, only the total is reported here.
3. The source data for California, Oregon, and Washington combine the charter and party boat categories. For the purposes of reporting, they are shown as charter boat trips here.

Sources: For Atlantic and GOM states (excluding Texas): NOAA Fisheries. MRIP Recreational Fisheries Statistics Query. Available at: https://www.st.nmfs.noaa.gov/recreational-fisheries/data-and-documentation/queries/index; accessed June 22, 2020. For Pacific states: Pacific Recreational Fisheries Information Network (RecFIN). CEE001 Effort Estimates Report and CTE001 Recreational Fishery Catch Estimate Report, as referenced in IEc (2018). For Alaska and Texas: NOAA Fisheries. MRIP Fisheries Statistics Query. 2012-016 and Alaska Department of Fish and Game (2018), as referenced in IEc (2018).

Though recreational fishing trips are not conducted for profit, they still provide value to participants. Despite the lack of an observable market price, this recreational value can be estimated through economic methods, such as travel cost analysis and contingent valuation. Travel cost methods estimate recreation fishing values based on the time and money spent to travel to and participate in a fishing trip, while contingent valuation methods directly survey recreational fishers on their willingness to pay (WTP) for access to recreational fishing opportunities.

Table 17-3 presents recreational fishing use value estimates for various coastal locations across the U.S. This table includes all of the studies used in BOEM's Offshore Environmental Cost Model (OECM) as well as additional studies identified in Oregon State University's Recreation Use Values Database (Oregon State University 2016). As the table shows, recreation fishing value estimates are highly variable, reflecting location-specific factors such as site quality, amenities (e.g., a fishing pier), and crowds.

Table 17-3. Recreational fishing trip valuation estimates

Study	Location	Estimated Value per User Day (2019$)
Haab et al. (2000)	North Carolina	$23
Haab et al. (2000)	South Carolina	$10
Haab et al. (2000)	Georgia	$4
Haab et al. (2000)	Florida (South Atlantic)	$17
Haab et al. (2000)	Florida (Gulf)	$66
Haab et al. (2000)	Alabama	$2
Haab et al. (2000)	Mississippi	$5
Haab et al. (2000)	Louisiana	$17
McConnell and Strand (1994)	Delaware	$21
McConnell and Strand (1994)	Maryland	$51
McConnell and Strand (1994)	Virginia	$88
McConnell and Strand (1994)	New York	$111
McConnell and Strand (1994)	New Jersey	$64
English and McConnell (2015)	North Gulf (Louisiana, Mississippi, Alabama, Florida)	$38–$42
Haab et al. (2006)	Washington	$80
Haab et al. (2006)	Oregon	$34
Haab et al. (2006)	Northern CA	$122
Haab et al. (2006)	Southern CA	$298
Johnston et al. (2002)	New York	$63
Hamel et al. (2000)	Alaska	$152
Hausman et al. (1995)	Prince William Sound, AK	$271
Soto et al. (2014)	San Francisco Bay Area, CA	$44
English (2010)	San Francisco Bay Area, CA	$60

Note: Values converted to 2019 dollars using the Bureau of Economic Analysis (BEA) Implicit Price Deflator for Gross Domestic Product. Source: IEc (2019) and Oregon State University (2016).

17.2 Impacts of Oil Spills on Commercial and Recreational Fishing

Oil spills can adversely impact commercial and recreational fisheries through a variety of pathways. Focusing specifically on commercial fisheries, direct oil impacts can result in reduced landings immediately following a spill to the extent that oiling affects mature adult fish. Spills can also result in reduced landings years in the future to the extent that larval or juvenile fish stages are harmed.

Additionally, median-range oil spills may result in the temporary closure of commercial and recreational fisheries. Such closures can greatly reduce commercial fishery landings and recreational fishing activity due to real or perceived contamination or health risks. If commercial fishing operators are able to maintain operations at all during such closures, they may need to travel farther and otherwise expend more time and resources to achieve the same catch. Similarly, recreational fishing participants may need to travel farther and to less preferred sites to find recreational fishing opportunities. Finally, median-range oil spills, particularly those that generate large amounts of media coverage, can result in reduced consumer demand for seafood due to real or perceived health risks. Reduced demand can lower the market value of commercial landings from fisheries near a spill site, reducing revenues to commercial fishing operators.

Recreational fishing typically recovers more quickly than commercial fishing following an oil spill due to the different objectives of the two activities. Though the commercial fishing industry catches fish to sell for consumption, most participants in recreational fishing have a variety of non-consumptive motivations that make them more likely to return sooner (Sumaila et al. 2012). The magnitude of spill impacts on commercial and recreational fisheries are influenced by a variety of factors, including the following:

- **Spill Location**. Spills near valuable fisheries will result in greater impacts than spills in areas with less valuable fisheries (or less fishing activity).
- **Season**. Variation in harvest seasons and fish locations by season can influence the impacts of spills on fishing resources.
- **Spill volume**. Larger spills with larger surface sheens and a higher concentration of oil in the water column are more likely to result in fish exposure to oil and fishing closures.
- **Effectiveness of response actions**. Response actions, such as inlet protection strategies to keep oil from entering estuaries and lagoons, use of dispersants, and mechanical on-water recovery, can influence the impacts of spills on commercial and recreational fishing.
- **Environmental conditions that affect oil transport**. The presence of wind, waves, and ocean currents can increase the dispersion of spilled oil and result in greater damages to commercial and recreational fishing.

Ideally, the recreational fishing damages caused by an oil spill would be estimated using data on: 1) the baseline level of recreational fishing activity; 2) the change in the level of recreational fishing activity following the spill; and 3) the per unit (e.g., per fishing day or per fishing trip) value that individuals place on these activities in the baseline and during the impact period. Similarly, commercial fishing damages would ideally be estimated based on the baseline level of commercial fishing revenue and the change in revenue attributable to the spill. In practice, it is often difficult to isolate the impact of an oil spill on commercial fisheries, because fish populations are subject to considerable natural variation based on factors such as predation, disease mortality, water temperature, and turbidity.

The sections below describe median-range oil spills with documented impacts to commercial or recreational fishing. **Table 17-4** summarizes the documented impacts to commercial and recreational fishing from these spills. The table includes quantified economic losses, if available. The *Cosco Busan* spill was a heavy oil, but it is included here because of the paucity of fishing data from other spills.

Table 17-4. Studies with documented or estimated impacts to and recovery of commercial or recreational fishing from crude oil, No. 2 fuel oil, and heavy fuel oil spills (500–20,000 bbl)

Oil Spill	Degree of Oiling (oil volume & type)	Commercial Fishing		Recreational Fishing		Spatial Extent of Quantified Impacts	Recovery
		Documented Effects/Impacts	Quantified Impacts ($)	Documented Effects/Impacts	Quantified Impacts ($)		
M/V *Cosco Busan,* San Francisco Bay, CA[a]	1,275 bbl of heavy fuel	Commercial fishery closure 14-29 November 2007.	No quantified impacts.	58,500 lost shore fishing days. 11,000 lost boat fishing days.	Economic losses of $2.6 million for shore fishing and $0.66 million for boat fishing.	San Francisco Bay area: Sonoma County to San Mateo County	3 months
T/B *North Cape,* South Kingston, RI[b]	19,714 bbl No. 2 fuel oil	None documented.	None documented.	3,305 party boat and charter boat angler-trips lost.	Boat-based fishing losses of $414,000.	In and around Block Island Sound, Rhode Island	6 months
M/V *Glacier Bay,* Cook Inlet, AK[a]	3,780 to 4,942 bbl Alaska North Slope crude	Reduced catch for the drift gillnet and setnet fleets for the sockeye salmon fishery in Cook Inlet.	For the drift gillnet fleet, interviews with fishermen suggested $81.9 million in damages. Historical catch data and ADF&G's indicator fishery suggested damages of $25.4 million to $34.8 million. For setnet fisheries, interviews with fishermen suggest damages of $23 million.	Impacts were not identified for recreational fishing.	Not applicable.	Cook Inlet	<1 year
All quantified impacts converted to 2019 dollars using the BEA Implicit Price Deflator for Gross Domestic Product. *Indicates that the source studies did not explicitly quantify the duration of impacts. The provided estimate indicates the number of years with quantified impacts in the source studies, and thus reflect a lower bound on the duration of impacts.							

Source: [a]Cosco Busan Oil Spill Trustees (2012); [b]NOAA et al. (1999); [c]Northern Economics (1990)

17.2.1 M/V *Cosco Busan*

On the morning of 7 November 2007, the container ship *Cosco Busan* struck a tower of the San Francisco-Oakland Bay Bridge. The gash in the hull of the vessel created by the allision caused the release of 1,275 bbl of heavy fuel oil into the San Francisco Bay over the course of approximately 53 minutes. Winds and currents caused the spill to spread rapidly and moved some of the oil outside of the Bay, impacting an area from Half Moon Bay to Point Reyes. Seven days after the spill, on 14 November, the State of California closed the commercial and recreational marine fisheries from Point Reyes to San Pedro Point. The closure was lifted on November 29, and an advisory was issued stating the following: "It is possible that residual oil may remain on the water over the next several months. Recreational and commercial fishers should avoid exposure of their take to these residual pockets" (Cosco Busan Oil Spill Trustees 2012). Though this spill was of a heavy fuel oil, it is included because of the paucity of data on the impacts of median-range spills to recreational fishing resources.

As part of the NRDA following the spill, the Trustees estimated the economic losses to recreational anglers. In the first stage of the analysis, the number of fishing days lost due to the spill was estimated based on data from the California Recreational Fisheries Survey (CRFS), specifically CRFS data for the San Francisco Bay area, including all saltwater recreational angling within 4.8 km of the coast from Sonoma County to San Mateo County. Using CRFS data from the November to January period for the 2 years prior to the spill and 2 years after, the Trustees developed baseline estimates of recreational fishing activity. Comparing these baseline estimates to CRFS data for the 3 months after the spill (November 2007 to January 2008), the Trustees estimated 58,500 lost shore fishing days and 11,000 lost boat fishing days. To value these lost trips, the Trustees relied upon a benefits-transfer approach and adapted estimates from the literature to arrive at a value of $44.30 per day for shore fishing and $59.65 per day for boat fishing (Cosco Busan Oil Spill Trustees 2012). Based on these values, economic losses were estimated to be $2.6 million for shore-based fishing and $0.66 million for boat-based fishing (Cosco Busan Oil Spill Trustees 2012).

17.2.2 T/B *North Cape*

On 19 January 1996, the *North Cape*, a tank barge carrying 94,000 bbl of home heating oil, went aground near Moonstone Beach in South Kingstown, Rhode Island, after the tugboat towing it caught fire and was abandoned during a storm. With winds reaching 93 km/hour, an estimated 19,714 bbl of home heating oil were spilled. The Trustees determined that boat-based recreational fishing was the only category of recreational use with measurable impacts. To assess these impacts, Trustees conducted interviews with charter and party boat captains, who provided information on the number of forgone boat trips associated with the spill. Based on the information collected, Trustees estimated that 3,305 boat-based recreational fishing trips were lost (NOAA 1999). Applying benefits transfer, the Trustees estimated boat-based fishing losses of $414,000 (NOAA 1999).

17.2.3 T/V *Glacier Bay*

On 2 July 1987, the tanker *Glacier Bay* hit a submerged obstacle while transporting 380,000 bbl of Alaska North Slope crude oil to the offloading facilities of Kenai Pipeline Company at Nikiski. The resulting spill led to the release of 3,780 to 4,942 bbl of crude oil, creating an oil slick at least 16 km long within Cook Inlet (Northern Economics 1990). At the time of the spill, the commercial fishing season for sockeye salmon had just gotten underway in Cook Inlet.

The spill resulted in commercial fishing damages for both the drift gillnet fleet in Cook Inlet and the setnet fleet. Based on interviews with commercial fishers, the drift gillnet fleet experienced damages of approximately $81.9 million (Northern Economics 1990). Using an alternative approach based on the historic average daily catch and the Alaska Department of Fish & Game's indicator fishery yielded a lower estimate of losses. Applying this approach, estimated losses to the gillnet fleet were in the range of $25.4 million to $34.8 million. Estimated damages for setnet fisheries were based on interviews with 58 setnet fishermen. The responses for this group were more varied than for drift gillnet fishermen, with the variance attributed to differences in location. Overall, the interviews suggested damages of $23 million for setnet fisheries (Northern Economics 1990).

Trustees also interviewed charter boat operators to gauge potential impacts to the region's sport fishing industry. Based on these interviews, the spill did not result in sport fishing damages. For example, the President of the Homer Charterboat Association indicated that its fleet was not impacted, as they were not subject to time/area closures, their boats and gear were not oiled, and the fish caught by their customers were not fouled by oil. Individual charter operators provided similar feedback (Northern Economics 1990).

17.3 Summary and Information Need for Assessing Impacts to Commercial and Recreational Fishing

Oil spills can adversely impact commercial and recreational fisheries through a variety of pathways. Specifically, oil spills may reduce commercial fishery landings and reduce the quantity or value of recreational fishing trips due to actual or perceived degradation in resource quality. The magnitude of spill impacts on commercial and recreational fisheries depends on a variety of factors, including spill size, location, and duration of any fishery closures. All else being equal, median-range spills near valuable fisheries result in greater impacts.

Impacts to commercial and recreational fisheries have been documented for only a limited number of median-range spills. Additional research on several research questions would therefore provide useful insights into spill-related impacts for commercial and recreational fisheries. For example, given differences in the mobility of shellfish and finfish, impacts to these fisheries are likely to differ, but a comparative assessment of the magnitude of spill impacts for these fisheries does not exist. In addition, because perceptions of oiling can affect fisheries even in areas with no documented oiling, additional research into the factors that affect these perceptions (i.e., factors that increase or decrease these perceptions) would inform understanding into the breadth of fisheries impacts associated with potential spills. Additional research focusing on median-range spill impacts to recreational fisheries in general would also provide helpful insights, as the available studies on these impacts are few in number. Finally, many existing studies of both commercial and recreational fisheries do not explicitly quantify the duration of spill impacts. As a result, the impacts of any future spill on commercial and recreational fisheries are considerably uncertain and are likely to depend on spill-specific characteristics.

18 Employment and Income

18.1 Resource Description

This chapter focuses on the effects of spills of 500–20,000 bbl of crude oil, condensate, or diesel on employment and income among populations in affected areas. Coastal and marine resources both directly and indirectly support a wide variety of industries and the employment of a large number of workers nationwide. Industries supported by coastal and marine environments include industries directly dependent on resources, such as the oil and gas industry and commercial fishing, as well as downstream industries, such as tourism and hospitality near the ocean. All of these industries and economic activities provide jobs and income for local workers.

The National Ocean Economics Program provides key metrics on the "coastal economy" and the "ocean economy," including employment and income by region and by industry. The ocean economy represents only industries directly dependent on marine resources, including construction, resource extraction, tourism, and transportation, while the coastal economy represents all economic activity within the coastal zone, even if not directly dependent on marine resources. Both segments of the economy are potentially vulnerable to oil spills and the corresponding disruption in economic activity. As shown in **Table 18-1**, the ocean economy can represent a large proportion of the overall coastal economy (e.g., as much as 19% of jobs and 20.4% of wages in the North Pacific).

Table 18-1. Measures of 2016 employment and income by region for ocean and coastal economies

Coastal Region	Ocean Economy		Coastal Economy		Ocean as Percent of Coastal Economy	
	Employment (Thousands)	Wages (Millions 2019$)	Employment (Thousands)	Wages (Millions 2019$)	Employment (%)	Wages (%)
West Coast	733	31,700	16,100	1,140,000	4.56	2.78
Gulf of Mexico	675	39,200	6,960	391,000	9.70	10.0
Southeast	324	8,690	5,670	285,000	5.72	3.05
Mid-Atlantic	788	31,500	14,800	1,050,000	5.32	2.99
Northeast	256	10,200	4,670	331,000	5.47	3.07
North Pacific	47.6	2,950	251	14,400	19.0	20.4
Pacific	118	4,790	638	33,000	18.5	14.5
Total	**2,940**	**129,000**	**49,100**	**3,250,000**	**5.99**	**3.97**

The coastal regions include the following states:

- West Coast: California, Oregon, and Washington
- Gulf of Mexico: Alabama, Florida Gulf Coast, Louisiana, Mississippi, and Texas
- Southeast Region: Florida Atlantic Coast, Georgia, North Carolina, and South Carolina
- Mid-Atlantic Region: Delaware, Maryland, New Jersey, New York, Pennsylvania, and Virginia
- Northeast Region: Connecticut, Maine, Massachusetts, New Hampshire, and Rhode Island
- North Pacific: Alaska
- Pacific: Hawaii

Source: National Ocean Economics Program (NOEP) (2020a; 2020b). Coastal Economy Data accessed at: https://www.oceaneconomics.org/Market/coastal/coastalEcon.asp. Ocean Economy Data accessed at: https://www.oceaneconomics.org/Market/ocean/oceanEcon.asp?ci=N. Wage values inflated from 2012$ to 2019$ using the GDP Deflator.

Not only do ocean-based industries comprise a large sector of the overall coastal economy, but they also represent a large proportion of overall sectoral output. **Table 18-2** displays the total employment and labor income of several ocean-based industries relative to the overall coastal industry. For example, the natural resources and mining industry in coastal counties is made up of largely by ocean-based resources and extraction (e.g., oil and gas extraction and support activities), comprising 71.8% of all wages in 2016, and 43.3% of all jobs. The ocean-based tourism and recreation industry also comprises a large segment of the overall leisure and hospitality industry in coastal areas, with 29.5% of wages and 30.6% of leisure and hospitality employment in coastal areas. Overall, because employment in ocean-based industries represents a larger proportion of total coastal employment than total wages (5.10% relative to 3.4%), the data show that the average worker in ocean-dependent industries earns a lower wage than the average worker in the overall coastal economy–especially in the living resources and minerals sectors.

Table 18-2. Measures of 2016 employment and income by industry for ocean and coastal economies

Ocean Industries			All Coastal Industries			Ocean Industries Percent of Coastal	
Ocean Economic Sector	Employment (Thousands)	Wages (Millions 2019$)	Coastal Economic Sector	Employment (Thousands)	Wages (Millions 2019$)	Employ-ment (%)	Wages (%)
Construction	42.6	3,310	Construction	2,760	187,000	1.54	1.77
Living Resources and Minerals	215	25,700	Natural Resources and Mining	496	35,800	43.3	71.8
Ship and Boat Building	150	10,900	Manufactur-ing	4,160	340,000	3.60	3.19
Tourism and Recreation	2,130	57,700	Leisure and Hospitality	6,960	195,000	30.6	29.5
Transporta-tion	406	31,400	Trade, Transporta-tion, and Utilities	11,200	576,000	3.62	5.45
NA	NA	NA	Other	32,100	2,400,000	NA	NA
Total	**2,940**	**129,000**	**Total**	**57,700**	**3,740,000**	**5.10**	**3.45**

Source: NOEP (2020a; 2020b), Coastal Economy Data. Accessed at: https://www.oceaneconomics.org/Market/coastal/coastalEcon.asp; NOEP (2020a; 2020b), Ocean Economy Data. Accessed at: https://www.oceaneconomics.org/Market/ocean/oceanEcon.asp?ci=N. Wage values inflated from 2012$ to 2019$ using the GDP Deflator (Table 1.1.9) from the BEA Analysis National Income and Product Accounts, last updated June 25, 2020.

Though the above two tables show large portions of local coastal economies that are *directly* dependent on marine resources, the data shown do not capture the entire cumulative impact of the jobs and wages paid within the industries shown. In addition to the direct employment and wages paid within a given sector, each industry also makes intermediate purchases from other industries, further increasing the employment and wages supported by a single industry. Further, each worker in the industries shown in **Table 18-2** also supports other sectors of the economy through the spending of labor income (e.g., on food, housing, and entertainment).

18.2 Impacts of Oil Spills on Employment and Income

18.2.1 Impacts of Oil Spills of 500–20,000 bbl on Employment and Income

The existing literature on the employment and income impacts of spills of crude, condensate, or diesel includes no studies focusing on median-range spills. Instead, the existing literature examines the employment and income impacts of larger spill events, such as the *Deepwater Horizon* and *Exxon Valdez* spills. These studies are summarized in Michel (2021). All else equal, the employment and income impacts associated with median-range spills would likely be smaller than those associated with larger spill events.

18.2.2 Characterization of Impacts

Impacts to employment and income can take many pathways after an oil spill. This section separates impacts into two larger categories of impacts: direct and indirect. Direct impacts include impacts to industries directly disrupted by a spill. For example, severe oiling or perceived tainting of commercially harvested species may cause a stoppage of some types of commercial fishing, resulting in direct job loss in the commercial fishing sector. Indirect impacts include secondary impacts that are still caused by the spill but are not directly related to a disrupted sector (e.g., job losses for firms that provide supplies to the commercial fishing industry). Direct impacts in the literature mainly focus on the following four sectors:

- **Commercial fishing.** This sector may be directly impacted by the decrease of either landings or sales if the resource is either depleted or demand decreases due to real or perceived health risks of consumption. This may result in both reductions in the number of people working in the commercial fishing industry and reductions in income earned by those that continue to fish.
- **Recreation and tourism.** Coastal recreation and tourism may be disrupted if demand drops for recreational tourism experiences and services in response to a spill. The disruption of industries that provide these services may result in layoffs of employees and/or reductions in pay to compensate for reduced demand.
- **Commercial shipping.** Commercial shipping may be disrupted if an oil spill and corresponding response operations cause the closure or blockage of a major port or waterway. The commercial shipping industry may experience employment or wage decreases to compensate for the reduction in shipping activity.
- **Oil and gas.** The oil and gas industry may be disrupted in the event of an oil spill if there is a planned (or unplanned) stoppage of drilling, transportation, or other activities. The most salient example of this was the Gulf of Mexico drilling moratorium after the *Deepwater Horizon* spill. Rig workers and other employees may face job losses or pay reductions in response to a stoppage in work.

Other chapters in this document focus more broadly on several of these sectors (see Chapter 17 – Commercial Fishing and Chapter 22 – Recreation and Tourism). This chapter focuses specifically on jobs and income impacted by oil spills in these and other sectors. This includes direct job and employment impacts in these sectors and the indirect and induced impacts for other industries.

In addition to the mostly negative industry impacts related to an oil spill, response activities can provide an influx of both government and private economic activity that can mitigate employment and income loss. Thus, the duration of both the spill response and the economic shock of the spill itself are important for fully characterizing employment and income impacts. The spill response effort can bring new jobs and income to the impacted region in the short term, but the negative impacts of the spill may outlive the short-term stimulus associated with the response.

Factors that influence the magnitude of spill impacts on employment and income include the following:

- **Location**. Spills near large ports or centers of economic activity will likely result in greater disruptions of economic activity, with a higher magnitude of employment and income impacts.
- **Season**. A spill and its associated response during the summer will likely have a higher impact on recreation and tourism industries and corresponding reductions in employment and income in those industries than a spill occurring during off-peak months.
- **Spill volume**. All else being equal, larger spills will likely result in higher reductions in employment and income in affected industries and a larger response effort. A larger response effort may result in a high number of short-term high-paying jobs.
- **Effectiveness and intensity of response actions**. Timely, effective responses can not only mitigate the impacts of the oil spill but can also inject additional economic activity into the local economy.

18.3 Summary and Information Needs for Assessing Impacts to Employment and Income

Overall, the impacts of median-range oil spills can be disruptive to local and regional economies. Specifically, job and income losses can occur in industries both directly and indirectly affected by a spill. This includes industries that are dependent on marine resources, such as commercial fishing, recreation and tourism, marine shipping, and the oil and gas industry, as well as other sectors connected to these industries. Oil spill response efforts (either in jobs or other monetary compensation), however, can be a complicating and important factor when trying to assess the overall employment and income impacts of a spill.

The literature includes no studies quantifying the employment and income impacts of median-range oil spills. In addition to assessing the magnitude of such impacts, economic research that develops in this area could examine several other important questions. For example, researchers could assess income impacts for individuals who are temporarily unemployed due to a spill but who find work within several months of a spill event. Detailed study of these individuals could provide insights into whether they find employment in the same industry and what their income trajectory is over time relative to what it would have been in the absence of a spill. In addition, future studies could examine the factors influencing the duration of spill-related employment and income effects, other than the duration of the spill itself. Such factors may include the size and structure of the local economy, as well as the presence or absence of specific policy interventions to minimize business interruption for affected industries or to connect displaced workers with other employment opportunities.

19 Cultural Resources

This chapter focuses on the effects of spills of 500–20,000 bbl of crude oil, condensate, or diesel on cultural resources in coastal environments, including historic properties, coastal archaeological resources, traditional subsistence practices, and intangible cultural heritage. Cultural resources include a broad range of "stories, knowledge, people, places, structures, and objects, together with their associated environment, that contribute to the maintenance of cultural identity and/or reveal the historic and contemporary human interactions with an ecosystem" (Ball et al. 2015). As such, cultural resources may be tangible (e.g., historic structures, monuments, places, and landscapes) or intangible (e.g., knowledge, songs and dances, practices, and values). Coastlines throughout the U.S. are home to both tangible and intangible cultural resources. In some states, the greatest concentration of such resources is located within the coastal zone.

The management and protection of cultural resources fall within the scope of numerous federal and state agencies, laws, and policies, including but not limited to the National Historic Preservation Act of 1966 (NHPA), the American Antiquities Act of 1906, and the Native American Graves Protection and Repatriation Act (NAGPRA). This chapter focuses on cultural resources primarily located in coastal environments. Oil spill impacts on certain types of cultural resources, such as historic shipwrecks, have been studied more intensively in marine environments. These resources are discussed separately in Chapter 20 – Marine Archaeological Resources.

19.1 Historic Properties

Historic properties include buildings, structures, objects, and landscapes that collectively make up the historic environment. Properties that possess the requisite significance and integrity are eligible for the National Register of Historic Places (National Register). As defined in the NHPA, historic properties are "any prehistoric or historic district, site, building, structure, or object included in, or eligible for inclusion on the National Register" 16 U.S.C. § 470(w)(5). This includes artifacts, records, and material remains that are related to the district, site, building, structure, or object.[3]

Oil spill impacts to historic properties are typically characterized in terms of the criteria for National Register eligibility. Potential effects resulting from oil spills that may impact eligibility include effects on:

- The "physical features" of the property "necessary to convey the aspect of prehistory or history with which it is associated" (NPS 1997), such as design features of a structure that make it illustrative of its historic context;
- The information potential of the property (e.g., the intactness of key data contained in the property) as it relates to questions of historical, anthropological, and archaeological interest, such as characteristics of an historic farm that would contribute to testing a hypothesis about past agricultural practices;
- The setting (i.e., the physical environment) of the property in ways that alter its character or the relationship between the property and its surrounding environment, such as a residence that derives its historic significance from a picturesque coastal setting; and
- The integrity of the property and of its essential physical features, including their presence, visibility, condition, and other relevant considerations.

[3] For the purposes of this chapter, archaeologically significant historic properties are discussed separately.

Because National Register eligibility criteria are primarily qualitative, specific metrics for oil spill impacts are not available. The magnitude of spill impacts may be influenced by the location, scale, and nature of the historic property; its physical composition and condition; the nature of the property's setting; the location and size of the spill; the type and effectiveness of response actions; environmental conditions that affect oil transport; and conditions that affect the assessment and response of conservationists and other management professionals.

Cultural resources are inherently irreplaceable. Unlike biological resources, there is no natural recovery process. Some forms of damage, such as those that affect the surface, appearance, or chemical composition of materials, may be mitigated through the application of appropriate in-situ or laboratory-based conservation techniques (Jariwala and Striegel 2020). For other forms of damage, including the loss of physical features and damage that cannot be addressed through conservation, as well as damage that alters the spatial and physical association of artifacts, sediments, and other components of the archaeological record, recovery and restoration may not be possible. Due to these distinctions from biological resources, time to recovery for cultural resources has not been estimated in the literature.

19.1.1 Historic Properties Types and Inventories

Historic properties in coastal environments that may be subject to oil spill impacts include historic buildings and structures such as lighthouses, military installations, residences, piers and beachfront facilities, highways, and bridges. Specific types of historic properties with potential National Register eligibility include historic aviation properties (e.g., aircraft, military air stations, and missile launch sites), aids to navigation, battlefields, cemeteries and burial places, designed landscapes, rural landscapes, properties associated with significant persons, and traditional cultural properties (see NPS National Register bulletins for additional examples and descriptions). Historic shipwrecks may also be impacted by oil spills; these are discussed in Chapter 20 – Marine Archaeological Resources.

Under NHPA, both pre-contact (i.e., prior to the arrival of Europeans) and post-contact archaeological resources are considered historic properties. Archaeological resources are defined as any material remains of human life or activities that are at least 50 years of age and can provide scientific or humanistic understanding of past human behavior through scientific or scholarly study (30 CFR 550.105). Archaeologists and specialists in adjacent fields examine sites, landscapes, and artifacts to understand past human behavior, cultural adaptation, and related topics. Significant archaeological resources are those that meet the criteria of significance and integrity for eligibility on the National Register, as defined in 36 CFR 60.4. This section discusses archaeological resources along the coastline, including the intertidal zone. Submerged archaeological resources are discussed separately in Section 20.1. Coastal archaeological resources contribute specifically to scientific understanding of human coastal adaptation, maritime population dispersals, and variation in subsistence and production based on marine and coastal resources. They provide a record of past human interaction with shoreline and marine environments, and of economic, social, and cultural responses to changing coastal habitats (Sassaman et al. 2014).

Coastal archaeological resources may include sites, landscapes, artifacts, and other objects. Specific types of sites often located in the coastal zone include settlements, camps, middens, trails, and pathways. These sites may contain artifacts associated with subsistence activities, trade and exchange, sacred and ceremonial practices, and other aspects of coastal lifeways. In addition to pre-contact sites, many coastal archaeological sites relate to post-contact histories and processes such as colonization and industrialization. Both pre-contact and post-contact sites may include human remains.

Properties listed in the National Register are inventoried in databases maintained by the National Archives and Records Administration and the National Park Service (NPS) (see **Figure 19-1**). At the state level, State Historic Preservation Offices (SHPOs) maintain records of historic properties, including those eligible, ineligible, or not yet evaluated for National Register eligibility. Other inventories of historic properties have been generated by studies and surveys. For example, ICF International et al. (2013) identified 2,383 coastal cultural resources on the Pacific coast of the contiguous U.S. These inventories may be used to identify properties that may be impacted by a spill or response actions.

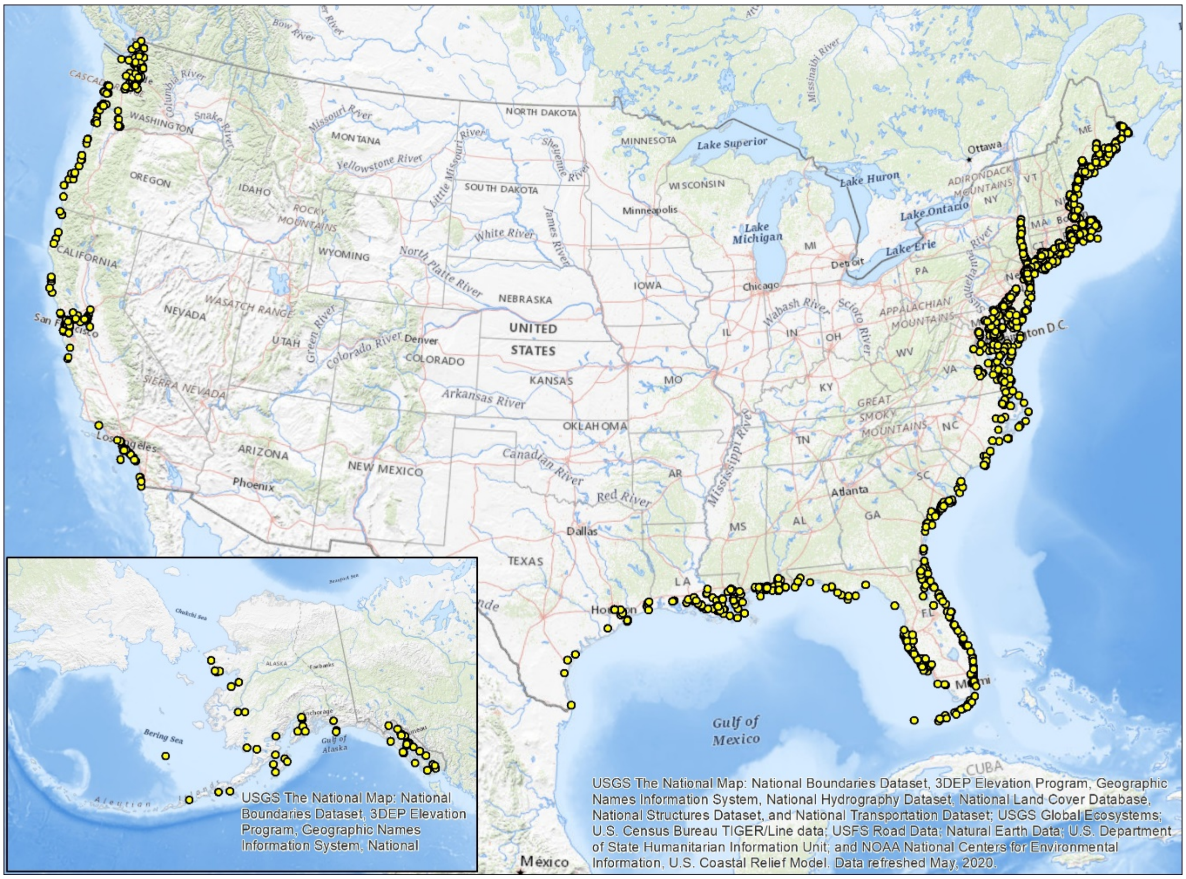

Figure 19-1. Map of public historic properties (excluding sensitive or restricted properties) listed on the National Register of Historic Places

Map shows sites located from 0.16 km inland to 16 km offshore on the coast of the contiguous U.S. and Alaska. This dataset has not been fully updated to reflect the complete list of properties on the National Register. From Stutts (2014).

Records of and information about coastal archaeological resources may be found in various databases and inventories. These include site inventories maintained by SHPOs, which inventory historical cultural resources generally without regard to the NRHP historical significance criteria. To protect resources, full access to these databases is often restricted. Other inventories of archaeological resources include the National Archaeological Database (NADB), maintained by the Center for Digital Antiquity at Arizona State University, and the Digital Index of North American Archaeology (DINAA), a collaborative effort of several universities and research institutions. Regionally focused archaeological surveys are another source of information about archaeological resources. Such surveys vary widely in scale, level of detail, and data quality. Survey reports are often published, but unpublished reports may be difficult to access. Those that are not publicly available may be on file with SHPOs.

19.1.2 Effects of Spills on Historic Properties

An oil spill may physically alter, damage, or destroy part or all of a historic property. Oil may come into direct physical contact with historic properties, causing deterioration or destruction of historically significant materials. The resulting damage may prevent or impede scientific study, such as chronometric analysis of materials (e.g., radiocarbon dating). These effects may result in the loss of significant scientific, historic, and cultural information contained in or associated with the historic property.

Oil released from spills may interact with historic properties in a variety of ways depending on context and the nature of the property. Key factors include the location of the resource and its surrounding environment, such as topography, geology, marine currents, and tidal range. Oil may accumulate more heavily and persist in areas at the high tide line and upper shore, where resources are more sheltered from waves. The same factors may contribute to preservation of the archaeological record in these areas, meaning better-preserved sites are susceptible to oil accumulation.

Additionally, the spill may affect the property's visual, auditory, atmospheric, and other environmental characteristics. A property may also become dissociated from its setting, or its setting's characteristics may be altered in ways that are out of character with the property. Any of these effects may interfere with the historic property's significance and integrity as they relate to National Register eligibility criteria (described above).

Archaeological sites and artifacts may consist of both organic and inorganic materials. Oil may be absorbed by organic materials, such as shells and plant remains. Oil contamination of these materials may make geochemical analysis, including radiocarbon dating, more difficult. It may also impose human health risks if materials are excavated, analyzed, and stored. Oil may also accumulate on inorganic components of archaeological sites and artifacts. If accumulated oil penetrates archaeological sites or deposits, it may be difficult or impossible to remove without disturbing the site, which would likely affect its integrity and information potential.

In addition to direct impacts from oil contact, historic properties may be affected by spill response actions. Ground-disturbing activities, including excavation, construction, cleanup, and staging of response actions, may directly or indirectly impact historic properties.

The literature includes no studies that assess the impacts of median-range oil spills to historic properties. A limited number of studies, however, have documented such impacts in the context of larger oil spills (>20,000 bbl). These studies are summarized in Michel (2021).

19.1.3 Impacts of Oil Exposure on Preservation of Historic Properties

Impacts on the preservation of historic properties can result from spills directly due to oil exposure as well as indirectly due to response activities. Potential impacts have been discussed in the literature, but examples of documented impacts on preservation are not readily available.

Direct impacts from oil exposure may vary according to the nature of the historic property, the materials used in its construction, and the conditions of exposure. Because of their age and manufacturing techniques, historic building materials may be more vulnerable to damage from oil contact than modern materials. For example, historic brick is softer, more porous, and more friable than modern brick. Oil can trap moisture in and corrode materials such as brick and lime mortar, as well as discolor or otherwise affect their appearance (Chin and Church 2011). In marine environments, oil-degrading microorganisms may produce acidic substances that can degrade masonry, stucco, metals, wood, and other site components (Gu et al. 1998). Indirect impacts of spills may result from the effects of cleaning materials, techniques, and activities taking place in and around historic properties.

Detailed studies describing impacts of oil spills on coastal archaeological resources are limited to a few crude oil spills >20,000 bbl. The *Exxon Valdez* and *Deepwater Horizon* spills resulted in extensive studies of both direct and indirect impacts to cultural resources. For impacts resulting from median-range spills, the literature is minimal; therefore, information from two median-range spills of heavy fuel oil in Alaska is provided.

The 1997 *Kuroshima* spill (928 bbl of heavy fuel oil) impacted a single known archaeological site in Summer Bay, Unalaska Island, Alaska, dating to approximately 2,500 before present (NOAA 2002; Knecht and Davis 1999). The site, UNL-92, is a concentration of lithic and faunal artifacts, including tools and fish, mammal, bird, and shellfish remains, along with remnants of seasonal shelters (Knecht and Davis 1999). The site surface was coated with a layer of windborne oil, which subsequently formed "patties" of oil and sand up to 1 cm thick. Surface oil was no longer visible about 7 months after the spill, but some oil patties had been reburied. Archaeologists excavated most of the upper 1 m of archaeological deposits to ensure that oil would not contaminate deeper portions of the site. Pre- and post-spill radiocarbon dating of a hearth feature indicated that oil had not penetrated the site below the surface, and archaeologists were "reasonably confident" that the site had not been subjected to long-term damage (Knecht and Davis 1999, p. 17).

For a preassessment study for the 2004 *Selendang Ayu* spill (8,300 bbl of intermediate fuel oil and marine diesel) near Unalaska Island, Alaska, archaeologists contracted by the U.S. Coast Guard reported that no archaeological resources had been impacted (Kohout and Meade 2008). During response efforts, an on-site archaeologist and support staff in Anchorage monitored sites and conducted consultation, training, and planning activities (Owens et al. 2005). Any reports generated from these efforts are not publicly available. However, the final NRDA plan for the *Selendang Ayu* spill reported that no cultural resource impacts had been identified (NOAA 2016).

For other median-range spill incidents, the capacity to study potential damage to archaeological and other cultural resources may be constrained despite recognition that such resources are present. Findings from studies conducted after larger spills generally apply to potential impacts of median-range spills. Although direct exposure to oil can result in damage to archaeological sites and materials, exposure does not necessarily cause damage, and oil contamination may affect sites without interfering with the recovery of scientific information from them. The presence of oil may pose a health hazard to researchers, which could limit the safe recovery of scientific information from archaeological resources (D'Andrea and Reddy 2014). Oil may persist in archaeological sites for at least a decade after the spill (Reger et al. 2000). Removing oil contamination may be impossible without disturbing sites or deposits.

Studies of larger spills suggest that response activities are more likely to damage archaeological resources than oil exposure itself. Heavy deposits of spilled oil may obscure the view of archaeological sites or materials during response activities, making sites difficult to protect during response activities (Haggarty et al. 1991). After the *Exxon Valdez* spill, response activities, including excavations, high-pressure water spraying, and the movement of workers, were the primary cause of damage to coastal sites (Jesperson and Griffin 1992). While there are no studies documenting the effects of median-range spill response activities on archaeological resources, they would likely be comparable. Materials used in response activities, such as sorbents and solidifiers, have not been documented as causing damage to archaeological resources.

19.1.4 Impacts of Oil Spills on Scientific Value of Historic Properties

Both oil exposure and response activities can affect the scientific value of archaeological resources and other historic properties (i.e., their capacity to provide information relevant to questions of archaeological, anthropological, and historic significance). In particular, oil contamination may affect analytical techniques commonly used to obtain chronological or compositional data from archaeological materials. Such techniques include radiocarbon dating, mass spectrometry, and neutron activation analysis. Superficial windborne oiling of the archaeological site UNL-92 after the *Kuroshima* spill did not appear to impede the recovery of scientific information (Knecht and Davis 1999). However, oiling may affect other types of sites, and other forms of oil mobilization may result in different impacts, as suggested by studies conducted in larger spill contexts. For example, as documented in BOEM's review of impacts associated with spills >20,000 bbl, oil residue in site sediments may increase the need for sample processing (e.g., solvent extraction) when performing radiocarbon dating to ensure the accuracy of results (Michel 2021). The literature consulted in that review also found that oil contamination may affect geochemical traces of past human activity in tidal and subtidal sediments (Michel 2021)

19.2 Intangible Cultural Heritage

Intangible cultural heritage encompasses a broad array of non-material cultural resources that constitute the cultural and spiritual environment, community expressions, cultural identities, and senses of place (Basso 1996; Feld and Basso 1996; UNESCO 2003). Examples include traditional knowledge and values, oral histories, language, dances and songs, toponymies, spiritual sites, and place-based beliefs, values, and practices.

Intangible cultural heritage in areas potentially impacted by oil spills may relate to geographies, histories, lifeways, and industries that are unique to coastal and marine environments. As such, determinations of oil spill impacts for intangible cultural heritage are highly context-dependent. Appropriate metrics may differ based on the affected groups or cultural communities and the relevant cultural components (Tandon 2018). Procedures for assessing spill impacts may include:

a) Identifying the affected groups and/or cultural communities;
b) Identifying the specific forms of affected intangible cultural heritage;
c) Describing the context of affected heritage, including resources required, practitioners, seasonality/scheduling, and specific places or areas needed to access or participate in the affected heritage; and
d) Effects on health and wellbeing of practitioners, availability of materials and other resources, access to necessary sites or areas, and transmission of knowledge or skills.

Impact intensity may be measured by the extent to which oil spill impacts affect the ability of individuals or groups to access, practice, or participate in intangible cultural heritage. The duration of these impacts may influence the recoverability, deterioration, or loss of intangible heritage. In addition, the magnitude of spill impacts may be influenced by the location and size of the spill; the type, timing, and effectiveness of response actions; and environmental conditions that affect oil transport and persistence. Because intangible cultural heritage may be linked to specific individuals, places, or small communities, the localized impacts of median-range oil spills may have major impacts on intangible cultural heritage.

Cultural and social resources are not analogous to natural resources, and the concept of natural recovery does not apply to intangible cultural heritage. Knowledge, languages, and songs, among many other forms of heritage, may be unrecoverable once lost. As such, time to recovery for intangible cultural heritage is not measured or estimated in the literature.

The literature does not include any studies that assess the impacts of individual median-range spills on intangible cultural heritage. Michel (2021), however, includes a review of such impacts as they relate to larger spills (>20,000 bbl).

19.3 Summary and Information Needs for Assessing Impacts to Cultural Resources

No studies on the impacts to cultural resources resulting from spills of 500–20,000 bbl of crude oil, condensate, or diesel were identified. Research on the impacts to cultural resources from oil spills has been conducted almost exclusively in response to larger spills, such as the *Exxon Valdez* and *Deepwater Horizon* spills. While findings from studies of these large spills are broadly applicable to median-range spills, equivalent research on median-range spill impacts on various forms of cultural resources has not been conducted or made publicly available. Studies of impacts on historic properties have examined the effectiveness and risks of various conservation techniques, given the unique requirements of removing oil from structures and materials in coastal and marine environments. Focusing on coastal archaeological resources, while research has generally indicated that direct impacts from oil exposure have been fairly limited, these impacts may affect approaches for analyzing these resources and necessitate special processing and/or handling of materials. In general, indirect impacts from response activities for the *Exxon Valdez* and *Deepwater Horizon* spills posed a more substantial potential threat to archaeological resources than oil exposure but were also fairly limited due to rapid-response efforts to document and protect archaeological sites.

Understanding the impacts of median-range spills crude oil, condensate, or diesel on cultural resources is constrained by the absence of studies conducted specifically in response to such spills. Because spill impacts are highly site-dependent, the small number of existing case studies are unlikely to fully account for or anticipate impacts for spills in other contexts. For historic properties, including coastal archaeological resources, a lack of research into long-term impacts means that little is known about the lasting effects of oil exposure or response activities. For intangible cultural heritage, a lack of research in diverse contexts makes drawing broader conclusions difficult. Findings from social scientific research on other types of small-scale or localized environmental and technological disasters may be applicable to median-range oil spills. For example, a study of impacts to Diné (Navajo) cultural practices resulting from the 2015 Gold King Mine spill found a significant reduction in the frequency and duration of cultural practices (Van Horne et al. 2021). Such research may suggest areas of focus for future efforts to assess oil spill impacts on cultural heritage.

20 Marine Archaeological Resources

This chapter focuses on the effects of spills of 500 to 20,000 bbl of crude oil, condensate, or diesel on marine archaeological resources, including submerged archaeological and historic resources and historic shipwrecks and aviation wrecks. Marine archaeological resources are defined as any material remains of human life or activities in the marine environment that are at least 50 years of age and can provide scientific or humanistic understanding of past human behavior through scientific or scholarly study (30 CFR 550.105). The management and protection of marine archaeological resources fall within the scope of numerous federal and state agencies, laws, and regulations, including but not limited to the NHPA, the OCS Lands Act Amendments of 1978, the National Marine Sanctuaries Act, Office of National Marine Sanctuaries regulations (15 CFR 922), the American Antiquities Act of 1906, and NAGPRA. The NPS Archeology Program maintains a compilation of submerged cultural resources laws for the States, the District of Columbia, and U.S. territories.[4]

20.1 Submerged Archaeological and Historic Resources

Submerged archaeological resources are defined as archaeological resources, not including historic shipwrecks and aviation wrecks (see Section 20.2), located beneath or substantially beneath tidal waters at mean low tide. Archaeologists and specialists in adjacent fields examine submerged sites, structures, landscapes, and artifacts to understand past human behavior, cultural adaptation, and related topics. Significant submerged archaeological resources are those that meet the criteria of significance and integrity for eligibility on the National Register, as defined in 36 CFR 60.4. The presence of "submerged maritime heritage resources of special historical, cultural, or archaeological significance" is among the criteria considered by NOAA in establishing national marine sanctuaries (15 CFR 922.10). This section discusses oil spill impacts on submerged archaeological and historic resources, including sites and structures that were historically above tidal waters but are now submerged, such as lighthouses, piers, harbors, and military installations. Impacts on terrestrial archaeological resources along the coastline, including those located above or substantially above the mean low tide line, are discussed separately (see Chapter 19 – Cultural Resources).

Submerged archaeological and historic resources specifically provide information about maritime lifeways, human adaptation to coastal and marine environments, and seafaring. Because of relative sea level rise in many coastal regions of North America since the end of the Pleistocene (ca. 11,700 years ago), numerous low-elevation coastal sites older than ca. 4,500 years are now inundated (Faught 2004). Expanses of the continental shelf along both the Atlantic and Pacific coasts of North America are paleolandscapes exposed during the period between the Last Glacial Maximum (ca. 21,500 cal B.P.) and the Clovis period (ca. 13,000–12,750 cal B.P.).[5] In the U.S. Southeast, these ancient landscapes consisted of an "open parkland, savannah-type environment" suited to Paleoindian subsistence strategies and technologies (Garrison and Thulman 2019). Consequently, many Paleoindian sites may be located on submerged ancient shorelines (Dunbar and Thulman 2019; **Figures 20-1** to **20-3**). More recent sites may also be submerged due to changing geographic and hydrographic conditions. An example is Dog Island Lighthouse (1839-1873), located south of Dog Island, Florida, which is now submerged due to landward migration of the island (Meide et al. 2001).

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[4] This compilation is available at https://www.nps.gov/archeology/SITES/stateSubmerged/index.htm.
[5] Radiocarbon dating measurements converted to calendar ages are reported as "cal B.P.," indicating calibrated years before present. By convention, "present" is assumed to be 1950.

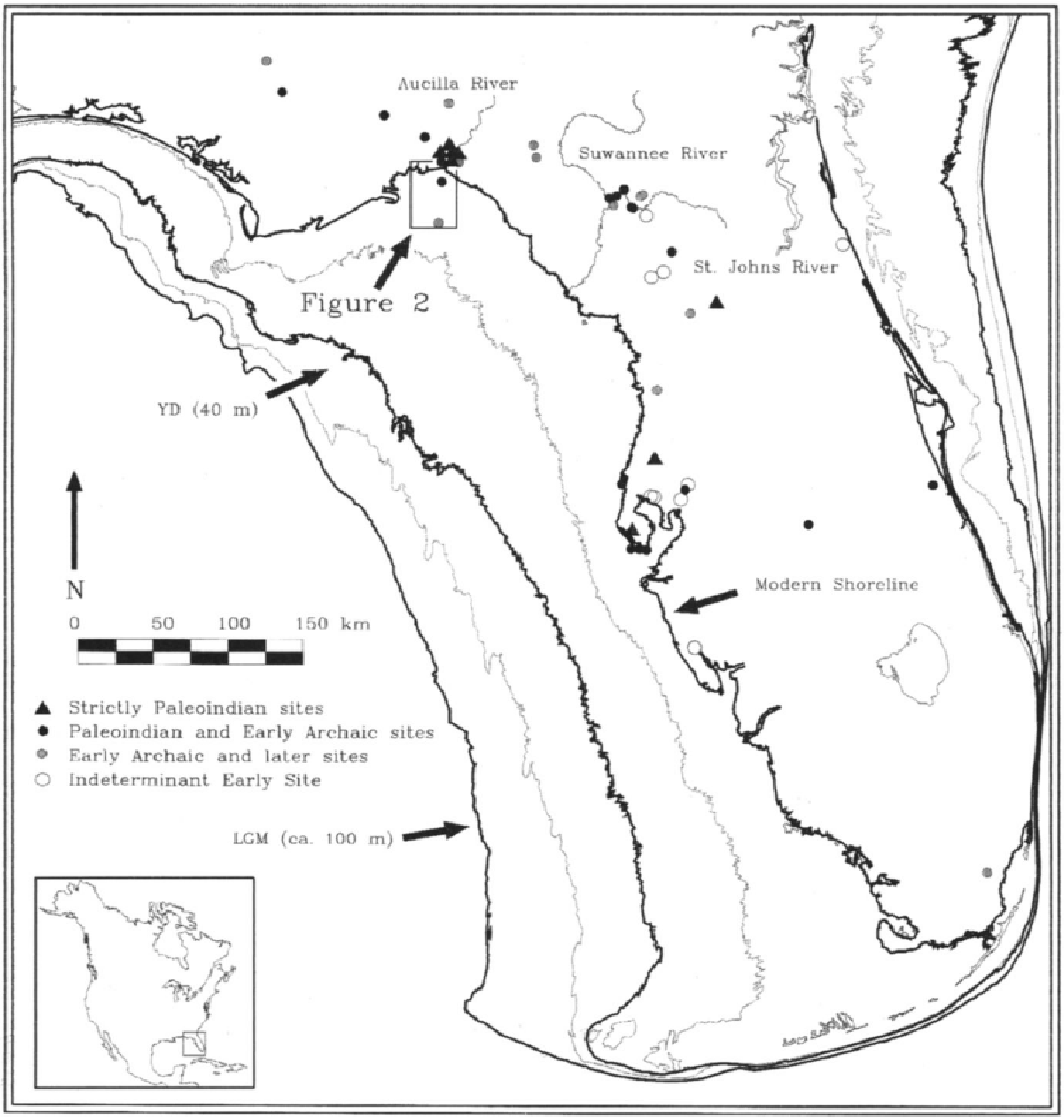

Figure 20-1. Submerged coastlines of the Florida peninsula
From Faught (2004).

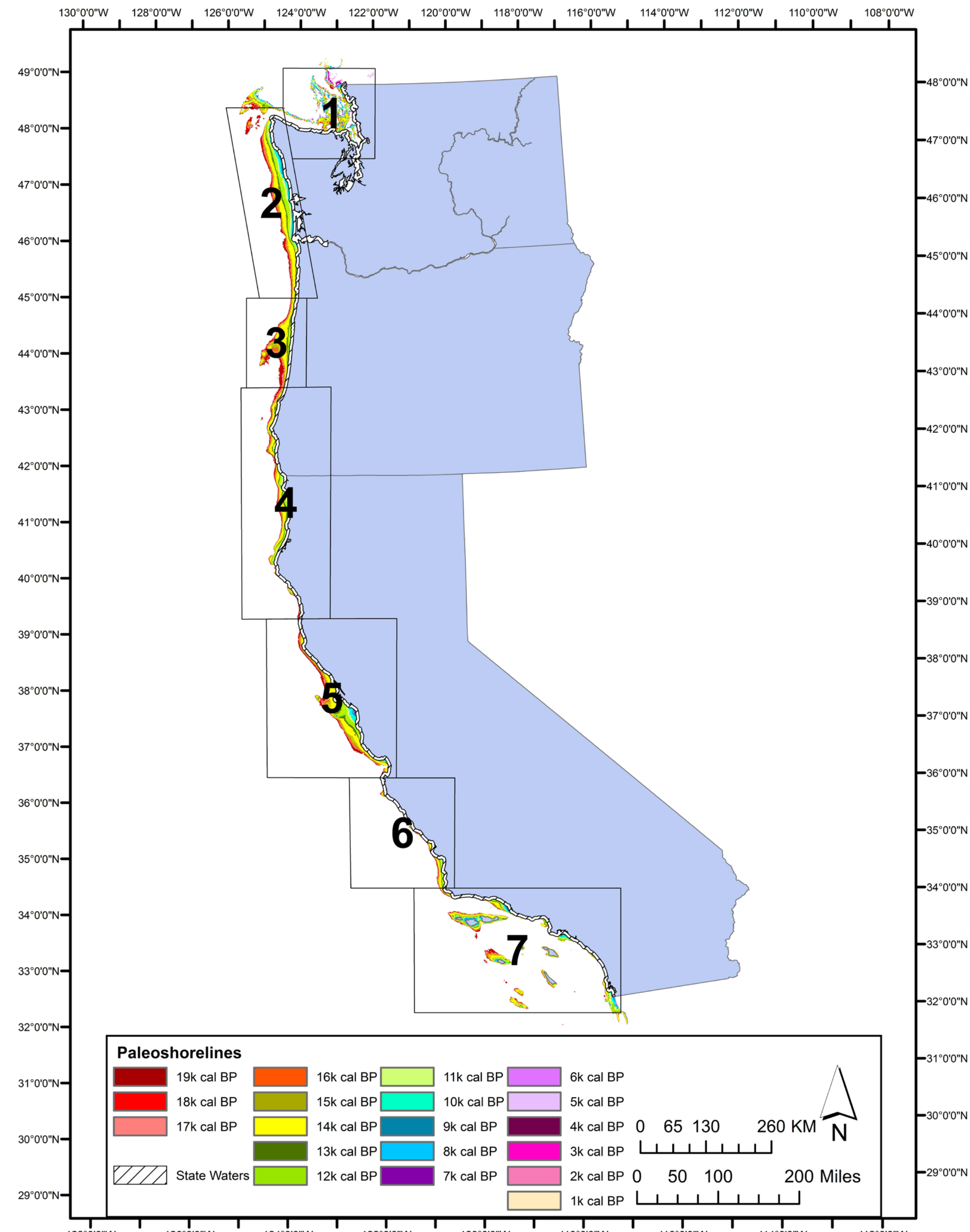

Figure 20-2. Paleoshorelines of the Pacific OCS and subdivided areas used to model potential archaeological site locations
From ICF International et al. (2013).

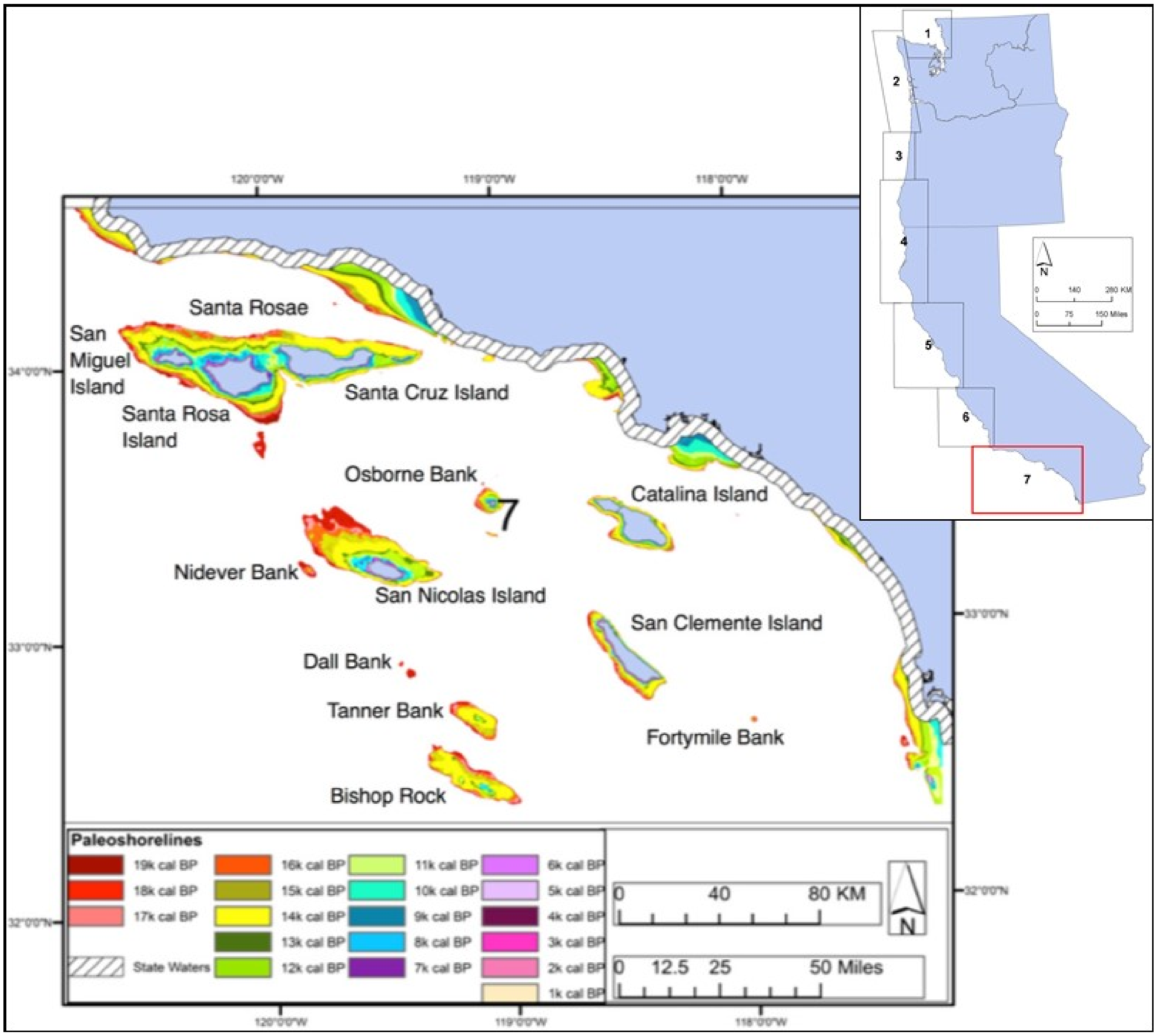

Figure 20-3. Shoreline contours of the exposed Pacific OCS landscape during the last glacial maximum at 19,000 years before present and each subsequent millennium, Subdivision 7
From ICF International et al. (2013).

Submerged archaeological resources have additional scientific value because terrestrial sites from the late Pleistocene are relatively rare in North America. They provide a critical record of some of the earliest human populations in North America, long-term human adaptations to changing climatic and environmental conditions, and the cultural and economic responses to shoreline movement.

Impact determinations for submerged archaeological and historic resources are based on the criteria for National Register eligibility. Potential effects resulting from oil spills include effects on:

- The "physical features" of the resource "necessary to convey the aspect of prehistory or history with which it is associated" (NPS 1997);
- The information potential of the resource (e.g., the intactness of key data contained in a site or object) as it relates to questions of anthropological, archaeological, and historical interest; and
- The integrity of the resource, including the preservation of physical elements; the spatial association of structures, artifacts, and deposits; and the chemical composition of physical materials and the surrounding environment.

20.1.1 Submerged Archaeological and Historic Resource Types, Descriptions, and Inventories

Submerged archaeological and historic sites include settlements; campsites; manufacturing workshops; sites associated with hunting, gathering, fishing, and other subsistence activities; burial places; locations of ceremonial activities; military installations; industrial facilities; and sites and structures associated with maritime navigation and other activities. Artifacts associated with these sites include inorganic materials such as lithic (stone), bone, and metal tools; organic materials such as wood and plant fibers; and dietary remains such as shellfish, bones, and seeds. Submerged sites may also include human remains, which may be subject to NAGPRA.

Records of and information about submerged archaeological and historic resources may be found in various databases and inventories. These include site inventories maintained by SHPOs, which inventory historical cultural resources generally without regard to historical significance criteria of the National Register. To protect sensitive resources, full public access to these databases is typically restricted. Other inventories of archaeological resources include the NADB maintained by the Center for Digital Antiquity at Arizona State University, and DINAA, a collaborative effort of several universities and research institutions. In addition, reports and surveys contain information about submerged archaeological resources at varying scales, including BOEM's coastal inventories and modeling (ICF International et al. 2013; **Figures 20-2** and **20-3**; TRC Environmental Corporation 2012). For submerged archaeological resources other than historic shipwrecks and aviation wrecks, these studies model the potential for site occurrence on the OCS rather than provide inventories of known sites. Specific sites may be identified in the results of remote-sensing surveys conducted by the oil and gas industry for NHPA Section 106 compliance, in areas where such surveys are required (MMS 2005).

20.1.2 Oil Interaction with Components of Submerged Archaeological and Historic Resources

Because the National Register eligibility criteria are primarily qualitative, specific metrics for spill-related impacts are not available. The magnitude of spill impacts may be influenced by the location, scale, and nature of the resource; its physical composition and condition; the nature of the resource's setting or context; the location and size of the spill; the type and effectiveness of response actions; environmental conditions that affect oil transport; and conditions that affect assessment and response of archaeologists, conservators, and other management professionals.

Archaeological and historic resources are inherently irreplaceable. Unlike biological resources, there is no natural recovery process. Resources damaged in ways that affect their surface, appearance, or chemical composition may be subject to some degree of restoration through the application of appropriate in-situ or laboratory-based conservation techniques. When damage involves changes to the spatial and physical association of artifacts, sediments, and other resource components, restoration may not be possible. As such, time to recovery for submerged archaeological and historic resources has not been measured or estimated in the literature.

With the exception of shipwrecks and associated biota, which are discussed below, oil interaction with submerged archaeological and historic resources is poorly understood. Archaeological investigations after larger spills such as the *Exxon Valdez* and *Deepwater Horizon* spills have examined terrestrial sites (Haggarty et al. 1991; Mobley et al. 1990; Rees et al. 2019), but studies related to median-range spills have not been conducted or made publicly available. Some types of impacts on submerged sites would be expected to be similar to those on resources in the coastal and/or intertidal zone (see Chapter 19 –

Cultural Resources). For example, oil contamination of submerged artifacts and sediments may affect analytical techniques, such as pottery residue analysis, in similar ways to terrestrial artifacts.

Oil spill impacts on submerged archaeological and historic resources may occur primarily as a consequence of the effects of oil and dispersant exposure on microorganisms that influence the preservation of submerged sites, structures, and artifacts. Concretions known as rusticles, which consist of bacteria, fungi, and mineral compounds such as iron oxides, may help preserve archaeological materials such as coal and glass (Cullimore and Johnston 2008). The microorganisms that constitute these formations are sensitive to environmental disturbances such as those caused by the introduction of oil or dispersants. Findings from recent research focused on shipwrecks (see Section 20.2 – Historic Shipwrecks and Aviation Wrecks) may improve understanding of how oil and dispersant interact with microorganisms associated with other types of submerged archaeological resources.

20.1.3 Impacts of Oil Spills on Preservation of Submerged Archaeological and Historic Resources

Information is not readily available to assess the effects of oil exposure or response activities on the preservation of submerged archaeological and historic resources, except for shipwrecks (see Section 20.2). To the extent that marine biota interact with submerged archaeological resources, the effects of oil exposure on biota may have consequences for archaeological resources. For example, microbial communities that form on archaeological materials may play a role in inhibiting or accelerating deterioration. Anchoring associated with ship deployment of booms and other spill-containment measures may cause physical damage. Physical disturbance or damage from response activities (e.g., excavation or staging) is presumed to be less likely for submerged resources than for terrestrial resources.

20.1.4 Impacts of Oil Spills on Scientific Value of Submerged Archaeological and Historic Resources

Little research has examined the effects of oil exposure or response activities on the scientific value of submerged archaeological and historic resources or on their suitability for analytical techniques. Some of the effects may be comparable to those documented for coastal/intertidal resources (see Chapter 19 – Cultural Resources).

20.2 Historic Shipwrecks and Aviation Wrecks

There are an estimated 20,000 shipwrecks and aviation wrecks in U.S. waters, including thousands of known wrecks on the OCS (Garrison et al. 1989; ICF International et al. 2013; NOAA 2013a; Pearson et al. 2003; TRC Environmental Corporation 2012). Historic shipwrecks and aviation wrecks are defined as those listed in or eligible for listing in the National Register. Per the National Archives and Records Administration database, which includes records through 2012, 169 shipwrecks are listed in the National Register; numerous other shipwrecks are eligible but have not yet been listed.[6] In the Gulf of Mexico, 12 historic shipwrecks have been listed and approximately 30 are considered eligible but have not yet been listed. Historic shipwrecks in the Gulf of Mexico have also been the subject of recent archaeological studies aimed at improving understanding of maritime history, site preservation, and the ecological effects of anthropogenic structures and objects in marine environments (Brooks et al. 2016; Damour et al. 2015;

[6] See https://www.nps.gov/subjects/nationalregister/upload/national_register_listed_20200108.xlsx

Church and Warren 2008). Numerous historic shipwrecks along the Atlantic, Pacific, and Alaskan coasts have also been listed or are considered eligible for listing in the National Register.

Shipwrecks and aviation wrecks often have both historical and ecological significance. A large proportion of shipwrecks along the Atlantic and Gulf coasts date to World War II and now serve as memorials and grave sites, as do shipwrecks from other time periods. Many of these shipwrecks, which include passenger and cargo vessels, oil tankers, and fishing boats, were chartered by the U.S. government for the war efforts and carried U.S. Navy personnel (Church et al. 2007). Due to their potential historical value, shipwrecks are among the submerged maritime heritage resources that contribute to meeting the national significance criteria for areas nominated as national marine sanctuaries. The Monitor National Marine Sanctuary, for example, preserves the remains of naval and merchant vessels from the Civil War period through World War II.[7] Many other national marine sanctuaries also contain historic shipwrecks. In addition to having historical and cultural significance, shipwrecks form artificial reefs that create habitat for a wide range of species and contribute to marine biodiversity (Brooks et al. 2016;Church et al. 2007; Church and Warren 2008; Paxton et al. 2019).

Impact determinations for historic shipwrecks and aviation wrecks are based on National Register eligibility criteria (NPS 1992). Potential effects include those on:

- The information potential of the vessel or aircraft (i.e., the capacity of its physical characteristics to provide information important to history); and
- The integrity of the vessel or aircraft, including the preservation of physical elements; spatial association of components, related objects or artifacts, and other remains; and the chemical composition of physical materials and the surrounding environment.

20.2.1 Historic Shipwrecks and Aviation Wrecks Types, Descriptions, and Inventories

Shipwrecks and aviation wrecks encompass a wide range of types and time periods. Historic shipwrecks date from the period of European colonization to the 20th century; known historic aviation wrecks postdate the Wright brothers' first successful flights in 1903 (NPS 1998). Records of and information about historic shipwrecks can be located in a variety of sources. For the OCS, the most comprehensive inventories are those compiled by BOEM (BOEM 2011; Evans et al. 2013; Garrison et al. 1989; ICF International et al. 2013; Pearson et al. 2003; Tornfelt and Burwell 1992; TRC Environmental Corporation 2012). These studies integrate records from numerous databases, inventories, and historical sources. To protect sensitive resources, the resulting databases are not publicly available, but they include 10,519 vessel records for the Atlantic OCS; 5,813 records for the Pacific OCS; and 2,106 records for the Gulf of Mexico OCS (**Figure 20-4**). There are also thousands of shipwrecks in waters off the coast of Alaska (BOEM 2011; Tornfelt and Burwell 1992). These statistics include records of vessels that do not necessarily have historical significance.

Other inventories include the NOAA Office of Coast Survey Automated Wrecks and Obstruction Information System (AWOIS), [8] which contains spatial and descriptive data for over 10,000 submerged wreck and obstructions in U.S. coastal waters; and the NOAA Resources and Undersea Threats (RUST) database, which includes approximately 20,000 shipwrecks located in U.S. waters (NOAA 2013b; **Figure 20-5)**.

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[7] See https://monitor.noaa.gov/shipwrecks/.

[8] See https://www.nauticalcharts.noaa.gov/data/wrecks-and-obstructions.html. Note, however, that the AWOIS database has not been updated since 2016.

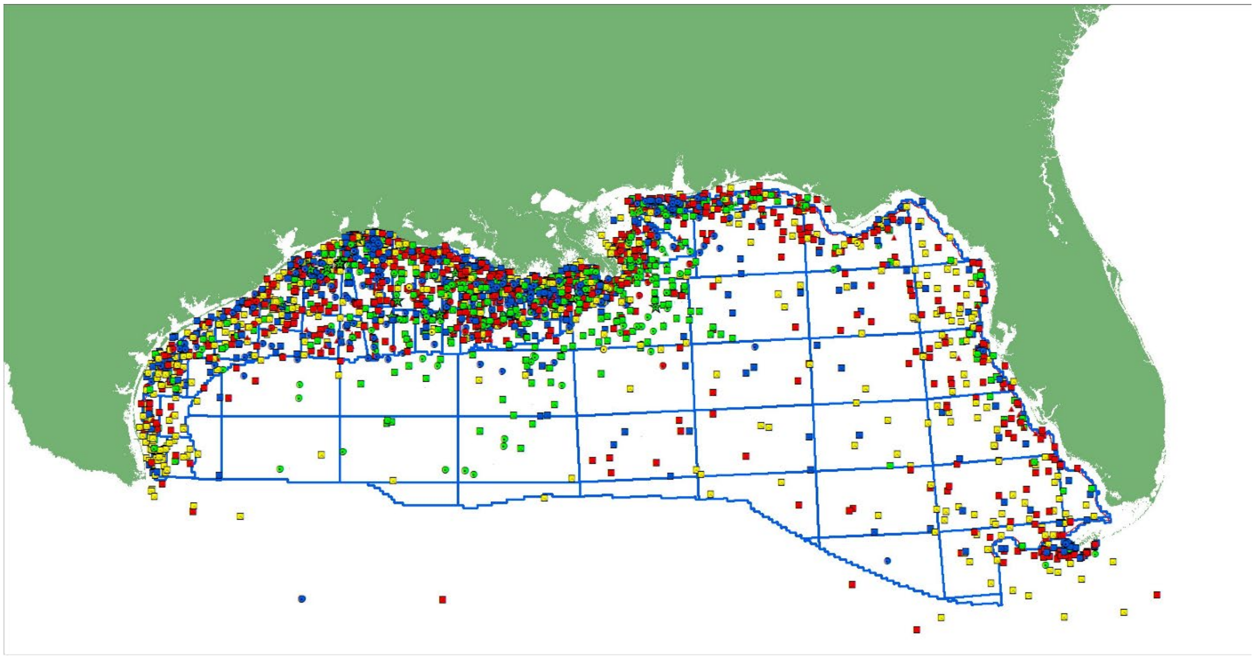

Figure 20-4. Positions of all entries classified as “Vessels” in the 2019 BOEM shipwreck database for the Gulf of Mexico OCS Region
Marker colors relate to accuracy of information about shipwreck positions.

20.2.2 Oil Interaction with Submerged Shipwrecks and Aviation Wrecks

Because National Register eligibility criteria are primarily qualitative, specific metrics for impacts to historic shipwrecks and aviation wrecks are not available. The magnitude of spill impacts may be influenced by the location, scale, and nature of the vessel or aircraft; physical composition and condition of the vessel/aircraft and associated remains; nature of its setting or context; location and size of the spill; type and effectiveness of response actions; environmental conditions that affect oil transport; and conditions that affect assessment and response of archaeologists, conservationists, and other management professionals.

Shipwrecks and aviation wrecks, as with other cultural resources, are inherently irreplaceable. Unlike biological resources, there is no natural recovery process. Resources damaged in ways that affect the surface, appearance, or chemical composition may be subject to some degree of restoration through the application of appropriate in-situ or laboratory-based conservation techniques. When damage involve changes to the state of preservation, or to the spatial and physical association of artifacts, sediments, and other components, restoration may not be possible. Because of these characteristics, time to recovery for shipwrecks and aviation wrecks has not been measured or estimated in the literature.

Few studies of the impacts of oil spills, dispersants, and other response activities on shipwrecks and aviation wrecks have been conducted (Damour et al. 2015). The limited number of studies available assessed the impacts of large oil spills, in particular, the *Deepwater Horizon* spill. This literature is summarized in Michel (2021). Impacts for median-range spills would likely be comparable to impacts for large spills, though no information is readily available to test this hypothesis.

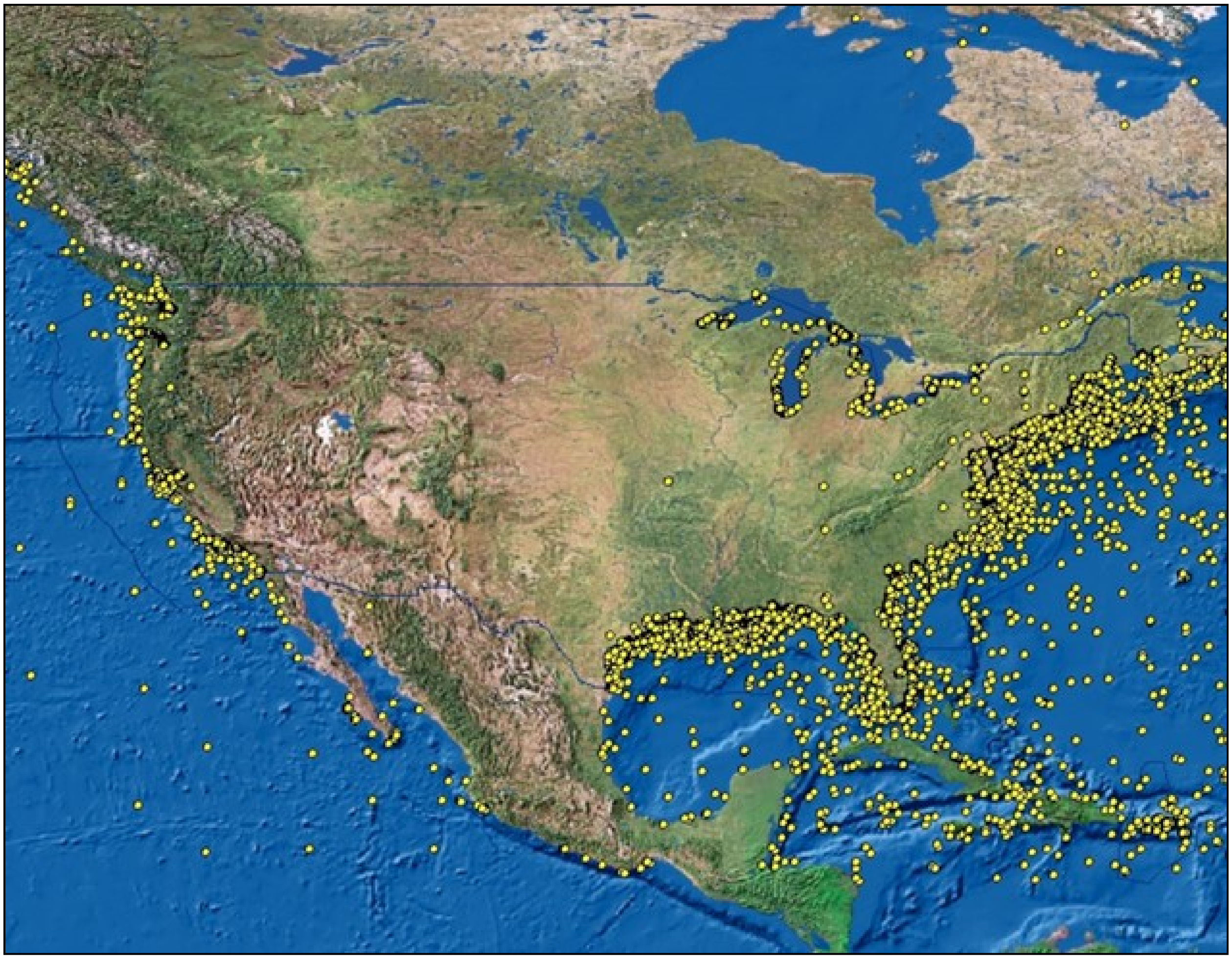

Figure 20-5. The NOAA Resources and Undersea Threats database includes 20,000 submerged vessels
From NOAA (2013a).[9]

Oil interaction with shipwrecks can result from both natural and accidental oil discharge from the seafloor mixing with water column particles. Other potential pathways of shipwreck exposure to oil include droplets formed by dispersants, sinking of oil residues from controlled burning of oil on the water surface, and transportation of oil-dispersant mixtures by deepwater currents (Damour et al. 2015; Passow et al. 2012). Research indicates that oil spill impacts on shipwrecks may be largely influenced by interaction between oil and microbial communities that form in association with submerged vessels. A wide variety of microorganisms can contribute to both degradation and preservation of shipwreck sites, depending on a number of factors including the material composition of the vessel and oceanographic conditions such as oxygen availability, temperature, salinity, and pH levels. The composition, structure, and function of microbial communities can vary among shipwrecks as well as within individual shipwreck sites. The microorganisms that make up these communities are highly sensitive to environmental disturbances,

[9] This figure, as originally published by NOAA, represents only the contiguous U.S.; spatial information for wrecks in waters off the coast of Alaska has not been published.

including the introduction of hydrocarbons or chemical dispersants resulting from oil spills (Damour et al. 2015; Hamdan et al. 2018). Reductions in microbial community diversity after initial exposure to oil and dispersants have been shown to persist for several years (Hamdan et al. 2018).

Recent studies focused on large spill impacts, particularly the *Deepwater Horizon* spill, have focused on how oil and dispersants affect the microbial communities known as biofilms that form on shipwrecks. As documented in Michel (2021), this literature suggests, based on shipwreck oiling observed following the *Deepwater Horizon* spill, that oil spill effects may interfere with the ability of researchers to study the ecological characteristics of deepwater shipwrecks. Depending on variables such as conditions at the time of the spill and the location of a shipwreck, the same may also be true for median-range spills.

In addition to sustaining a wide range of microorganisms, shipwrecks have an artificial reef effect on marine ecosystems that supports numerous macrofaunal invertebrate and vertebrate species (Church et al. 2009). A survey of seven World War II-era shallow to deepwater shipwreck sites in the Gulf of Mexico identified at least 79 invertebrate taxa and 105 vertebrate taxa within and around the sites. Species diversity and abundance were generally higher near the shipwrecks compared to farther away, although this effect was not seen for fish species at sites in deeper waters (Church and Warren 2008). Oil and dispersant exposure may affect the macrofaunal communities surrounding shipwrecks to the extent that they are sensitive to such environmental disturbances.

20.2.3 Impacts of Oil Spills on Preservation of Submerged Shipwrecks and Aviation Wrecks

Wooden-hulled shipwrecks in marine environments are susceptible to both physical and biological deterioration. Various biological agents, including shipworms, fungi, and bacteria, attack wood exposed above the seabed. Wood buried in sediment is partially protected from some forms of biological degradation but may still be subject to long-term decay caused by so-called “erosion” bacteria (Björdal 2012; Björdal and Nilsson 2008; Gregory et al. 2012; Hunt 2012). Physical disturbance of the seabed associated with response activities would have the potential to adversely affect wood shipwreck preservation by exposing previously buried wood components to biological agents.

Research on oil spill impacts has also focused on steel shipwrecks, particularly to determine whether injuries to biofilms caused by oil exposure contribute to the corrosion of submerged steel. The marine biofilms that contribute to increased biodiversity around shipwrecks and other submerged steel structures also play a role in slowing steel corrosion (Mugge et al. 2019). Oil residues may affect biofilm communities in ways that reduce their protective capacity, though no research on this effect has focused on any median-range spills. As documented in the review of spills >20,000 bbl in Michel (2021), studies focused on the *Deepwater Horizon* spill found metal loss in steel shipwrecks exposed to oil from the spill. Following the *Deepwater Horizon* spill, metal corrosion at the site closest to the spill origin accelerated due to its effects on marine biofilms (Mugge et al. 2019). Similar metal loss could potentially occur from exposure to oil released during median-range spills.

20.3 Summary and Information Needs for Assessing Impacts to Marine Archaeological Resources

There has been little research on the impacts of oil spills on marine archaeological resources, with a particular lack of research on impacts for median-range spills. Findings from studies focused on impacts from larger spills, however, may be transferable to median-range spills. This research has shown that microbiota associated with submerged shipwrecks that are exposed to oil are less diverse than those associated with shipwrecks that were not exposed. Oil spill impacts on marine biofilms may interfere with their ability to slow corrosion of steel shipwrecks and impacts on marine microbiota may affect the preservation of wooden-hulled shipwrecks in ways that are not yet understood.

Oil spill impacts on a broad array of archaeological resources, including sites and artifacts, located on submerged coastlines are largely unknown. Potential impacts on marine archaeological resources may be partly inferred from current understanding of impacts on coastal and intertidal resources, but the similarity of impacts across these resources has yet to be empirically demonstrated. As methodological and technological advances in underwater archaeology continue, submerged archaeological resources are likely to become increasingly important for the scientific study of the human past, particularly for questions about the earliest human populations in North America. Studies have not shown how impacts to these resources may vary according to type or quantity of oil, duration of oil exposure, or interaction between oil and resource-associated microbiota. Documentation is also very limited to show how spill response efforts, including mechanical disturbance and the use of dispersants, may affect the integrity and/or information potential of submerged archaeological resources. Because submerged archaeological resources include diverse organic and inorganic materials, impacts may differ considerably depending on the nature of the resource. Understanding oil spill impacts on submerged human remains is uniquely important given their significance to descendant communities, especially Native peoples; however, the rarity of locating submerged sites where human remains are present suggests that studies relevant to this question are unlikely to occur in the near future.

Key questions about potential oil spill impacts and research or information needed to address those questions are summarized in **Table 20-1**.

Table 20-1. Key questions and information needs for assessing oil spill impacts to marine archaeological resources

Question	Research and information needs
How do differences in oil behavior and persistence between large and median-range spill incidents affect impacts on marine archaeological resources?	Field and laboratory studies of oil interaction with submerged archaeological resources in median-range spill scenarios.
How do submerged archaeological sites, artifacts, and landscapes other than shipwrecks interact with released oil, dispersants, and other spill-related substances?	Field and laboratory studies of submerged archaeological sites and materials in areas impacted by oil spills and response activities.
What are the long-term impacts of oil spills and response activities on submerged archaeological resources, including effects on preservation and information potential?	Long-term studies of sites and artifacts impacted by oil spills and response activities.
How do hydrocarbons, dispersants, and other spill-related substances interact with specific archaeological materials in marine environments?	Actual and/or experimental analysis of chemical interaction and effects on commonly applied analytical techniques.
How do marine biota potentially affected by oil spills interact with organic and inorganic archaeological materials?	Actual and/or experimental analysis of interaction and effects on preservation and information potential of archaeological materials.

21 Vulnerable Coastal Communities

21.1 Resource Description

This chapter addresses the impacts of median-range spills (500–20,000 bbl) of crude oil, condensate, or diesel on vulnerable coastal communities. Vulnerable coastal communities include low-income, minority, and Indigenous communities that reside near the coast and are especially susceptible to the environmental, health, cultural, and economic impacts of oil spills.

Vulnerable coastal communities can be defined by different characteristics. **Table 21-1** presents two metrics of vulnerable coastal communities: the percent of the population that is minority (all races/ ethnicities other than white, non-Hispanic), and the percent of households living below the poverty line. Both metrics are often relied upon when considering Environmental Justice in BOEM NEPA documents.

Table 21-1. Socioeconomic data for coastal counties by state and OCS Planning Area
Bolded values represent coastal areas with a greater percentage than the U.S. average values. Minority is defined here as the total population, minus the total white, Non-Hispanic population. State results include the sum of all coastal counties. From 2015-2018 ACS 5-year estimates, U.S. Census Bureau.

OCS Planning Area	State	% Minority	% of Households Below the Poverty Threshold	Total Coastal County Population
Central Gulf of Mexico (GOM)	Alabama	34	**16**	622,766
	Mississippi	32	**18**	391,293
	Louisiana	**44**	**18**	1,548,069
Multiple	Alaska	**40**	10	624,340
Northern California, Central California, Southern California	California	**64**	12	26,631,231
North Atlantic	Connecticut	34	10	2,235,936
	Maine	7	11	728,866
	Massachusetts	29	12	3,584,183
	New Hampshire	7	5	305,129
	New Jersey	**49**	12	4,636,256
	New York	**59**	**15**	12,256,993
	Rhode Island	27	**14**	1,056,611
Mid-Atlantic	Maryland	**42**	11	3,041,740
	Delaware	37	11	949,495
	Virginia	**47**	9	3,324,792
	North Carolina	27	**15**	1,016,449
South Atlantic	South Carolina	32	**14**	1,025,354
	Georgia	**46**	**15**	535,850
South Atlantic, Straits of Florida, Eastern GOM	Florida	**46**	13	15,515,312
Western GOM	Texas	**68**	**15**	6,679,508
Oregon/ Washington	Oregon	16	**16**	675,539
	Washington	33	10	4,776,427
None	Hawaii	**78**	10	1,422,029

Relative to the total percent of U.S. households below the poverty level (14%), 10 states' combined coastal counties eclipse that mark. Relative to the 40% of the entire U.S. population characterized as a racial or ethnic minority, 11 total states have a coastal region with a greater prevalence of minorities, in proportional terms. Overall, 17 out of 23 total coastal states either have proportionately more minorities than the national average or a proportionately greater number of households living in poverty than the entire U.S. Further, **Table 21-1** shows that the areas with the highest percentage of households below the poverty threshold may be in the closest proximity to the highest concentration of drilling activity (the Central and Western Planning Areas of the Gulf of Mexico), with Texas, Louisiana, Alabama, and Mississippi each with a higher than average percentage of households living below the poverty line.

Vulnerable coastal communities often engage in culturally unique activities based on natural resources such as hunting and gathering wild foods, collecting medicinal plants, and managing the environment according to traditional knowledge and practices. Researchers have characterized such communities as *natural resource communities* or *renewable resource communities* (Gill 1994), defined as "population[s] of individuals who live within a bounded area and whose primary cultural, social, and economic existences are based on the harvest and use of renewable natural resources" (Picou and Gill 1996). In Alaska Native communities, as well as many other traditional communities, traditional natural resource uses include not only subsistence harvesting and consumption of wild foods but also extended social networks of reciprocity and exchange (Gill and Picou 1997; Ritchie and Gill 2008). Beyond subsistence, such communities rely on natural resources for medicine, craft and tool materials, and as components of settings for cultural and spiritual activities. Injuries to natural resources and diminished access to resources resulting from oil spills may thus be experienced by natural resource communities as cultural losses as well as economic losses (Kirsch 2001). Because many natural resource communities are also place-based communities, the more localized impacts of median-range spills may nonetheless have major impacts on communities that are closely tied to specific locales, habitats, and landscapes.

21.2 Impacts of Oil Spills on Vulnerable Coastal Communities

21.2.1 Characterization of Impacts

The impacts of oil spills on vulnerable communities include direct impacts related to a spill and indirect impacts related to the aftermath of the spill and the response. Direct impacts include the disruption of subsistence fisheries or other subsistence resources to native populations and the loss of jobs and/or income dependent on coastal resources. Indirect impacts include exposure to hazardous and other wastes associated with the spill response, health impacts to vulnerable populations (including psychological impacts), and differences in spill compensation. Indirect impacts may also arise from the stress that spill response activities place on local resources. Due to the limited food, fuel, housing, and other resources in many vulnerable coastal communities (particularly in Alaska), spill response activities and an influx of response workers may overwhelm these communities, causing disruption to community members' way of life (Coastal Response Research Center 2010). In addition, the remoteness of some vulnerable communities may exacerbate spill impacts for these communities. For example, if a spill were to occur in a remote area off the Alaskan coast far removed from critical assets (e.g., heavy lift helicopters and emergency salvage capacity), the spill response would likely be delayed, potentially worsening the impacts of both the spill itself and the spill response for vulnerable communities in the area (Coastal Response Research Center 2010).

Factors that influence the magnitude of spill impacts on vulnerable communities include the following:

- **Location**. The proximity of a spill in relation to minority and/or low-income communities will affect the severity of the spill's impacts to these communities.

- **Season**. Vulnerable communities may be more impacted by spills that interrupt seasonal employment opportunities (i.e., the summer season).
- **Spill duration**. A spill with greater duration, in terms of both the release of oil and response activities, may affect to what extent vulnerable communities are impacted.
- **Spill volume**. All else equal, a larger spill is likely to cause greater impacts to vulnerable communities than smaller spills.
- **Effectiveness of response actions**. The type of, effectiveness, and spatial distribution of response actions can either mitigate or exacerbate the impacts of spills on vulnerable communities.

In addition to these factors, impacts to traditional natural resource communities may be assessed in terms of their cultural, social, and economic significance, including the degree to which a spill:

- Inhibits cultural practices, including those related to spirituality, sacred sites, medicinal practices, and rituals or ceremonies;
- Affects community and institutional structures, political and social resources, kinship, health, and education. Examples include changes in reciprocity (sharing of resources within families or communities) or mobility associated with natural resource uses; and
- Results in economic impacts affecting subsistence, trade, and employment. Examples include increased dependence on non-traditional foods and other resources, or increased need to engage in the labor market.

21.2.2 Impacts of Oil Spills of 500–20,000 bbl of Crude Oil, Condensate, or Diesel

No studies on the impacts to vulnerable coastal communities resulting from spills of 500–20,000 bbl of crude oil, condensate, or diesel were identified. Social and economic measures of oil spill impacts for these communities, such as changes in community cohesion, the subsistence component of diets, labor market engagement, and psychological wellbeing, have been quantitatively assessed in the literature, though in studies generally focusing on spills >20,000 bbl (e.g., IAI et al. 1998 ;Palinkas et al. 1993). Several of the findings for larger spill events summarized in this literature, as described in Michel (2021), are likely to be applicable to median-range spills as well. For example, low-income communities are likely to be particularly vulnerable to swings in income related to median-range spills. The high prevalence of low-income populations near areas where OCS oil and gas activity is most common (as shown in **Table 21-1**) suggests that spill-related impacts in coastal regions will be more substantial for vulnerable populations. Similarly, the level of psychological and emotional stress associated with median-range spills is likely to be disproportionately high for vulnerable populations. Such effects include depression, elevated stress levels, and post-traumatic stress disorder. Communities with heightened vulnerability may also be especially susceptible to health risks posed by median-range oil spills, such as potential increased asthma risk due to the inhalation of volatile organic compounds related to oil, dispersants, and other chemicals involved in spill response.

Due to the lack of studies of median-range spills of crude, condensate, or diesel, two median-range heavy fuel oil spills in Alaska are discussed here because they generated qualitative information about impacts to traditional subsistence activities in vulnerable coastal communities. Both spills occurred near Unalaska Island, Alaska, an area that has been continuously inhabited by Alaska Native communities for over 9,000 years (IAI 2011). Coastal natural resources of importance to these communities include marine mammals (sea lions, harbor seals), wild fish (salmon, halibut), shellfish, crab, birds, and plants (Veltre and Veltre 1982; IAI 2011).

The 1997 M/V *Kuroshima* spill (928 bbl of heavy fuel oil) occurred in Summer Bay near Dutch Harbor on Unalaska Island, impacting an area where traditional subsistence resource activities occur. While only some community members participate directly in collecting subsistence resources, the resources are distributed through networks of sharing to nearly the entire community. Subsistence resources constitute a major component, in some cases as much as 50%, of the diet for Alaska Native households (Veltre and Veltre 1982). The environmental damage assessment for the *Kuroshima* spill found that oil exposure and response activities injured resources along the shoreline and beaches, including vegetation, shellfish and other intertidal biota, and salmonids (NOAA 2002). Analysis of shellfish samples indicated that PAH levels were below human health concern (Helton 2003). However, concerns about oil contamination among Native communities contributed to avoidance and reduced the cultural value of subsistence resources. Native residents who had relied on the injured resources for subsistence raised concerns that subsistence resources would be unwholesome and unsafe to consume. In addition, advisories against consumption of shellfish were issued, and access to the shoreline area impacted by the spill was closed or restricted for several months. This resulted in substantial lost use of shellfish resources for subsistence activities (NOAA 2002). The damage assessment determined that oil could be present in shellfish harvesting areas for at least 10–15 years after the spill, although oil would gradually diminish over time (NOAA 2002; Helton 2003).

The 2004 M/V *Selendang Ayu* spill (8,300 bbl of intermediate fuel oil and marine diesel), also near Unalaska Island, oiled land and shorelines that belong to a number of Alaska Native corporations and tribes. Impacts of the spill on natural resource activities by local communities are described extensively by IAI (2011). Following the spill, public access restrictions and response activities interfered with subsistence activities by the local population (IAI 2011; Kohout and Meade 2008). The spill raised concerns among Native communities that subsistence foods would be contaminated, particularly given memories of the *Kuroshima* and *Exxon Valdez* spill impacts and information about long-term oil persistence from those events (IAI 2011; Kohout and Meade 2008; Ritchie and Gill 2008). After the *Selendang Ayu* spill, testing results from shellfish species recognized as subsistence resources exceeded risk-based screening criteria for PAHs in only one composite mussel sample collected prior to shoreline treatment operations. Other samples from the most heavily impacted area showed elevated benzo(a)pyrene (BaP) equivalents compared to other areas, but levels for these samples did not exceed risk-based criteria (Mauseth et al. 2008). However, studies based on such criteria may not adequately address the concerns of Native communities, who may continue to avoid resources that they do not believe are safe (Arquette et al. 2002). Native residents interviewed after these resources were officially designated as safe to consume continued to express concerns about resources in the impacted area (IAI 2011). Interviews with Native residents also showed that reduced access to subsistence foods and a lack of alternative sources threatened to interrupt cultural continuity, especially as younger community members were deprived of the opportunity to experience eating such foods (Ritchie and Gill 2008). The interruption of natural resource activities can produce intense psychological and social distress for members of affected communities, as well as economic losses. The Qawalangin Tribe of Unalaska calculated total cultural, recreational, and economic losses of slightly over $1 million, although the claim was rejected by the National Pollution Fund Center on technical grounds (IAI 2011, p. 61).

As highlighted by the above discussion regarding the M/V *Kuroshima* and M/V *Selendang Ayu* spills, assessment of cultural impact measures related to traditional natural resource activities, such as changes in perceptions of the environment, spiritual practices, and traditional knowledge, typically requires qualitative approaches (e.g., Dyer et al. 1992; Ritchie and Gill 2008). Impacts on natural resource activities are often difficult to assess due to the confidentiality or sensitivity of information related to natural resource collection and traditional knowledge systems. In addition, for Native communities, conventional risk assessment methodologies may not fully account for unique exposure pathways, variations in susceptibility of groups such as elders and children, and cultural values. Studies based on such methodologies may be insufficient to allow Native communities to make well-informed decisions about using or consuming natural resources in spill-impacted environments (Arquette et al. 2002). The magnitude of spill impacts on natural resource activities may be influenced by the location, size, and nature of the affected community; affected cultural components; specific natural resources of importance; the location and size of the spill; the type and effectiveness of response actions; and environmental conditions that affect oil transport and persistence.

21.3 Summary and Information Needs for Assessing Impacts to Vulnerable Coastal Communities

The impacts of median-range oil spills to vulnerable coastal communities have been documented only for a limited number of heavy fuel oil spills. Although the impacts of any future median-range spill on vulnerable communities would be site-specific, additional research on this topic would provide policymakers and the public with valuable insights that could inform BOEM planning decisions. Such research could not only examine vulnerable community impacts in broad terms but could also address several more targeted research questions. For example, in addition to estimating the magnitude of impacts to vulnerable communities immediately after spills occur, future research could quantify longer-term impacts. Future studies of median-range spills could also examine the conditions under which spills may result in more (or less) severe impacts for vulnerable communities, for example whether such impacts are more severe in local economies dependent on commercial fishing or those dependent on tourism.

22 Recreation and Tourism

22.1 Resource Description

This chapter focuses on the effects of spills of 500–20,000 bbl of crude oil, condensate, or diesel on recreation and tourism activities. Coastal and marine resources provide opportunities for a variety of recreational uses, including beach use, boating, wildlife viewing, swimming, and diving. These coastal and marine resources provide value to residents of coastal areas as well as tourists that visit these areas in part to enjoy recreational opportunities.

Beach visitation is the dominant recreation activity in many coastal regions. **Table 22-1** presents estimates of annual beach visitation by state developed for BOEM's OECM based on a variety of state- and region-specific studies. Overall beach visitation across the U.S. is estimated at 582 million annual beach days. Nationwide estimates of activity levels for other types of coastal recreation are not as easily accessible.

In many coastal communities, recreation and tourism are major economic sectors. Tourists drawn to coastal regions by beaches and other coastal and marine resources spend money at hotels, restaurants, and recreation-based local businesses (e.g., kayak rental shops). **Table 22-2** presents estimates of the regional economic scale of businesses in the tourism and recreation sector that are ocean-dependent based on data from NOEP. The NOEP data indicate that ocean-dependent recreation and tourism businesses account for approximately $120 billion in Gross Domestic Product in coastal states and employ approximately 2.9 million workers.

Though coastal recreation and tourism are major economic sectors in many coastal areas, the value of coastal and marine resources to recreational users cannot be captured solely by measured economic activity. Many coastal recreational activities are freely available to the public, including beach visitation, swimming, and hiking. Despite the lack of an observable market price, such activities still provide value to recreators and tourists. These types of recreational values are not fully captured in the NOEP estimates of the economic scale of the ocean-dependent recreation and tourism sector. However, these recreational values can be estimated through economic methods applied in the literature. Examples include travel cost methods, which estimate recreation values based on the time and money spent to travel to and use a resource, and contingent valuation methods, which directly survey recreators on their willingness to pay (WTP) to use a resource.

Table 22-3 presents recreation use value estimates for various coastal locations across the U.S. This table includes all of the studies used in BOEM's OECM as well as additional studies identified in the Recreation Use Values Database (RUVD)[10]. As the table shows, recreation value estimates are highly variable, reflecting location-specific factors such as site quality (e.g., clarity of the water at a beach, softness of the sand, etc.), amenities, and crowds.

[10] Oregon State University (2016).

Table 22-1. Beach visitation by state

State	Annual Beach Days (millions)	Source/Method
WA	12.4	Estimate number of beach days from Washington SCORP (2013) using reported average days of participation, percent of beach use occurring at saltwater beaches, and percent of residents participating in beach activities.
OR	17.3	Use value reported in Oregon Parks and Recreation Department (2013) for saltwater beach activities.
CA	116	Extrapolate total beach visitation from the South Coast Recreation Survey (Chen 2015) based on distribution of state resident beach use in the CA Survey of Public Opinions and Attitudes (California State Parks 2014).
ME	18.3	Apply estimated beach use per beach mile for Rhode Island, as derived from Parsons and Firestone (2018) and EPA (2018a), to Maine beach miles reported in EPA (2018a).
NH	5.15	Apply estimated beach use per beach mile for Rhode Island, as derived from Parsons and Firestone (2018) and EPA (2018a), to New Hampshire beach miles reported in EPA (2018a).
MA	36.2	Parsons and Firestone (2018) report visitation only for the outer coast of Cape Cod. Extrapolate estimated day-trip beach use per beach mile on Cape Cod, as derived from Parsons and Firestone (2018) and EPA (2018b), to rest of Massachusetts beaches.
RI	14.8	Parsons and Firestone (2018) report visitation only for the ocean beaches in Rhode Island; extrapolate estimated beach use per beach mile, as derived from Parsons and Firestone (2018) and EPA (2018b), to rest of Rhode Island beaches (Narragansett Bay).
CT	9.15	Apply estimated beach use per beach mile for Rhode Island, as derived from Parsons and Firestone (2018) and EPA (2018a), to Connecticut beach miles reported in EPA (2018a).
NY	39.5	Parsons and Firestone (2018) report visitation only for the ocean beaches in New York; extrapolate estimated beach use per beach mile, as derived from Parsons and Firestone (2018) and EPA (2018b), to rest of New York beaches (Long Island Sound).
NJ	45.1	Estimated beach use from Parsons and Firestone (2018).
DE	11.3	Parsons and Firestone (2018) report visitation only for the ocean beaches in Delaware; calculated total beach use for Delaware based on the sum of estimated beach use from Parsons and Firestone (2018) and beach use at Delaware Bay beaches from Parsons et al. (2013).
MD	12.1	Use estimated beach use for ocean beaches in Maryland from Parsons and Firestone (2018); apply estimated beach use per bay beach mile for Delaware's bay beaches, as derived from Parsons et al. (2013), to Maryland bay beach miles reported in EPA (2018a). Total beach visitation is the sum of ocean and bay beach visits.
VA	13.7	Use estimated beach use for ocean beaches in Virginia from Parsons and Firestone (2018); extrapolate estimated beach use per bay beach mile for Delaware, as derived from Parsons et al. (2013) and EPA (2018b), to bay beach miles in Virginia reported in EPA (2018b).
NC	33.4	Estimated beach use from Parsons and Firestone (2018).
SC	47.2	Estimated beach use from Parsons and Firestone (2018).
GA	14.4	Apply estimated beach use per beach mile for North Carolina, as derived from Parsons and Firestone (2018) and EPA (2018a), to Georgia beach miles reported in EPA (2018a).
FL	78.2	Extrapolate *Deepwater Horizon* Damage Assessment beach-use values for Florida to entire state using the regional distribution of beach use estimated in the FL DEP (2013) and FL DEP (2016).
LA	3.14	Extrapolate *Deepwater Horizon* Damage Assessment beach use per mile for Mississippi and Alabama to Louisiana.
MS	1.52	Baseline visitation from *Deepwater Horizon* Damage Assessment.
AL	5.01	Baseline visitation from *Deepwater Horizon* Damage Assessment.
TX	48.1	Extrapolate *Deepwater Horizon* Damage Assessment beach use per mile for Mississippi and Alabama to Texas.
Total	**582**	

Table 22-2. Measures of the ocean-dependent tourism and recreation sector by state (2016)

State	Establishments	Employment	Wages (million 2019$)	GDP (million 2019$)
Alabama	1,300	31,000	350	710
Alaska	2,400	48,000	570	1,100
California	24,000	560,000	12,000	25,000
Connecticut	3,000	55,000	990	2,200
Delaware	1,400	28,000	420	900
Florida	23,000	500,000	10,000	21,000
Georgia	1,300	28,000	330	680
Hawaii	4,300	120,000	3,700	7,900
Louisiana	4,200	100,000	1,100	2,700
Maine	3,400	51,000	760	1,600
Maryland	4,700	100,000	1,500	3,500
Massachusetts	5,800	95,000	2,100	4,300
Mississippi	1,100	33,000	240	500
New Hampshire	590	15,000	170	370
New Jersey	8,900	140,000	2,100	4,100
New York	22,000	370,000	12,000	25,000
North Carolina	3,200	48,000	740	1,600
Oregon	2,400	37,000	570	1,200
Rhode Island	2,400	44,000	820	1,800
South Carolina	3,400	79,000	1,500	3,900
Texas	6,200	180,000	940	2,000
Virginia	4,100	120,000	1,200	2,300
Washington	6,700	130,000	2,000	4,800
Total	140,000	2,900,000	57,000	120,000

Source: NOEP. Ocean Economy Data. June 17, 2020 at: https://www.oceaneconomics.org/Market/ocean/oceanEcon.asp
Note: Values converted to 2019 dollars using the BEA Implicit Price Deflator for Gross Domestic Product.

Table 22-3. Recreation trip valuation estimates

Study	Location	Recreation Activity	Estimated Value (2019$)	Units
Parsons et al. (2009)	Padre Island National Seashore, TX	Beach use	$24	per trip
Lew and Larson (2008)	San Diego County, CA	Beach use	$32	per user day
Hausman et al. (1995)	Prince William Sound, AK	Boating	$303	per trip
Hausman et al. (1995)	Prince William Sound, AK	Hiking/viewing	$355	per trip
Gornik et al. (2013)	Channel Islands, CA	Boating	$59	per trip
Gornik et al. (2013)	Channel Islands, CA	Diving/Snorkeling	$59	per trip
Gornik et al. (2013)	Channel Islands, CA	Beach use/Hiking	$58	per trip

Study	Location	Recreation Activity	Estimated Value (2019$)	Units
Bouchard B-120 Oil Spill Lost Use Technical Working Group 2009	Buzzards Bay, MA	Beach use	$42	per user day
Parsons et al. (2013)	Delaware Bay, DE	Beach use	$38–$42	per user day
Bin et al. (2005)	North Carolina	Beach use	$15–$102	per user day
Landry and McConnell (2007)	Georgia	Beach use	$11–$13	per user day
Bell and Leeworthy (1990)	Florida	Beach use	$60	per user day
English and McConnell (2015)	Florida	Beach use	$38	per user day
English and McConnell (2015)	North Gulf (Louisiana, Mississippi, Alabama, Florida)	Beach use	$38–$42	per user day
English and McConnell (2015)	North Gulf (Louisiana, Mississippi, Alabama, Florida)	Boating	$17	per user day
Lew and Larson (2008)	San Diego County, CA	Beach use	$32	per user day
Leggett et al. (2014)	Southern CA	Beach use	$35	per user day
English (2010)	San Francisco Bay Area, CA	Beach use	$11–$28	per user day

Note: Values converted to 2019 dollars using the BEA Implicit Price Deflator for Gross Domestic Product.

22.2 Impacts of Oil Spills on Recreation and Tourism

As outlined above, coastal and marine resources provide services that support recreational activities such as beach use, boating, wildlife viewing, swimming, and diving. If the resources that provide these services are oiled, recreational use of the resources may not be possible or the value of the resources to recreational users may be diminished. Impacts to the recreational value of coastal and marine resources can be reflected in:

1. A reduction in the use of resources (e.g., reduction in number of beach trips) as a result of real or perceived impacts of a spill; and
2. A reduction in the WTP to use the affected resource due to actual or perceived degradation of the quality of the resource.

Immediately following a median-range oil spill, many recreational activities may not be possible at all due to ocean and/or shoreline oiling. For instance, oiled beaches are often closed to the public until shoreline cleanup operations are completed. As a result, some planned recreation trips to spill sites may shift to unaffected but less preferred sites and some trips may be cancelled altogether. Reductions in tourism and recreational use often remain even after spill response is complete due to public perceptions of degraded resource quality or increased health risk.

Though spills often lead to reductions in tourist trips to the affected region, some median-range spills result in an influx of response workers and media members to coastal communities. In the short term, the arrival of such workers can partially mitigate the negative spill impacts on some coastal tourism-focused businesses, including hotels.

The magnitude of spill impacts on recreation and tourism are influenced by a variety of factors, including the following:

- **Spill volume.** Median-range spills result in shoreline oiling, which makes it difficult to engage in coastal recreation at substitute sites near preferred sites. Median-range spills with surface sheens are also more likely to reduce boating opportunities or result in boating closures.

- **Spill Location.** Spills near popular coastal recreation/tourist sites will result in greater impacts than spills in remote areas. Similarly, spills closer to shore are more likely to result in shoreline oiling and closures of recreational areas.
- **Season.** Spills that occur during peak seasons for recreation and tourist activity will have greater impacts than spills that occur during seasons with lower recreation and tourist activity.
- **Effectiveness of response actions.** Response actions, such as inlet protection strategies to keep oil from entering estuaries and lagoons, use of dispersants, and mechanical on-water recovery, can reduce the dispersion of oil and shoreline oiling and thus reduce the impacts of spills on recreation and tourism resources.
- **Environmental conditions that affect oil transport.** The presence of wind, waves, and ocean currents can increase the dispersion of spilled oil and result in greater damages to recreation and tourism resources.

Ideally, the recreational damages caused by an oil spill would be estimated using data on: 1) the baseline level of recreational activity; 2) the change in the level of recreational activity following the spill; and 3) the per unit (e.g., per user day or per beach trip) value that individuals place on these activities in the baseline and during the impact period. Other metrics that characterize spill injury to recreation and tourism resources include changes in the number of recreational/tourist trips or recreational/tourist days, changes in recreational use value, changes in visitor spending, or changes in revenues or profits at recreational/tourist businesses. Spill impacts on recreation and tourism have been extensively studied for a limited number of median-range oil spills. The sections below describe several median-range oil spills with documented impacts to recreation and tourism. Two of these spills, the T/B *Bouchard B-120* and the M/V *Cosco Busan* were of a heavy fuel oil; however, they are included because of the paucity of data of median-range spill impacts to recreation and tourism resources.

22.2.1 T/B *Bouchard B-120*

On 27 April 2003, the barge *Bouchard-120* ruptured its hull after running aground and spilled approximately 2,333 bbl of No. 6 fuel oil into Buzzards Bay. Oil was driven by winds and currents throughout the bay and nearby coastal waters of Massachusetts and Rhode Island. Approximately 137 km of shoreline in Massachusetts and 24 km in Rhode Island were contaminated with oil.

Natural resource Trustees quantified lost recreational use services as the value of lost recreational shell fishing trips, lost recreational boating trips, and lost shoreline trips. Using visitation data from before and after the incident, the Trustees determined that 36,441 shoreline trips, 47,928 shell fishing trips, and 987 boating trips were lost (Bouchard B-120 Oil Spill Lost Use Technical Working Group 2009). They applied benefits-transfer methods to determine the value of lost shoreline and boating trips and conducted a site-specific study to determine the value of lost shell fishing trips. The site-specific study used shell fishing license data and a travel-cost model to determine the average value per shell fishing trip. The total value of lost recreational use services was calculated to be \$3.70 million[11] (Bouchard B-120 Oil Spill Lost Use Technical Working Group 2009).

22.2.2 M/V *Cosco Busan* Oil

On the morning of 7 November 2007, the container ship *Cosco Busan* struck a tower of the San Francisco-Oakland Bay Bridge. The gash in the hull of the vessel created by the allision resulted in the release of 1,275 bbl of heavy fuel oil into the San Francisco Bay over the course of approximately 53

[11] This estimate and all monetized estimates presented in this chapter are in year 2019 dollars.

minutes. Winds and currents caused the spill to spread rapidly and moved some of the oil outside of the Bay, impacting an area from Half Moon Bay to Point Reyes.

A benefits-transfer approach was employed to estimate damages to recreational use associated with the spill, including damages related to shoreline use and boating.[12] The study of recreational impacts combined data on the baseline number of different types of trips with survey data collected following the spill to understand the impact of the oil spill on the number of recreational trips made. Specifically, these studies compiled data on baseline park and marina use, along with visitation figures obtained post spill from park databases and visitor and telephone surveys. The value of the loss in activity was estimated separately for beach use and boating. For beach use, the Trustees performed a telephone survey of Bay Area residents to compile data on their recreational trips to shoreline sites in the Bay Area. The data collected from this survey were then used in a travel-cost model, yielding an estimate of $22.15 per lost trip, though this value declined over time from $27.49 to $10.80 (Cosco Busan Oil Spill Trustees 2012). This reduction in lost value reflected the increased availability of substitutes as the number of sites affected by the spill declined over time. For boating, the Trustees applied values of $92.17 per trip for sailboat and motorboat trips and $61.45 for dragon boats (Cosco Busan Oil Spill Trustees 2012). An estimated 984,451 shoreline trips and 26,600 boating trips were lost, with damages totaling $21.8 million for shoreline use and $2.4 million for boating (Cosco Busan Oil Spill Trustees 2012).

22.2.3 T/V *American Trader*

On 7 February 1990, the tanker *American Trader* ran aground off the coast of Southern California, spilling 9,919 bbl of crude oil. By February 12, nearly 160 km^2 of ocean were covered by oil. On 13 February, a storm washed much of the remaining oil over 22.5 km of shoreline. Many beaches were closed to the public due to the spill until early March, with the last beaches opening 14 March 1990. Offshore waters in the area were closed to boating and fishing for approximately 2 weeks.

The *American Trader* spill was the first time the recreational impacts of a spill were monetized in California. No large-scale original data collection efforts (e.g., a travel-cost survey) were undertaken, in part because the Trustees expected a settlement, but also because large-scale recreation use data for the area were already available. Some of the affected beaches were administered by the California Department of Parks and Recreation, which tracked beach use for fee purposes. Much of these use estimates were based on vehicle counts, so adjustments were made to account for the number of people per vehicle, as well as the ratio of walk-on beach goers to car-arriving beach goers. The Trustees incorporated these data into a parametric model of beach attendance. Based on this model, the Trustees estimated 454,281 lost beach trips during the beach closure period and 278,986 lost trips beyond the beach closure period through 31 March 1990, for a total of 733,267 lost trips (Chapman and Hanemann 2001). In addition, the Trustees estimated 30,485 lost surfing trips not captured in the available beach attendance data, yielding a total of 763,752 lost beach trips (Chapman and Hanemann 2001). Complementing the estimated number of lost beach trips, the Trustees also estimated 31,000 lost boating trips.

To value foregone beach recreation, the Trustees used a benefits-transfer approach, applying the estimated value per beach trip for Florida, $23.26, to lost beach trips in Southern California. For surfing, the Trustees applied a unit value of $29.89 based on the entrance fee at an inland water park in Southern California. Using these unit values, the Trustees estimated total losses of $18 million for beach recreation and surfing (Chapman and Hanemann 2001). For boating, the benefits-transfer approach applied by the Trustees resulted in estimated damages of $2.2 million.

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[12] The Trustees also examined recreational fishing impacts. These are discussed separately in Chapter 17.

22.2.4 Refugio Beach

On 19 May 2015 a 60-cm diameter onshore pipeline located along the Santa Barbara County coastline in Southern California ruptured, resulting in the release of 2,934 bbl of Monterey crude oil. The spilled oil flowed into a nearby culvert and ultimately discharged into the Pacific Ocean at Refugio State Beach. As a result of the oil discharge, Refugio State Beach and El Capitan State Beach were closed for 59 and 37 days, respectively. In addition, access to adjacent smaller beaches in the area was restricted for more than 3 months. Recreational fishing in the region was closed for 41 days (Refugio Beach Oil Spill Trustees 2020). Farther down the coast in Los Angeles County, beaches in Santa Monica Bay and Long Beach closed for 3 days after tar balls washed up on the beaches.

The Trustees' assessment of recreation-related damages focused on three specific categories of recreation: coastal camping, non-camping shoreline recreation, and boating and offshore recreation. To assess spill-related damages for each of these categories, the Trustees quantified the change in each activity (i.e., reduced number of recreational trips) and applied benefits-transfer methods to monetize the economic value of these changes. The Trustees' assessment of the change in activity was based on (1) site closures and posted advisories after the spill; (2) field data collected around the time of the spill in selected coastal recreation sites in the area (e.g., selected beaches in Santa Barbara and Ventura Counties); (3) compilations of existing data on spill effects or baseline use (e.g., parking fee data for select coastal lots between Santa Barbara and Malibu); and (4) field data collected on the first anniversary of the spill to fill in gaps in baseline data. The studies upon which the Trustees relied to value the estimated changes in recreational activity all focused on recreational activity along California's coast.

Based on the methods outlined above, the Trustees estimated more than 140,000 lost recreational user days. The monetized damages associated with this reduction in user days include $1.6 million in camping losses, $2.7 million in non-camping shoreline losses, and $0.16 million in boating and offshore recreation losses (Refugio Beach Oil Spill Trustees 2020).

22.2.5 UNOCAL, Avila Beach

On 3 August 1992 a UNOCAL oil pipeline ruptured and spilled approximately 600 bbl of San Joaquin Valley crude oil onto lands and waters near Avila Beach, California. The spill flowed from the pipeline, into a gully, down a cliff, and into marine waters. Although the Trustees maintained that recreational losses resulted from the spill, these losses were not quantified. However, Avila Beach and Olde Port Beach were officially closed for 5 days at the height of the summer beach season. In addition, Boulder Cove, which is also used by people for wildlife viewing, was closed to the public from 4 August until the end of the response (CDFG and USFWS 1999).

A local king salmon fishery was also affected by the spill. Approximately 50,000 state-owned king salmon were being raised in saltwater pens near the spill site and were scheduled for release in mid-August. Fish losses in the rearing pens had been minimal prior to the spill, but the fish began to show symptoms and behaviors consistent with vibriosis 8 days after oil arrived at the site. Vibriosis is caused by a marine bacterium, *Vibrio anguillarum*, that is always present in water, but typically becomes a problem only when the immune systems of fish are compromised, for example from the stress associated with an oil spill. The incubation period from infection by *Vibrio anguillarum* to the onset of symptoms is approximately 8 days, and the timing of the fish mortalities in the saltwater pens near the site is consistent with this incubation period. On the 6th day after oil arrived at the site only 11 fish were found dead, but that figure increased to 206 dead fish on the 9th day (CDFG and USFWS 1999). Based on underwater observations, the total loss from vibriosis over time was approximately 10,000 fish.

22.2.6 T/V *World Prodigy*

On 23 June 1989 the oil tanker *World Prodigy* grounded on Brenton Reef in Narragansett Bay near Newport, Rhode Island, causing the tanker to spill approximately 6,900 bbl of No. 2 fuel oil into the bay. Although the Trustees did not quantify recreational impacts, several beaches near the spill site were closed for approximately 2 months after the spill, during the summer beach season (NOAA 1996).

Table 22-4 summarizes the documented impacts to recreation and tourism from the spills identified above. When available, the table includes quantified economic losses.

22.3 Summary and Information Needs for Assessing Impacts to Recreation and Tourism

Oil spills can result in reductions in the use of recreation and tourism resources as a result of real or perceived oil spill impacts. They may reduce the value that recreators or tourists derive from the use of recreation resources due to actual or perceived degradation in resource quality. The magnitude of spill impacts on recreation and tourism resources depends on a variety of factors, including spill size, location, and effectiveness of response actions. All else being equal, spills that result in more shoreline oiling and spills that occur near popular areas for recreation and tourist activity result in greater impacts.

Impacts to recreation and tourism have been documented for only a limited number of median-range spills. As a result, the impacts of any future spill on recreation and tourism are considerably uncertain and are likely to depend on spill-specific characteristics. The small number of studies that have been published focus on recreational activity but do not address impacts to the tourism industry. A broader analysis that considers impacts to tourism-dependent businesses and workers would provide a more comprehensive understanding of spill impacts. In addition, data on baseline recreational use are not available at the local level for most areas, making it difficult to understand likely recreational and tourism impacts in advance of potential spills. Although recreational use data have been developed as part of the damage assessment process for specific communities or beaches where oil has reached shore, greater availability of reliable baseline data at the local level could enhance understanding of potential recreational impacts before a spill occurs and could inform policy decisions regarding use of the marine environment. The state level data summarized above may also inform such decisions, but data at the local level would provide a clearer understanding of potential spill impacts.

Table 22-4. Studies with documented or estimated impacts to and recovery of recreation and tourism from crude oil, No. 2 fuel oil, and heavy fuel oil spills (500–20,000 bbl)

Oil Spill	Oil Volume and Type	Affected Activity(s)	Documented Effects/Impacts	Monetized Impacts (in 2019 dollars)	Spatial Extent of Impacts	Duration of Impacts (years)
2003 T/*B Bouchard B-120*, Buzzard Bay, MA[a]	2,333 bbl No. 6 fuel oil	Shoreline use, boating, fishing	36,441 shoreline trips, 47,928 shellfishing trips, and 987 boating trips lost.	Total recreational value lost of $3.70 million.	137 km of shoreline in MA and approximately 24 km in RI.	1.5
2007, M/V *Cosco Busan*, San Francisco Bay, CA[b]	1,275 bbl heavy fuel oil	Tourism, beach visitation	984,451 lost shoreline trips and 26,600 lost boating trips.	$21.8 million in lost consumer surplus for shoreline use and $2.4 million in list consumer surplus for lost boating activity.	San Francisco Bay Area; Half Moon Bay to Point Reyes.	<1
1990, T/V *American Trader*, Huntington Beach, CA[c]	9,919 bbl Alaska North Slope crude	Beach visitation, shoreline use, boating	763,752 lost beach trips; 31,000 lost boat trips.	Beach recreational loss of $18.0 million. Boating losses of $2.2 million.	Beaches in Los Angeles County and Orange County, California.	<1
2015, Refugio Beach, Santa Barbara, CA[d]	2,934 bbl Monterey crude	Coastal camping, non-camping shoreline recreation, boating and offshore recreation	140,000 lost recreational user days across uses.	$1.6 million in camping losses. $2.7 million in non-camping shoreline losses. $0.16 million in boating & offshore recreation losses.	Coastal areas in Santa Barbara, Ventura, and Los Angeles Counties.	<1
1992 UNOCAL, Avila Beach, CA[e]	600 bbl San Joaquin Valley crude	Beach visitation, wildlife viewing, sport fishing	Beach closures; scenic bluff closure; and mortality of king salmon being raised for release.	None.	Avila Beach and Olde Port Beach.	<1
1989 T/V *World* Prodigy, Narragansett Bay, RI[f]	6,900 bbl No. 2 fuel oil	Beach visitation	Beach closures.	None.	Narraganset Bay.	<1

[a]Bouchard B-120 Oil Spill Lost Use Technical Working Group (2009); [b]Cosco Busan Oil Spill Trustees (2012); [c]Chapman and Hanemann (2001); [d]Refugio Beach Oil Spill Trustees (2020); [e]CDFG and USFWS (1999); [f]NOAA (1996)

23 References

Abt Associates. 2018. Open ocean oil response: methods for estimating oil slick coverage. Silver Spring (MD): National Oceanic and Atmospheric Administration. 86 p.

Allaway WG. 2009. Mangrove die-back in Botany Bay. Wetlands Australia. 2(1):2-7.

Alldredge AL, Silver MW. 1988. Characteristics, dynamics and significance of marine snow. Progress in Oceanography. 20(1):41-82.

Almeda R, Baca S, Hyatt C, Buskey EJ. 2014a. Ingestion and sublethal effects of physically and chemically dispersed crude oil on marine planktonic copepods. Ecotoxicology. 23(6):988-1003.

Almeda R, Hyatt C, Buskey EJ. 2014b. Toxicity of dispersant Corexit 9500A and crude oil to marine microzooplankton. Ecotoxicology and Environmental Safety. 106:76-85.

Alonso-Alvarez C, Munilla I, Lopez-Alonso M, Velando A. 2007. Sublethal toxicity of the Prestige oil spill on yellow-legged gulls. Environment International. 33:773-781.

Alves-Stanley CD, Worthy G, Bonde RK. 2010. Feeding preferences of West Indian manatees in Florida, Belize, and Puerto Rico as indicated by stable isotope analysis. Marine Ecology Progress Series. 402:255-267.

American Trader Trustee Council. 2001. Final restoration plan and environmental assessment for seabirds injured by the American Trader oil spill. Report of the American Trader Natural Resource Trustee Council, U.S. Fish and Wildlife Service, California Department of Fish and Game, and National Oceanic and Atmospheric Administration. 126 p.

Anderson DW, Gress F, Fry DM. 1996. Survival and dispersal of oiled brown pelicans after rehabilitation and release. Marine Pollution Bulletin. 32(10):711-718.

Anitra Oil Spill Natural Resource Trustees. 2004. Final natural resources restoration plan and NEPA environmental assessment for the *Anitra* oil spill of May 1996. Trenton (NJ): New Jersey Department of Environmental Protection, U. S. Fish and Wildlife Service, National Oceanic and Atmospheric Administration. 20 p.

Antrim LD, Thom RM, Gardiner WW, Cullinan VI, Shreffler DK, Bienert RW. 1995. Effects of petroleum products on bull kelp (*Nereocystis luetkeana*). Marine Biology. 122(1):23-31.

Apex Houston Trustee Council. 2011. Apex Houston trustee council final report - March 2011. 32 p.

Arquette M, Cole M, Cook K, LaFrance B, Peters M, Ransom J, Sargent E, Smoke V, Stairs A. 2002. Holistic risk-based environmental decision making: a Native perspective. Environmental Health Perspectives. 110(2):259-264.

ATSDR (Agency for Toxic Substances and Disease Registry). 1999. Total petroleum hydrocarbons (TPH) ToxFAQs. 1-2 p.

Baca B, Rosch E, DeMicco ED, Schuler PA. 2014. TROPICS: 30-year follow-up and analysis of mangroves, invertebrates and hydrocarbons. Proceedings of the International Oil Spill Conference; Savannah (GA). American Petroleum Institute. 1734-1748 p.

Bagby SC, Fischer GB, Reddy CM, Valentine DL. 2014. Deposition and biodegradation of submerged oil from the Deepwater Horizon. Proceedings of the AGU Fall Meeting; Washington (DC). American Geophysical Union. OS51B-0980 p.

Baguley JG, Montagna PA, Cooksey C, Hyland JL, Woo Bang H, Morrison C, Kamikawa A, Bennetts P, Saiyo G, Parsons E, et al. 2015. Community response of deep-sea soft-sediment metazoan meiofauna to the Deepwater Horizon blowout and oil spill. Marine Ecology Progress Series. 528:127-140.

Ball D, Clayburn R, Cordero R, Edwards B, Grussing V, Ledford J, McConnell R, Monette R, Steelquist R, Thorsgard E, et al. 2015. A guidance document for characterizing tribal cultural landscapes.

Camarillo (CA): Bureau of Ocean Energy Management. Interagency agreement no.: M12PG00035. OCS Study BOEM 2015-047. 38 p.

Ballou T, Dodge RE, Hess S, Knap A, Sleeter TD. 1987. Effects of a dispersed and undispersed crude oil on mangroves, seagrasses and corals. Washington (DC): Prepared for the American Petroleum Institute by NOVA Southeastern University. 196 p.

Balseiro A, Espi A, Marquez I, Perez V, Ferreras MC, Garcia Marin JF, Prieto JM. 2005. Pathological features in marine birds affected by the Prestige's oil spill in the north of Spain. Journal of Wildlife Diseases. 41(2):371-378.

Barbier E, Hacker S, Kennedy C, Koch E, Stier A, Silliman B. 2011. The value of estuarine and coastal ecosystem services. Ecological Monographs. 81(2):169-193.

Barron MG, Vivian DN, Heintz RA, Yim UH. 2020. Long-term ecological impacts from oil spills: comparison of *Exxon Valdez*, *Hebei Spirit*, and Deepwater Horizon. Environmental Science & Technology. 54(11):6456-6467.

Basso KH. 1996. Wisdom sits in places: landscape and language among the Western Apache. Albuquerque (NM): University of New Mexico Press. 192 p.

Beck C, Bonde R, Odell D. 1978. Manatee mortality in Florida during 1978. In: Brownell RL, Ralls K, editors. Proceedings of the The West Indian manatee in Florida; Tallahassee (FL). Florida Department of Natural Resource. 3-16 p.

Behr-Andres C, Wiegers JK, Forester SD, Conn J. 2001. Tundra spill cleanup and remediation tactics: a study of historic spills and literature. Fairbanks (AK): AMEC Earth & Environmental and Alaska Department of Environmental Conservation. 175 p.

Bejarano AC, Dunagan H, Michel J. 2015. Literature review: effects of oil, shoreline treatment, and physical disturbance on sand beach habitats. Seattle (WA): Assessment and Restoration Division, NOAA Office of Response and Restoration. 149 p.

Bejarano AC, Michel J. 2016. Oil spills and their impacts on sand beach invertebrate communities: a literature review. Environmental Pollution. 218:709-722.

Beland M, Biggs TW, Roberts DA, Peterson SH, Kokaly RF, Piazza S. 2017. Oiling accelerates loss of salt marshes, southeastern Louisiana. PLoS ONE. 12(8):e0181197.

Bell FW, Leeworthy VR. 1990. Recreational demand by tourists for saltwater beach days. Journal of Environmental Economics and Management. 18:189-205.

Berenshtein I, Paris CB, Perlin N, Alloy MM, Joye SB, Murawski S. 2020. Invisible oil beyond the Deepwater Horizon satellite footprint. Science Advances. 6(7):eaaw8863.

Bergen A, Alderson C, Bergfors R, Aquila C, Matsil MA. 2000. Restoration of a *Spartina alterniflora* salt marsh following a fuel oil spill, New York City, NY. Wetlands Ecology and Management. 8(2):185-195.

Bergeon Burns CM, Olin JA, Woltmann S, Stouffer PC, Taylor SS. 2014. Effects of oil on terrestrial vertebrates: predicting impacts of the Macondo blowout. BioScience. 64(9):820-828.

Bertness MD, Gaines SD, Hay ME. 2001. Marine community ecology. Sunderland: Sinauer Associates. 550 p.

Bin O, Landry CE, Ellis CL, Vogelsong H. 2005. Some consumer surplus estimates for North Carolina beaches. Marine Resource Economics. 20:145-161.

Björdal CG. 2012. Microbial degradation of waterlogged archaeological wood. Journal of Cultural Heritage. 13S:S118-S122.

Björdal CG, Nilsson T. 2008. Reburial of shipwrecks in marine sediments: a long-term study on wood degradation. Journal of Archaeological Science. 35:862-872.

Blaylock WM, Houghton JP. 1989. Infaunal recovery at Ediz Hook following the *Arco Anchorage* oil spill. Proceedings of the International Oil Spill Conference. American Petroleum Institute; San Antonio (TX). 421-426 p.

Blumer M, Sass J, Souza G, Sanders HL, Grassle JF, Hampson GR. 1970a. The West Falmouth oil spill: persistence of the pollution eight months after the accident. Woods Hole (MA): Woods Hole Oceanographic Institution. 60 p.
Blumer M, Souza G, Sass J. 1970b. Hydrocarbon pollution of edible shellfish by an oil spill. Marine Biology. 5:195-202.
BOEM (Bureau of Ocean Energy Management). 2011. Alaskan shipwreck table. Anchorage (AK): BOEM, Alaska OCS Region. 610 p.
BOEM (Bureau of Ocean Energy Management). 2016. Alaska OCS lease sale 244: 2016. Final Environmental Impact Statement. Anchorage (AK). BOEM 2016-069.
BOEM (Bureau of Ocean Energy Management). 2017. Gulf of Mexico OCS oil and gas lease sales: 2017-2022, Gulf of Mexico lease sales 249, 250, 251, 252, 253, 254, 256, 257, 259, and 261. Final multisale environmental impact statement. New Orleans (LA). BOEM 2017-009.
Bolten AB. 2003. Variation in sea turtle life history patterns: neritic vs. oceanic life history stages. In: Lutz PL, Musick JA, Wyneken J, editors. The biology of sea turtles vol II. Boca Raton (FL): CRC Press. 243-257 p.
Bossart GD. 2011. Marine mammals as sentinel species for oceans and human health. Veterinary Pathology Online. 48(3):676-690.
Bouchard B-120 Oil Spill Lost Use Technical Working Group. 2009. Bouchard B-120 oil spill Buzzards Bay, Massachusetts. Lost use valuation report. 290 p.
Bowen WD. 2017. Pinniped ecology. In: Wursig B, Thewissen JGM, Kovacs KM, editors. Encyclopedia of marine mammals. 3rd ed. Cambridge (MA): Academic Press. 852-861 p.
Brendel J. 1985. Revegetation of arctic tundra after an oil spill: a case history. Proceedings of the International Oil Spill Conference; Los Angeles (CA). American Petroleum Institute. 315-318 p.
Briggs KT, Yoshida SH, Gershwin ME. 1996. The influence of petrochemicals and stress on the immune system of seabirds. Regulatory Toxicology and Pharmacology. 23(2):145-155.
Brooks JM, Fisher C, Cordes E, Baums I, Bernard B, Church R, Etnoyer P, German C, Goehring E, MacDonald IR, et al. 2016. Exploration and research of Northern Gulf of Mexico deepwater natural and artificial hard-bottom habitats with emphasis on coral communities: reefs, rigs, and wrecks—“Lophelia II”. New Orleans (LA): U.S. Department of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region. OCS Study BOEM 2016-021. 268 p.
Brown ED, Baker TT, Hose JE, Kocan RM, Marty GD, McGurk MD, Norcross BL, Short J. 1996. Injury to the early life history stages of Pacific herring in Prince William Sound after the Exxon Valdez oil spill. In: Rice C, Spies R, Wolfe D, Wright B, editors. Proceedings of the American Fisheries Society Symposium; Bethesda (MD). 448-462 p.
Burger AE. 1993. Estimating the mortality of seabirds following oil spills: effects of spill volume. Marine Pollution Bulletin. 26(3):140-143.
Burger J. 1994. Before and after an oil spill: the Arthur Kill. New Brunswick (NJ): Rutgers University Press. 305 p.
Burger J. 1997. Effects of oiling on feeding behavior of sanderlings and semipalmated plovers in New Jersey. Condor. 99:290-298.
Burger J, Tsipoura N. 1998. Experimental oiling of sanderlings (*Calidris alba*): behavior and weight changes. Environmental Toxicology and Chemistry. 17(6):1154-1158.
Burns KA, Teal JM. 1971. Hydrocarbon incorporation into the salt marsh ecosystem from the West Falmouth oil spill. Woods Hole (MA): Woods Hole Oceanographic Institution.
Burns KA, Teal JM. 1979. The West Falmouth oil spill: hydrocarbons in the salt marsh ecosystem. Estuarine and Coastal Marine Science. 8:349-360.
Butler JN, Morris BF, Cadwallader J, Stoner AW. 1983. Studies of sargassum and the sargassum community. Special Publication No. 22. Bermuda Biological Station for Research, Ferry Reach. 307 p.

California State Parks. 2014. Survey on public opinions and attitudes on outdoor recreation in California. State of California Natural Resources Agency. 289 p.
Card JC, Meehan JA. 1991. Response to the *American Trader* oil spill. Proceedings of the International Oil Spill Conference. San Diego (CA). 305-311 p.
Carter HR, Lee VA, Page GW, Parker MW. 2003. The 1986 *Apex Houston* oil spill in central California: seabird injury assessment and litigation process. Marine Ornithology. 31:9-19.
Carter J, MacGregor C, Tidmarsh G, Chandler P, Parsons J, Bonke C. 1985. Monitoring programs associated with the Uniacke G-72 gas condensate blowout on the Scotian Shelf. Proceedings of the Eighth AMOP Technical Seminar; Ottawa (ON). Environment Canada. 20 p.
Casey JK. 1996. Assessment of the North Cape oil spill impacts to the 1996 piping plover breeding season at Trustom Pond NWR. Charlestown (RI): US Fish and Wildlife Service Report, Ninigret National Wildlife Refuge.
Catena J. 1996. Final M/V *World Prodigy* oil spill restoration plan and environmental assessment Narragansett Bay, Rhode Island. Gloucester (MA): National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service. 17 p.
CDFG (California Department of Fish and Game), USFWS (U.S. Fish and Wildlife Service). 1999. Restoration actions within the San Luis Obispo Creek watershed. Unocal oil spill, Avila Beach, 1992. 95 p.
CDPR (California Department of Parks and Recreation), CDFG (California Department of Fish and Game), USFWS (U.S. Fish and Wildlife Service). 2005. McGrath State Beach area Berry Petroleum oil spill, December 1993. Final restoration plan and environmental assessment. 104 p.
Chamberlain DW, Lindstedt-Siva J, Mancini ER, Levine RA, Miller JA. 1987. The *Arco Anchorage* oil spill: a review. Proceedings of the Tenth AMOP Technical Seminar; Ottawa (ON). Environment Canada. 10 p.
Chan EI. 1977. Oil pollution and tropical littoral communities: biological effects of the 1975 Florida Keys oil spill. Proceedings of the International Oil Spill Conference. American Petroleum Institute; New Orleans (LA). 539-542 p.
Chanton J, Zhao T, Rosenheim BE, Joye S, Bosman S, Brunner C, Yeager KM, Diercks AR, Hollander D. 2015. Using natural abundance radiocarbon to trace the flux of petrocarbon to the seafloor following the Deepwater Horizon oil spill. Environmental Science & Technology. 49:847-854.
Chapman DJ, Hanemann WM. 2001. Environmental damages in court: the *American Trader* case. The Law and Economics of the Environment, Paper No. 913. 43 p.
Chen C-L. 2015. Regulation and management of marine litter. In: Bergmann M, Gutow L, Klages M, editors. Marine anthropogenic litter. Cham (CH): Springer International Publishing. 395-428 p.
Chia FS. 1971. Diesel oil spill at Anacortes. Marine Pollution Bulletin. 2(7):105-106.
Chin CS, Church J. 2011. Field report: Fort Livingston, Grand Terre Island. U.S. Dept. of the Interior, National Park Service, National Center for Preservation Technology and Training.
Church R, Warren D, Cullimore R, Johnston L, Schroeder W, Patterson W, Shirley T, Kilgour M, Morris N, Moore J. 2007. Archaeological and Biological Analysis of World War II Shipwrecks in the Gulf of Mexico: Artificial Reef Effect in Deep Water. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2007-015. 387 p.
Church RA, Warren DJ. 2008. Viosca Knoll wreck: discovery and investigation of an early nineteenth-century wooden sailing vessel in 2,000 feet of water. New Orleans (LA): U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2008-018. OCS Study MMS 2008-018. 41 p.
Church RA, Warren DJ, Irion JB. 2009. Analysis of deepwater shipwrecks in the Gulf of Mexico: artificial reef effect of six World War II shipwrecks. Oceanography. 22(2):50-63.

Cimberg R, Mann S, Straughan D. 1973. A reinvestigation of Southern California rocky intertidal beaches three and one-half years after the 1969 Santa Barbara oil spill: a preliminary report. Proceedings of the Joint Conference on Prevention and Control of Oil Spills; Washington (DC). American Petroleum Institute. 697-702 p.

Coastal Response Research Center. 2010. *Deepwater Horizon* dispersant use meeting report: May 26-27, 2010. Durham (NH): University of New Hampshire. 108 p.

Colombo JC, Barreda A, Bilos C, Cappelletti N, Demichelis S, Lombardi P, Migoya MC, Skorupka C, Suarez G. 2005. Oil spill in the Rio de la Plata estuary, Argentina: 1. Biogeochemical assessment of waters, sediments, soils and biota. Environmental Pollution. 134(2):277-289.

Cosco Busan Oil Spill Trustees. 2012. *Cosco Busan* oil spill final damage assessment and restoration plan/environmental assessment. Prepared by California Department of Fish and Game, California State Lands Commission, National Oceanic and Atmospheric Administration, U.S. Fish and Wildlife Service, National Park Service, Bureau of Land Management.

Cosentino-Manning N. 2018. Refugio Beach oil spill effects of oiling on the seagrass *Phyllospadix torreyi*. SETAC North America 39th Annual Meeting 2018.

Cross WE, Martin CM, Thomson DH. 1987a. Effects of experimental releases of oil and dispersed oil on arctic nearshore macrobenthos. II. Epibenthos. Arctic. 40, SUPP. 1(1987):201-210.

Cross WE, Thomson DH. 1987. Effects of experimental releases of oil and dispersed oil on artic nearshore macrobenthos. I. Infauna. Arctic. 40, SUPP. 1(1987):184-220.

Cross WE, Wilce RT, Fabijan MF. 1987b. Effects of experimental releases of oil and dispersed oil on Arctic nearshore macrobenthos. III. Macroalgae. Arctic. 40, SUPP. 1(1987):211-219.

Crump R, Morley H, Williams A. 1998. West Angle Bay, a case study. Littoral monitoring of permanent quadrats before and after the "Sea Empress" oil spill. Behavioral Sciences. 9:207-225.

Culbertson JB, Valiela I, Olsen YS, Reddy CM. 2008a. Effect of field exposure to 38-year-old residual petroleum hydrocarbons on growth, condition index, and filtration rate of the ribbed mussel, *Geukensia demissa*. Environmental Pollution. 154(2):312-319.

Culbertson JB, Valiela I, Pickart M, Peacock EE, Reddy CM. 2008b. Long-term consequences of residual petroleum on salt marsh grass. Journal of Applied Ecology. 45(4):1284-1292.

Culbertson JB, Valiela I, Pickart M, Peacock EE, Reddy CM, Carter A, VanderKruik R. 2007. Long-term biological effects of petroleum residues on fiddler crabs in salt marshes. Marine Pollution Bulletin. 54(7):955-962.

Cullimore DR, Johnston LA. 2008. Microbiology of concretions, sediments and mechanisms influencing the preservation of submerged archaeological artifacts. International Journal of Historical Archaeology. 12(2):120-132.

Cullinane JP, McCarthy P, Fletcher A. 1975. The effect of oil pollution in Bantry Bay. Marine Pollution Bulletin. 6(11):173-176.

Cunningham F, Dean K, Hanson-Dorr K, Harr K, Healy K, Horak K, Link J, Shriner S, Bursian S, Dorr B. 2017. Development of methods for avian oil toxicity studies using the double crested cormorant (*Phalacrocorax auritus*). Ecotoxicology and Environmental Safety. 141:199-208.

D'Andrea MA, Reddy GK. 2014. Crude oil spill exposure and human health risks. Journal of Occupational and Experimental Medicine. 56(10):1029-1041.

Daly KL, Passow U, Chanton J, Hollander D. 2016. Assessing the impacts of oil-associated marine snow formation and sedimentation during and after the *Deepwater Horizon* oil spill. Anthropocene. 13:18-33.

Damour M, Church R, Warren D, Horrell C, Hamdan L. 2015. Gulf of Mexico shipwreck corrosion, hydrocarbon exposure, microbiology, and archaeology (GOM-SCHEMA) project: studying the effects of a major oil spill on submerged cultural resources. Proceedings of the Underwater Archaeology. 50-59 p.

Danovaro R, Fabiano M, Vincx M. 1995. Meiofauna response to the *Agip Abruzzo* oil spill in subtidal sediments of the Ligurian Sea. Marine Pollution Bulletin. 30(2):133-145.

Dauvin J. 1998. The fine sand *Abra alba* community of the Bay of Morlaix twenty years after the Amoco Cadiz oil spill. Marine Pollution Bulletin. 36(9):669-676.

Dauvin J. 2000. The muddy fine sand Abra alba–*Melinna palmata* community of the Bay of Morlaix twenty years after the Amoco Cadiz oil spill. Marine Pollution Bulletin. 40(6):528-536.

de la Huz R, Lastra M, Lopez J. 2011. Oil spills. Encyclopedia of Environmental Health. Elsevier. 251-255 p.

de Putron S, Ryland J. 1998. Effects of the 'Sea Empress' oil spillage on reproduction and recruitment of *Alcyonidium* (Bryozoa) populations on *Fucus serratus*. In: Edwards R, Sime H, editors. The Sea Empress oil spill. Lavenham (UK): Terence Dalton and the Chartered Institution of Water and Environmental Management. 457-466 p.

DeAlteris J, Thompson N, Skrobe L, Mauseth G, Erickson G. 1999. Effect of the T/B North Cape oil spill on the surf clam (*Spisula solidissima*) in the coastal waters of Southern Rhode Island. Proceedings of the International Oil Spill Conference. 993-997 p.

Dean TA, Jewett SC, Laur DR, Smith RO. 1996. Injury to epibenthic invertebrates resulting from the *Exxon Valdez* oil spill. In: Rice C, Spies R, Wolfe D, Wright B, editors. Proceedings of the American Fisheries Society Symposium; Bethesda (MD). 424-439 p.

Deep-C Consortium. 2013. Deep-C consortium update. Deep-C Consortium. http://archive.constantcontact.com/fs116/1104107396812/archive/1115864225729.html.

DeMicco E, Schuler PA, Omer T, Baca B. 2011. Net environmental benefit analysis (NEBA) of dispersed oil on nearshore tropical ecosystems: TROPICS–the 25th year research visit. Proceedings of the International Oil Spill Conference (IOSC). American Petroleum Institute. 1-14 p.

Demopoulos AWJ, Bourque JR, Cordes E, Stamler KM. 2016. Impacts of the Deepwater Horizon oil spill on deep-sea coral-associated sediment communities. Marine Ecology Progress Series. 561:51-68.

Derocher AE, Stirling I. 1991. Oil contamination of polar bears. Polar Record. 27(160):56-57.

Desvousges WH, Dunford RW, Mathews KE. 1992. Natural resource damages valuation: *Arthur Kill* oil spill. Research Triangle Institute.74.

Dethier MN. 2010. Overview of the ecology of Puget Sound beaches. Puget Sound shorelines and the impacts of armoring—proceedings of a state of the science workshop. U.S. Geological Survey Scientific Investigations Report 2010-5254. 35-42 p.

Dodge RE, Baca BJ, Knap A, Snedaker S, Sleeter T. 1995. The effects of oil and chemically dispersed oil in tropical ecosystems: 10 years of monitoring experimental sites. Washington (DC): Marine Spill Response Corporation. 95-014. 82 p.

Donohoe R, Baker G, Witting D. 2019. The 2015 Refugio Beach oil spill: oil exposure and potential effects to fish, invertebrate early life stages, and kelp. Draft Final Report. Monterey (CA): California Department of Fish and Wildlife, Office of Spill Prevention and Response and National Oceanic and Atmospheric Administration. 13 p.

Dooley JK. 1972. Fishes associated with the pelagic Sargassum complex, with a discussion of the Sargassum community. Marine Science. (16):1-32.

Dorr BS, Hanson-Dorr KC, Assadi-Porter FM, Selen ES, Healy KA, Horak KE. 2019. Effects of repeated sublethal external exposure to deep water horizon oil on the avian metabolome. Scientific Reports. 9(1):1-12.

Driskell WB, Payne JR. 2018. Macondo oil in northern Gulf of Mexico waters - part 2: dispersant-accelerated PAH dissolution in the *Deepwater Horizon* plume. Marine Pollution Bulletin. 129(1):412-419.

Dugan JE. 2018. Population survey results on intertidal talitrid amphipods for the Refugio Beach oil spill NRDA. Santa Barbara (CA): Marine Science Institute, University of California. 9 p.

Duke NC. 2016. Oil spill impacts on mangroves: recommendations for operational planning and action based on a global review. Marine Pollution Bulletin. 109(2):700-715.

Duke NC, Burns KA. 1999. Fate and effects of oil and dispersed oil on mangrove ecosystems in Australia. Australian Institute of Marine Science and CRC Reef Research Centre. 232-247 p.

Duke NC, Heydon L, Zahmel K, Prosser A. 2005. Lytton oil spill mangrove ecosystem monitoring project: final report. University of Queensland (AUS): Marine Botany Group, Centre for Marine Studies. 46 p.

Duke NC, Pinzon ZS, Prada MC. 1999. Recovery of tropical mangrove forests following a major oil spill: a study of recruitment and growth, and the benefits of planting. In: Yáñez-Arancibia A, Lara-Domínguez AL, editors. Ecosistemas de manglar en América Tropical (Mangrove ecosystems in tropical America). Silver Spring (MD): Instituto de Ecologia, Xalapa, Mexico. 231-254 p.

Dunbar JS, Thulman DK. 2019. Early paleoindian potentials on the continental shelf in the southeastern United States. In: Thulman DK, Garrison EG, editors. New directions in the search for the first Floridians. Gainesville (FL): University Press of Florida. 101-121 p.

Dupuis A, Ucan-Marin F. 2015. A literature review on the aquatic toxicology of petroleum oil: an overview of oil properties and effects to aquatic biota. Ottawa (ON): Fisheries and Oceans Canada, Winnipeg, Manitoba. 57 p.

DWH NRDA Trustees (Deepwater Horizon Natural Resource Damage Assessment Trustees). 2014. *Deepwater Horizon* oil spill: programmatic and phase III early restoration plan and early restoration programmatic environmental impact statement. NOS (National Ocean Service); DWH (Deepwater Horizon); NEPA (National Environmental Policy Act). 225 p.

DWH NRDA Trustees (Deepwater Horizon Natural Resource Damage Assessment Trustees). 2016. *Deepwater Horizon* oil spill: final programmatic damage assessment and restoration plan and final programmatic environmental impact statement. Silver Spring (MD): National Oceanic and Atmospheric Administration. 70 p.

Dyer CL, Gill DA, Picou JS. 1992. Social disruption and the Valdez oil spill: Alaskan Natives in a natural resource community. Sociological Spectrum. 12(2):105-126.

Edgar GJ, Kerrison L, Shepherd SA, Veronica Toral-Granda M. 2003a. Impacts of the *Jessica* oil spill on intertidal and shallow subtidal plants and animals. Marine Pollution Bulletin. 47(2003):276-283.

Edgar GJ, Marshall PA, Mooney P. 2003b. The effect of the Jessica grounding on Galápagos reef fish communities adjacent to the wreck site. Marine Pollution Bulletin. 47(7):296-302.

Edgar GJ, Snell H, Lougheed LW. 2002. Biological impacts of the Jessica oil spill on the Galapagos environment. Galapagos, Ecuador: Charles Darwin Foundation for the Galapagos Islands, Puerto Ayora. 6 p.

Eleuterius LN. 1970. Observations of *Claviceps purpurea* on *Spartina alterniflora* in the coastal marshes of Mississippi. Gulf Research Reports. 3(1):289-292.

Elliott-Smith E, Haig SM, Powers BM. 2009. Data from the 2006 international piping plover census: U.S. geological survey data series 426. Reston (VA): US Geological Survey. 340 p.

Engelhardt FR. 1983. Petroleum effects on marine mammals. Aquatic Toxicology. 4(3):199-217.

Engelhardt FR, Geraci JR, Smith TG. 1977. Uptake and clearance of petroleum hydrocarbons in the ringed seal, *Phoca hispida*. Journal of the Fisheries Research Board of Canada. 34(8):1143-1147.

English E. 2010. Appendix J: damage estimate for shoreline recreation. Boulder (CO): Prepared by Stratus Consulting Inc., and prepared for Cosco Busan Natural Resource Damage Assessment. 55 p.

English E, McConnell K. 2015. DWH lost recreational use NRDA technical working group report: overview of recreation assessment. 11 p.

EPA (U.S. Environmental Protection Agency). 1973. Oil spill, Long Island Sound, March 21,1972. Environmental effects. Final report. Washington (DC). 135 p.

EPA (U.S. Environmental Protection Agency). 2018a. National list of beaches: interactive version. 5 p.

EPA (U.S. Environmental Protection Agency). 2018b. Beaches NHDPlus indexed dataset. 8 p.
Eppley Z. 1992. Assessing indirect effects of oil in the presence of natural variation: the problem of reproductive failure in South Polar skuas during the Bahia Paraiso oil spill. Marine Pollution Bulletin. 25(9-12):307-312.
Evans AM, Keith ME, Voisin EE, Hesp PA, Cook GD, Allison MA, de Silva GM, Swanson EA. 2013. Archaeological analysis of submerged sites on the Gulf of Mexico continental shelf. New Orleans (LA): U.S. Department of the Interior, Bureau of Ocean Energy Management. Gulf of Mexico OCS Region. OCS Study BOEM 2013-011110.
Evers DC, Sperduto M, Gray CE, Paruk JD, Taylor KM. 2019. Restoration of common loons following the North Cape Oil Spill, Rhode Island, USA. Science of The Total Environment. 695:37.
Falkowski PG, Barber RT, Smetacek V. 1998. Biogeochemical controls and feedbacks on ocean primary production. Science. 281:200-206.
Faught MK. 2004. The underwater archaeology of paleolandscapes, Apalachee Bay, Florida. American Antiquity. 69(2):275-289.
Feld S, Basso KH. 1996. Senses of place. Santa Fe (NM): School of American Research Press. 293 p.
Field CB, Behrenfeld MJ, Randerson JT, Falkowski P. 1998. Primary production of the biosphere: integrating terrestrial and oceanic components. Science. 281(5374):237-240.
Field ME, Cochran SA, Evans KR. 2002. US coral reefs; imperiled national treasures. https://pubs.usgs.gov/fs/2002/fs025-02/images/blkdgrm.gif: USGS. block diagram showing the environment in which colonial hard corals form p.
Fingas M, Brown CE. 2018. A review of oil spill remote sensing. Sensors. 18:1-18.
Fingas MF. 1995. The evaporation of oil spills: development and implementation of new prediction methodology. Proceedings of the 18th Arctic and Marine Oilspill Program Technical Seminar; Ottawa (ON). Environment Canada. 43-60 p.
Finley JHW, Hanifen JG, Roscigno PF, Debusschere K, Newell MJ, Julius BE, von Beek J. 1995. A cooperative damage assessment: lessons from the *Greenhill* spill. Proceedings of the International Oil Spill Conference. American Petroleum Institute. 355-358 p.
FL DEP (Florida Department of Environmental Protection). 2013. Outdoor recreation in Florida. Tallahassee (FL): State of Florida Division of Recreation and Parks.
FL DEP (Florida Department of Environmental Protection). 2016. Florida statewide comprehensive outdoor recreation plan participation study 2016-2017 - Final report. Institute for Service Research (ISR). 297 p.
Fodrie FJ, Able KW, Galvez F, Heck KL, Jensen OP, López-Duarte PC, Martin CW, Turner RE, Whitehead A. 2014. Integrating organismal and population responses of estuarine fishes in Macondo spill research. BioScience. 64(9):778-788.
Fonseca M, Piniak GA, Cosentino-Manning N. 2016. Susceptibility of seagrass to oil spills: a case study with eelgrass, *Zostera marina* in San Francisco Bay, USA. Marine Pollution Bulletin. 115(1-2):29-38.
Ford R, Bonnell M. 1998. Final report bird injury assessment for the Torch/Platform Irene pipeline oil spill, September 1997. Portland (OR): Prepared for California Department of Fish and Game Office of Spill Prevention and Response. 45 p.
Foster MS, De Vogelaere AP, Harold C, Pearse JS, Thum AB. 1988. Causes of spatial and temporal patterns in rocky intertidal communities of central and northern California. Memoirs of the California Academy of Science Number 9. San Francisco (CA): California Academy of Sciences. 50 p.
Frazier J. 2005. Marine turtles: the role of flagship species in interactions between people and the sea. Mast. 4(1):5-38.
French-McCay D. 2004. Oil spill impact modeling: development and validation. Environmental Toxicology and Chemistry. 23(10):2441-2456.

French-McCay D, Horn M, Li Z, Jayko K, Spaulding M, Crowley D, Mendelsohn D. 2018. Modeling distributions, fate, and concentrations of Deepwater Horizon oil in subsurface waters of the Gulf of Mexico. In: Stout SA, Wang Z, editors. Case studies in oil spill environmental forensics. Oxford (GB): Butterworth-Heinemann: Elsevier. 683-736 p.

Fritts TH, McGehee MA, (U.S. Fish and Wildlife Service Belle Chasse LA). 1982. Effects of petroleum on the development and survival of marine turtle embryos. New Orleans (LA): U.S. Department of the Interior, Minerals Management Service. Contract No.: 14-16-0009-80-946. Report No. FWS/Obs-82/37.

Fry D, Addiego L, Grau C, (Nero & Associates Portland OR). 1987. Seabird oil toxicity study. Cabrillo (CA): US Department of the Interior, Minerals Management Service. Report No.: OCS Study MMS. 185 p.

Garrison EG, Giammona CP, Kelly FJ, Tripp AR, Wolff GA, (Texas A&M Research Foundation College Station TX). 1989. Historic shipwrecks and magnetic anomalies of the northern Gulf of Mexico. New Orleans (LA): U.S. Dept. of the Interior, Minerals Management Service. MMS 89-0023, MMS 89-0024, MMS 89-0025.

Garrison EG, Thulman DK. 2019. The state of Florida paleoindian studies: early paleoindian potentials on the continental shelf. In: Thulman DK, Garrison EG, editors. New directions in the search for the first Floridians. Gainesville (FL): University Press of Florida. 276-282 p.

Garshelis DL, Johnson CB. 2013. Prolonged recovery of sea otters from the *Exxon Valdez* oil spill? A re-examination of the evidence. Marine Pollution Bulletin. 71(1-2):7-19.

Gelin A, Gravez V, Edgar GJ. 2003. Assessment of *Jessica* oil spill impacts on intertidal invertebrate communities. Marine Pollution Bulletin. 46(2003):1377-1384.

Geraci JR. 1990. Physiologic and toxic effects on cetaceans. In: Geraci JR, St Aubin DJ, editors. Sea mammals and oil: confronting the risks. New York (NY): Academic Press, Inc. 211-122 p.

Geraci JR, St Aubin DJ. 1980. Offshore petroleum resource development and marine mammals: a review and research recommendations. Marine Fisheries Review. (November) 42:1-12.

Geraci JR, St Aubin DJ. 1990. Sea mammals and oil: confronting the risks. New York (NY): Academic Press, Inc. 282 p.

Geraci JR, St. Aubin DJ, editors (Battelle Memorial Institute V, CA). 1988. Synthesis of effects of oil on marine mammals. Sterling (VA): Minerals Management Service: MMS 88-0049. 314 p.

Gesteira JLG, Dauvin JC. 2000. Amphipods are good bioindicators of the impact of oil spills on soft-bottom macrobenthic communities. Marine Pollution Bulletin. 40(11):1017-1027.

Gill DA. 1994. Environmental disaster and fishery co-management in a natural resource community: impacts of the Exxon Valdez oil spill. In: Dyer CL, McGoodwin JR, editors. Folk management in the world's fisheries: implications for fisheries managers. Boulder (CO): University of Colorado Press. 207-235 p.

Gill DA, Picou JS. 1997. The day the water died: cultural impacts of the *Exxon Valdez o*il spill. In: Picou JS, Gill DA, Cohen M, editors. The *Exxon Valdez* disaster: readings on a modern social problem. Dubuque (IA): Kendall-Hunt. 167-191 p.

Gill SD, Bonke CA, Carter J. 1985. Management of the Uniacke G-72 incident. Proceedings of the 1985 International Oil Spill Conference; Los Angeles (CA). American Petroleum Institute. 311-313 p.

Glémarec M, Hussenot E. 1982. A three-year ecological survey in Benoit and Wrac'h Abers following the Amoco Cadiz oil spill. Netherlands Journal of Sea Research. 16:483-490.

Godard-Codding CAJ, Collier T. 2018. Chapter 3. The toxicological effects of oil exposure on cetaceans. In: Fossi MC, Panti C, editors. Marine mammal ecotoxicology: impacts of multiple stressors on population health. Academic Press. 75-93 p.

Golet GH, Seiser PE, McGuire AD, Roby DD, Fischer JB, Kuletz KJ, Irons DB, Dean TA, Jewett SC, Newman SH. 2002. Long-term direct and indirect effects of the "*Exxon Valdez*" oil spill on

pigeon guillemots in Prince William Sound, Alaska. Marine Ecology Progress Series. 241:287-304.
Gomez Gesteira JL, Dauvin JC. 2005. Impact of the Aegean Sea oil spill on the subtidal fine sand macrobenthic community of the Ares-Betanzos Ria (Northwest Spain). Marine Environmental Research. 60(3):289-316.
Gomez Gesteira JL, Dauvin JC, Salvande Fraga M. 2003. Taxonomic level for assessing oil spill effects on soft-bottom sublittoral benthic communities. Marine Pollution Bulletin. 46(5):562-572.
Goodbody-Gringley G, Wetzel DL, Gillon D, Pulster E, Miller A, Ritchie KB. 2013. Toxicity of Deepwater Horizon source oil and the chemical dispersant, Corexit(R) 9500, to coral larvae. PLoS ONE. 8(1):e45574.
Gornik K, Lin T, McDonald G, Ng N, Quigley C, Viana D. 2013. The non-market value of private recreational boating in the Channel Islands National Marine Sanctuary. Santa Barbara (CA): Bren School of Environmental Science and Management, University of Santa Barbara. 48 p.
Graham MH, Dayton PK, Erlandson JM. 2003. Ice ages and ecological transitions on temperate coasts. Trends in Ecology & Evolution. 18(1):33-40.
Graham WM, Condon RH, Carmichael RH, D'Ambra I, Patterson HK, Linn LJ, Hernandez Jr FJ. 2010. Oil carbon entered the coastal planktonic food web during the *Deepwater Horizon* oil spill. Environmental Research Letters. 5(4):1-6.
Gregory D, Jensen P, Strætkvern K. 2012. Conservation and in situ preservation of wooden shipwrecks from marine environments. Journal of Cultural Heritage. 13S:S139-S148.
Gu J-D, Ford TE, Berke NS, Mitchell R. 1998. Biodeterioration of concrete by the fungus *fusarium*. International Biodeterioration & Biodegradation. 41:101-109.
Gustitus SA, Clement TP. 2017. Formation, fate, and impacts of microscopic and macroscopic oil-sediment residues in nearshore marine environments: a critical review. Reviews of Geophysics. 55(4):1130-1157.
Guzmán HM, Burns KA, Jackson JB. 1994. Injury, regeneration and growth of Caribbean reef corals after a major oil spill in Panama. Marine Ecology Progress Series. 105:231-241.
Haab TC, Hicks RL, Whitehead JC. 2006. The economic value of marine recreational fishing: analysis of the MRFSS 1998 Pacific add-on. Proceedings of the Fisheries, Fishermen, and Fishing Communities: Socioeconomic Perspectives on the West Coast, Alaska and Hawaii Symposium 2005 American Fisheries Society Annual Meeting. Anchorage (AK).
Haab TC, Whitehead JC, Mcconnell T. 2000. The economic value of marine recreational fishing in the southeast United States. 1997 economic data analysis. https://www.st.nmfs.noaa.gov/st5/RecEcon/Publications/SE_vol2.pdf. 102 p.
Haggarty JC, Wooley CB, Erlandson JM, Crowell A. 1991. The 1990 Exxon cultural resource program: site protection and maritime cultural ecology in Prince William Sound and the Gulf of Alaska. Anchorage (AK). 358 p.
Hamdan L, Salerno JL, Reed A, Joye SB, Damour M. 2018. The impact of the Deepwater Horizon blowout on historic shipwreck-associated sediment microbiomes in the northern Gulf of Mexico. Scientific Reports. 8:9057.
Hamel C, Hermann M, Lee ST, Criddle KR. 2000. An economic discussion of the marine sport fisheries in central and lower Cook Inlet. Proceedings of the Tenth Biennial Conference of the International Institute of Fisheries Economics & Trade. Corvallis (OR). 1-10 p.
Hampson G. 2000. Destruction and recovery of the Winsor Cove, Cataumet MA salt marsh from a #2 fuel oil spill: a 25 year history. Environment Cape Cod. 3(1):32-39.
Hampson GR, Moul ET. 1978. No. 2 fuel oil spill in Bourne, Massachusetts: immediate assessment of the effects on marine invertebrates and a 3-year study of growth and recovery of a salt marsh. Journal of the Fisheries Research Board of Canada. 35(5):731-744.

Harr K, Rishniw M, Rupp T, Cacela D, Dean K, Dorr B, Hanson-Dorr K, Healy K, Horak K, Link J. 2017. Dermal exposure to weathered MC252 crude oil results in echocardiographically identifiable systolic myocardial dysfunction in double-crested cormorants (*Phalacrocorax auritus*). Ecotoxicology and Environmental Safety. 146:76-82.
Hartmann AC, Sandin SA, Chamberland VF, Marhaver KL, de Goeij JM, Vermeij MJA. 2015. Crude oil contamination interrupts settlement of coral larvae after direct exposure ends. Marine Ecology Progress Series. 536:163-173.
Hartung R. 1963. Ingestion of oil by waterfowl. Papers of the Michigan Academy of Science Arts and Letters. 48:49-55.
Hausman JA, Leonard GK, McFadden D. 1995. A utility-consistent, combined discrete choice and count data model: assessing recreational use losses due to natural resource damage. Journal of Public Economics. 56:1-30.
Haycock R, Baines M, Earl S. 1998. Effects of the *Sea Empress* oil spill on breeding seabirds in South West Wales. In: Edwards R, Sime H, editors. Proceedings of the International Oil Spill Conference; Cardiff (UK). The Chartered Institution of Water and Environmental Management. 227-246 p.
Hayes MO, Michel J. 2001. A primer for response to oil spills on gravel beaches. Proceedings of the International Oil Spill Conference Global Strategies for Prevention, Preparedness, Response, and Restoration. American Petroleum Institute; Tampa (FL). 1275-1279 p.
Hays GC, Ashworth JS, Barnsley MJ, Broderick AC, Emery DR, Godley BJ, Henwood A, Jones EL. 2001. The importance of sand albedo for the thermal conditions on sea turtle nesting beaches. Oikos. 93(1):87-94.
Hays GC, Hawkes JA. 2018. Satellite tracking sea turtles: opportunities and challenges to address key questions. Frontiers in Marine Science. 5(432):1-12.
Helm RC, Carter HR, Ford RG, Fry DM, Moreno RL, Sanpera C, Tseng FS. 2015. Overview of efforts to document and reduce impacts of oil spills on seabirds. In: Fingas M, editor. Chapter 17 Handbook of oil spill science and technology. New York (NY): John Wiley & Sons. 429-453 p.
Helton D. 2003. Damage assessment and restoration planning for the *M/V Kuroshima* oil spill. Proceedings of the 2003 International Oil Spill Conference. Vancouver (CA). 219-223 p.
Henry C, Helton D, Michel J, Woodle C. 2008. Bayou Perot and the unusual situation of stranded oil adhered to mud flats. Proceedings of the International Oil Spill Conference. American Petroleum Institute; Savannah International Trade & Convention Center, Savannah (GA). 551-556 p.
Hester M, Mendelssohn I. 2000. Long-term recovery of a Louisiana brackish marsh plant community from oil-spill impact: vegetation response and mitigating effects of marsh surface elevation. Marine Environmental Research. 49(3):233-254.
Hicken CE, Linbo TL, Baldwin DH, Willis ML, Myers MS, Holland L, Larsen M, Stekoll MS, Rice SD, Collier TK, et al. 2011. Sublethal exposure to crude oil during embryonic development alters cardiac morphology and reduces aerobic capacity in adult fish. National Academy of Sciences. 108(17):7086-7090.
Hinga K. 2003. Degradation rates of low molecular weight PAH correlate with sediment TOC in marine subtidal sediments. Marine Pollution Bulletin. 46:466-474.
Hirvi J. 1990. Summary of the environmental effects of the *Antonio Gramsci* oil spill in the Gulf of Finland in 1987. Proceedings of the Thirteenth Arctic and Marine Oil Spill Program Technical Seminar; Ottawa (ON). Environment Canada. 17 p.
Ho K, Patton L, Latimer JS, Pruell RJ, Pelletier M, McKinney R, Jayaraman S. 1999. The chemistry of toxicity of sediment affected by oil from the *North Cape* spilled into Rhode Island Sound. Marine Pollution Bulletin. 38(4):314-323.

Hoff RZ. 1995. Responding to oil spills in coastal marshes: the fine line between help and hindrance. Seattle (WA): National Oceanic and Atmospheric Administration, Hazardous Material Response and Assessment Division. 17 p.

Hoff RZ, Michel J, Hensel P, Proffitt EC, Delgado P, Shigenaka G, Yender R, Mearns AJ. 2014. Oil spills in mangroves: planning & response considerations. Seattle (WA): U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration. 73 p.

Hoff RZ, Shigenaka G, Henry Jr. CB. 1993. Salt marsh recovery from a crude oil spill: vegetation, oil weathering, and response. Proceedings of the International Oil Spill Conference. American Petroleum Institute; Tampa (FL). 307-311 p.

Horn M, French-McCay D. 2014. Shelburne Basin venture exploration drilling project appendix G: spill fate and behaviour modelling. Trajectory modelling in support of the Shelburne Basin exploration drilling program. Model description, approach, and summary of results. South Kingstown (RI): RPS ASA. 6 p.

Hughes B, Steward B, Brown M, Hearn R, Cranswick P, B H, Bullock I. 1998. The effect of oiling and body condition on the diet of common scoters *Melanitta nigra nigra* in Carmarthen Bay following the *Sea Empress* oil spill. The Chartered Institution of Water and Environmental Management; Cardiff (UK). 247-267 p.

Hughes JB. 1999. Cytological-cytogenetic analyses of winter flounder embryos collected from the benthos at the barge North Cape oil spill. Marine Pollution Bulletin. 38(1):30-35.

Hunt D. 2012. Properties of wood in the conservation of historical wooden artifacts. Journal of Cultural Heritage. 13S:S10-S15.

Hunt H, Vavrek M, McCurdy M. 2002. Invertebrate indicators of onshore oil spill severity. Louisiana Applied Oil Spill Research and Development Program, OSRADP Technical Report Series. 26 p.

Hurst RJ, Watts PD, Oritsland NA. 1991. Metabolic compensation in oil-exposed polar bears. Journal of Thermal Biology. 16(1):53-56.

IAI (Impact Assessment Inc). 2011. Critical human dimensions of maritime oil spills as identified through examination of the Selendang Ayu incident. La Jolla (CA). OCS Study BOEM 053-2011. 95 p.

IAI (Impact Assessment Inc), Russell JC, Downs MA, Strick BR, Galginaitis MS. 1998. Final report: *Exxon Valdez o*il spill, cleanup, and litigation: a collection of social-impacts information and analysis. Anchorage (AK): U.S. Department of the Interior, Minerals Management Service, Alaska OCS Region. OCS Study MMS 99-0007 Technical Report No. 161 AK-96-01. 97 p.

ICF International, Southeastern Archaeological Research, Davis Geoarchaeological Research. 2013. Inventory and analysis of coastal and submerged archaeological site occurrence on the Pacific Outer Continental Shelf. Camarilla (CA): U.S. Department of the Interior, Bureau of Ocean Energy Management, Pacific OCS Region. OCS Study BOEM 2013-0115.

IEc (Industrial Economics Inc). 2018. Forecasting environmental and social externalities associated with outer continental shelf (OCS) oil and gas development, volume 2: supplemental information to the 2018 revised offshore environmental cost model (OECM). Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM 2018-067.

IEc (Industrial Economics Inc). 2019. Forecasting environmental and social externalities associated with outer continental shelf (OCS) oil and gas development, volume 1: 2018 revised offshore environmental cost model (OECM). Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM 2018-066.

Incardona JP, Collier TK, Scholz NL. 2011. Oil spills and fish health: exposing the heart of the matter. J Expo Sci Environ Epidemiol. 21(1):3-4.

Incardona JP, Vines CA, Anulacion BF, Baldwin DH, Day HL, French BL, Labenia JS, Linbo TL, Myers MS, Olson OP, et al. 2012a. Unexpectedly high mortality in Pacific herring embryos exposed to

the 2007 Cosco Busan oil spill in San Francisco Bay. National Academy of Sciences. 109(2):E51-E58.
Incardona JP, Vines CA, Linbo TL, Myers MS, Sloan CA, Anulacion BF, Boyd D, Collier TK, Morgan S, Cherr GN, et al. 2012b. Potent phototoxicity of marine bunker oil to translucent herring embryos after prolonged weathering. PLoS ONE. 7(2):e30116.
Incident Analysis Team. 2000. Response to the *Laura d'Amato* oil spill. (AU): Incident Analysis Team. 44 p.
IPIECA (International Petroleum Industry Environmental Conservation Association). 2005. Biological impacts of oil pollution: rocky shores. London (UK). 25 p.
ITOPF (International Tanker Owners Pollution Federation Ltd). 2014. Technical information papers (TIPs). London (UK). 229 p.
ITOPF (International Tanker Owners Pollution Federation Ltd). 2018. Oil spill modelling. https://www.itopf.org/knowledge-resources/documents-guides/fate-of-oil-spills/oil-spill-modelling/.
Jacobs RPWM. 1980. Effects of the 'Amoco Cadiz' oil spill on the seagrass community at Roscoff with special reference to the benthic infauna. Marine Ecology Progress Series. 2(3):207-212.
Jariwala V, Striegel M. 2020. Removal of crude oil from cultural resources. Proceedings of the 6-ISCs Joint Meeting. Virtual.
Jenssen BM, Ekker M. 1991. Dose dependent effects of plumage-oiling on thermoregulation of common eiders *Somateria mollissima* residing in water. Polar Research. 10(2):579-584.
Jesperson M, Griffin K. 1992. An evaluation of archaeological injury documentation, *Exxon Valdez o*il spill. Anchorage (AK): Office of History and Archaeology.
Johnston RJ, Grigalunas TA, Opaluch JJ, Mazzotta M, Diamantedes J. 2002. Valuing estuarine resource services using economic and ecological models: the peconic estuary system study. Coastal Management. 30(1):47-65.
Jones AR. 2003. Ecological recovery of amphipods on sandy beaches pollution: an interim assessment. Journal of Coastal Research. SI35:66-73.
Jones CG, Lawton JH, Shachak M. 1997. Positive and negative effects of organisms as physical ecosystem engineers. Ecology. 78(7):1946-1957.
Jorgenson MT. 1997. Effects of petroleum spills on tundra ecosystems. In Proceedings: Science, Traditional Knowledge, and the Resources of the Northeast Planning Area of the National Petroleum Reserve in Alaska. April 16-18, 1997, Anchorage (AK). OCS Report MMS 97-0013. Anchorage, AK: USDOI, MMS, and BLM. 99 pp plus attachments and the bibliography: Petroleum spills on tundra therein. 151 p.
Kennicutt MC, Sweet ST. 1992. Hydrocarbon contamination on the Antarctic peninsula: III. The Bahia Paraiso—two years after the spill. Marine Pollution Bulletin. 25(9-12):303-306.
Kennicutt MC, Sweet ST, Fraser WR, Culver M, Stockton WL. 1991a. The fate of diesel fuel spilled by the *Bahia Paraiso* in Arthur Harbor, Antarctica. Proceedings of the 1991 International Oil Spill Conference; San Diego (CA). American Petroleum Institute. 493-500 p.
Kennicutt MC, Sweet ST, Fraser WR, Stockton WL, Culver M. 1991b. Grounding of the *Bahia Paraiso* at Arthur Harbor, Antarctica. 1. Distribution and fate of oil spill related hydrocarbons. Environmental Science & Technology. 25(3):509-518.
Kenworthy J, Hall MO, Hammerstrom KK, Merellow M, Schwartzschild A. 2018. Restoration of tropical seagrass beds using wild bird fertilization and sediment regrading. Ecological Engineering. 112:72-81.
Kenworthy WJ, Cosentino-Manning N, Handley L, Wild M, Rouhani S. 2017. Seagrass response following exposure to *Deepwater Horizon* oil in the Chandeleur Islands, Louisiana (USA). Marine Ecology Progress Series. 576:145-161.

Kenworthy WJ, Hammerstrom K, Fonseca MS. 2006. Scientific evaluation of a sediment fill technique for the restoration of motor vessel injuries in seagrass beds of the Florida Keys National Marine Sanctuary. Beaufort (NC): NOAA, NOS, NCCOS, and Center for Coastal Fisheries and Habitat Research.

Kinako P. 1981. Short-term effects of oil pollution on species numbers and productivity of a simple terrestrial ecosystem. Environmental Pollution Series A, Ecological and Biological. 26:87-91.

Kirsch S. 2001. Lost worlds: environmental disaster, "culture loss," and the law. Current Anthropology. 42(2):167-198.

Kittle Jr. LJ, Burge R, Butler R, Cook W, Fresh K, Kyte M, Lowell R, Pease R, Penttila D, Scholz A, et al. 1987. Marine resource damage assessment report for the *Arco Anchorage* oil spill December 21, 1985 into Port Angeles Harbor and the Strait of Juan de Fuca. Washington State (WA): Washington State Department of Ecology. 386 p.

Klasing SA, Brodberg RK, Smith W. 2015. Risk assessment of seafood consumption following the Refugio Beach oil spill incident in Santa Barbara County, California. Sacramento (CA): Fish, Ecotoxicology, and Water Section Pesticide and Environmental Toxicology Branch, Office of Environmental Health Hazard Assessment California Environmental Protection Agency. 44 p.

Knecht R, Davis R. 1999. Oil spill response and restoration at the Summer Bay archaeological site (UNL-92). 28 p.

Kohout J, Meade N. 2008. Preassessment data report #12: potential human use losses associated with the *Selendang Ayu* oil spill. U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration. 7 p.

Krebs CT, Burns KA. 1977. Long-term effects of an oil spill on populations of the salt-marsh crab *Uca pugnax*. Science. 197(4302):484-487.

Kruczynski WL, Fletcher PJ. 2012. Tropical connections South Florida's marine environment. IAN Press, University of Maryland Center for Environmental Science.

Kubitz JA, Fichera M, Markarian R, Slocomb J. 2011. Use of chemical/oil spill impact module (COSIM) to assess the toxicity of petroleum to estuarine organisms. Poster Presented at the 2011 International Oil Spill Conference. Washington (DC): American Petroleum Institute.

Lamparelli C, Rodrigues F, deMoura D. 1997. Long-term assessment of an oil spill in a mangrove forest in São Paulo, Brazil. Mangrove Ecosystem Studies in Latin America and Africa. Paris (FR): UNESCO. 191-203 p.

Landry CE, McConnell KE. 2007. Hedonic onsight cost model of recreation demand. Land Economics. 83(2):253-267.

Lauritsen AM, Dixon PM, Cacela D, Brost B, Hardy R, MacPherson SL, Meylan A, Wallace BP, Witherington B. 2017. Impact of the Deepwater Horizon oil spill on loggerhead turtle *Caretta caretta* nest densities in northwest Florida. Endangered Species Research. 33:83-93.

Leahy JG, Colwell RR. 1990. Microbial degradation of hydrocarbons in the environment. Microbiology and Molecular Biology Reviews. 54(3):305-315.

Lee K, Boufadel M, Bing C, Foght J, Hodson P, Swanson S, Venosa A. 2015. Expert panel report on the behaviour and environmental impacts of crude oil released into aqueous environments. Ottawa (ON): The Royal Society of Canada. 461 p.

Lee RF. 1977. Fate of oil in the sea. Proceedings of the 1977 Oil Spill Response Workshop FWS/OBS-77/24; Washington (DC). U.S. Fish and Wildlife Service, Office of Biological Services. 43-54 p.

Lee RF, Koster M, Paffenhofer GA. 2012. Ingestion and defecation of dispersed oil droplets by pelagic tunicates. Journal of Plankton Research. 34(12):1058-1063.

Leggett CG, Scherer N, Curry MS, Bailey R, Haab TC. 2014. Assessing the economic benefits of reductions in marine debris: a pilot study of beach recreation in Orange County, California. Cambridge (MA): Industrial Economics, Inc.

Lei YL, Li TG, Bi H, Cui WL, Song WP, Li JY, Li CC. 2015. Responses of benthic foraminifera to the 2011 oil spill in the Bohai Sea, PR China. Marine Pollution Bulletin. 96(1-2):245-260.
Leighton FA. 1993. The toxicity of petroleum oils to birds. Environmental Reviews. 1(2):92-103.
Leighton FA, Peakall DB, Butler RG. 1983. Heinz-Body hemolytic anemia from the ingestion of crude oil: a primary toxic effect in marine birds. Science. 220:871-873.
Levine BM, White JR, DeLaune RD. 2017. Impacts of the long-term presence of buried crude oil on salt marsh soil denitrification in Barataria Bay, Louisiana. Ecological Engineering. 99:454-461.
Lew DK, Larson DM. 2008. Valuing a beach day with a repeated nested logit model of participation, site choice, and stochastic time value. Marine Resource Economics. 23:233-252.
Lewis RR. 1983. Impact of oil spills on mangrove forests. In: Teas H, editor. Biology and ecology of mangroves Tasks for vegetation science 8. Tampa (FL): Springer, Dordrecht. 171-183 p.
Lim SSL, Yong AYP. 2015. Rapid population recovery of *Ocypode ceratophthalmus* (Pallas, 1772) (Crustacea: Brachyura: Ocypodidae) after an oil spill in Singapore. Conservation & Ecology. 63:270-278.
Linden O, Mattsson J, Notini M. 1983. A spill of light fuel oil in the Baltic Sea. Proceedings of the 1983 International Oil Spill Conference; San Antonio (TX). American Petroleum Institute. 517-520 p.
Lindstedt-Siva J, Chamberlain DW, Mancini ER. 1987. Environmental aspects of the ARCO Anchorage oil spill, Port Angeles, Washington. Proceedings of the International Oil Spill Conference; Baltimore (MD). American Petroleum Institute. 407-410 p.
Linkins AE, Johnson LA, Everett KR, Atlas RM. 1984. Oil spills: damage and recovery in Tundra and Taiga. In: Cairns J, Jr., Buikema A, Jr., editors. Restoration of habitats impacted by oil spills. Stoneham (MA): Butterworth-Heinemann. 10 p.
Little A, Little D, Morgan C. 1990. Ecological impact of the '*El Omar*' oil spill, Milford Haven December 1988. Field Studies Council Research Centre. Report to West of England Shipowners Insurance Services Ltd. Report No. FSC/RC/38/89. 119 p.
Little C, Kitching JA. 1996. The biology of rocky shores. New York (NY): Oxford University Press. 240 p.
Little DI, Little AE. 1991. Estuarine oil spill effects in the context of dispersant use changes. Proceedings of the 1991 International Oil Spill Conference; San Diego (CA). American Petroleum Institute. 507-518 p.
Locke C, White M, Michel J, Henry C, Sellars JD, Aslaksen Jr ML. 2008. Use of vertical digital photography at the Bayou Perot, LA spill for oil mapping and volume estimation. Proceedings of the International Oil Spill Conference. American Petroleum Institute; Savannah (GA). 127-130 p.
Loehefener RR, Hoggard W, Roden CL, Mullin KD, Rogers CM. 1989. Petroleum structures and the distribution of sea turtles. Proceedings of the Spring Ternary Gulf of Mexico Studies Meeting; New Orleans (LA). Minerals Management of the Service, U.S. Department of the Interior.
Lopes CF, Milanelli JCC, Prosperi VA, Zanardi E, Truzzi AC. 1997. Coastal monitoring program of São Sebastião channel: assessing the effects of 'Tebar V' oil spill on rocky shore populations. Marine Pollution Bulletin. 34(11):923-927.
Lorentz WP, Hall J, Finley H, Hanifen J, Hamilton D, Pace L, Penn T, Kern J, Goatcher B, Markarian R. 2001. The Lake Barre oil spill NRDA: from response to restoration. Proceedings of the International Oil Spill Conference. American Petroleum Institute. 667-670 p.
LOSC (Louisiana Oil Spill Coordinator), Louisiana Department of Environmental Quality, Louisiana Department of Natural Resources, Louisiana Department of Wildlife and Fisheries, National Oceanic and Atmospheric Administration, United States Fish and Wildlife Service. 2005. Final damage assessment and restoration plan and environmental assessment Equinox oil company crude oil discharge, Lake Grande Ecaille, Louisiana September 22, 1998. 98 p.
Lougheed LW, Edgar GJ, Snell HL. 2002. Biological impacts of the Jessica oil spill on the Galapagos environment. Puerto Ayora (EC): Final Report v.1.10. Charles Darwin Foundation. 134 p.

Loughlin TR, Ballachey BE, Wright BA. 1996. Overview of studies to determine injury caused by the *Exxon Valdez* oil spill to marine mammals. In: Rice C, Spies R, Wolfe D, Wright B, editors. Exxon Valdez Oil Spill Symposium, 2-5 February 1993, Anchorage, AK. Bethesda (MD): American Fisheries Society. 798-808 p.

Louis Berger & Associates Inc. 1991. Arthur Kill oil discharge study: final report. East Orange (NJ): Louis Berger & Associates Inc. Technical Report. 56 p.

Lugo AE, Snedaker SC. 1974. The ecology of mangroves. Annual Review of Ecology and Systematics. 5(1):39-64.

Lutcavage ME, Lutz PL, Bossart GD, Hudson DM. 1995. Physiologic and clinicopathologic effects of crude oil on loggerhead sea turtles. Archives of Environmental Contamination and Toxicology. 28:417-422.

Malcolm HM, Shore RF. 2003. Effects of PAHs on terrestrial and freshwater birds, mammals, and amphibians. In: Douben P, editor. PAHs: an ecotoxicological perspective. Wiley. 225-242 p.

Mancini ER, Lindstedt-Siva J, Chamberlain DW. 1989. Environmental impacts of the 1985 *Arco Anchorage* oil spill: 1988 conclusions. Proceedings of the International Oil Spill Conference (IOSC). American Petroleum Institute; San Antonio (TX). 459-462 p.

Marshall PA, Edgar GJ. 2003. The effect of the Jessica grounding on subtidal invertebrate and plant communities at the Galápagos wreck site. Marine Pollution Bulletin. 47(7-8):284-295.

Mathewson PD. 2018. Experimental and modeled thermoregulatory costs of repeated sublethal oil exposure in the double-crested cormorant, *Phalacrocorax auritus*. Marine Pollution Bulletin. 135:216-233.

Mauseth GS, Csulak FG. 2003. Damage assessment and restoration following the JULIE N oil spill: a case study. Proceedings of the International Oil Spill Conference. American Petroleum Institute; Vancouver, British Columbia (CN). 409-412 p.

Mauseth GS, Svarny-Livingston S, Arnold SM, Erickson GM. 2008. The M/V *Selendang Ayu* oil spill subsistence seafood sampling program and public health evaluation process. Proceedings of the International Oil Spill Conference. American Petroleum Institute. 1185-1191 p.

McConnell K, Strand IE. 1994. The economic value of Mid and South Atlantic sportfishing.

McDonald SJ, Kennicutt MC, Brooks JM. 1992. Evidence of polycyclic aromatic hydrocarbon (PAH) exposure in fish from the Antarctic peninsula. Marine Pollution Bulletin. 25(9):313-317.

McFarlin K, Prince R, Perkins R, Leigh M. 2014. Biodegradation of dispersed oil in Arctic seawater at -1 degrees C. PLoS One. 9(1):e84297.

McFarlin KM, Perkins MJ, Field JA, Leigh MB. 2018. Biodegradation of crude oil and Corexit 9500 in arctic seawater. Frontiers in Microbiology. 9(1788):14.

McIntyre AD. 1982. Oil pollution and fisheries. Philosophical Transactions of the Royal Society of London B, Biological Sciences. 297(1087):401-411.

McKee KL. 1995. Mangrove species distribution and propagule predation in Belize: an exception to the dominance-predation hypothesis. Biotropica. 27(3):334-345.

McLachlan A, Brown A. 2006. The ecology of sandy shores 2nd Edition. Burlington (MA): Academic Press. 392 p.

McLaren P. 1985. Behaviour of diesel fuel on a high energy beach. Marine Pollution Bulletin. 16:191-196.

McMillan C, Sherrod CL. 1986. The chilling tolerance of black mangrove, Avicennia germinans, from the Gulf of Mexico coast of Texas, Louisiana and Florida. Galveston (TX): Texas A&M University Galveston Campus. 9-16 p.

Meehan S. 2015. MC252 / *Deepwater Horizon* quantification of physical response injury to submerged aquatic vegetation. Silver Spring (MD): NOAA Office of Respone and Restoration. SAV Technical Working Group.

Meide C, McClean JA, Wiser E. 2001. Dog Island shipwreck survey 1999: report of historical and archaeological investigations. Tallahassee (FL): Florida State University.

Mendelssohn IA, Andersen GL, Baltz DM, Caffey RH, Carman KR, Fleeger JW, Joye SB, Lin Q, Maltby E, Overton EB, et al. 2012. Oil impacts on coastal wetlands: implications for the Mississippi River delta ecosystem after the *Deepwater Horizon* oil spill. BioScience. 62(6):562-574.

Michael AD, Van Raalte CR, Brown LS. 1975. Long-term effects of an oil spill at West Falmouth, Massachusetts. Proceedings of the International Oil Spill Conference. American Petroleum Institute. 573-582 p.

Michel J. 1996. Assessment of the extent of injury to salt pond marshes: summary report. Report to NOAA. 28 June. 22 p.

Michel J. 2021. Oil spill effects literature study of spills of greater than 20,000 barrels of crude oil, condensate, and diesel. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. OSC Study BOEM 2020-058. 326 p.

Michel J, Csulak F, French D, Sperduto M. 1997. Natural resource impacts from the North Cape oil spill. Proceedings of the International Oil Spill Conference; Fort Lauderdale (FL). American Petroleum Institute. 841-850 p.

Michel J, Fegley S, Dahlin J. 2015. *Deepwater Horizon* sand beach injury assessment technical report. Research Planning, Inc. Columbia (SC): Prepared for U.S. Fish and Wildlife Service, Atlanta (GA). 124 p.

Michel J, Fegley SR, Dahlin JA, Wood C. 2017. Oil spill response-related injuries on sand beaches: when shoreline treatment extends the impacts beyond the oil. Marine Ecology Progress Series. 576:203-218.

Michel J, Lehmann S, Henry C. 1998. Oiling cleanup issues: wetlands, M/T Julie N spill, Portland Maine. Seattle (WA): NOAA Technical Memorandum NOS ORCA 132. 34 p.

Michel J, Nixon Z, Dahlin J, Betenbaugh D, White M, Burton D, Turley S. 2009. Recovery of interior brackish marshes seven years after the Chalk Point oil spill. Marine Pollution Bulletin. 58:995-1006.

Michel J, Nixon Z, Hinkeldey H, Miles S. 2003. Use of in situ burning as an oil spill response tool: follow-up of four case studies. Proceedings of the 2003 International Oil Spill Conference. American Petroleum Institute; Washington (DC). 123-128 p.

Michel J, Rutherford N. 2013. Oil spills in marshes: planning and response considerations. Seattle (WA): Office of Response and Restoration, National Ocean Service, National Oceanic and Atmospheric Administration. 126 p.

Michel J, Zengel S. 2021. Good practices for in-situ burning of marshes based on two decades of responses in Louisiana. Proceedings of the International Oil Spill Conference. American Petroleum Institute; New Orleans (LA).

Miller JA. 1989. Physical and chemical recovery of intertidal and shallow subtidal sediments impacted by the ARCO Anchorage oil spill, Ediz Hook, Washington. Proceedings of the International Oil Spill Conference. Washington (DC). 6 p.

Miller RJ, Page HM, Reed DC. 2015. Trophic versus structural effects of a marine foundation species, giant kelp (*Macrocystis pyrifera*). Oecologia. 179(4):1199-1209.

Mitchelmore CL, Bishop CA, Collier TK. 2017. Toxicological estimation of mortality of oceanic sea turtles oiled during the Deepwater Horizon oil spill. Endangered Species Research. 33:39-50.

Mitra S, Kimmel DG, Snyder J, Scalise K, McGlaughon BD, Roman MR, Jahn GL, Pierson JJ, Brandt SB, Montoya JP, et al. 2012. Macondo-1 well oil-derived polycyclic aromatic hydrocarbons in mesozooplankton from the northern Gulf of Mexico. Geophysical Research Letters. 39(1):L01605.

Mitsch W, Gosselink J. 2015. Wetlands, 5th edition. Wiley. 456 p.

MMS (Minerals Management Service). 2005. Notice to lessees and operators of federal oil, gas, and sulphur leases and pipeline right-of-way holders in the outer continental shelf, Gulf of Mexico OCS region: archaeological resource surveys and reports. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.
Mobley CM, Haggarty JC, Utermohle CJ, Eldridge M, Reanier RE, Crowell A, Ream BA, Yesner DR, Erlandson JM, Buck PE. 1990. The 1989 *Exxon Valdez c*ultural resource program. Anchorage (AK): Exxon Shipping Company and Exxon Company, USA.
Montagna P, Brenner J, Gibeaut J, Morehead S. 2011. Chapter 4: coastal impacts. In: Schmandt J, North GR, Clarkson J, editors. The impact of global warming on Texas. Austin (TX): University of Texas Press. 26 p.
Moore SE. 2008. Marine mammals as ecosystem sentinels. Journal of Mammalogy. 89(3):534-540.
Moreno V, Iribarne O, Escalante A, Aizpun J, Janiot L, Heras H, Menone M, Miglioranza K, Pereze C, Isacch J, et al. 2004. Monitoring coastal *Rio de la Plata* oil spillage. Proceedings of the Interspill 2004. Trondheim (NO). Presentation No. 415. 20 p.
Mugge RL, Brock ML, Salerno JL, Damour M, Church R, Lee JS, Hamdan L. 2019. Deep-sea biofilms, historic shipwreck preservation and the *Deepwater Horizon* spill. Frontiers in Marine Science. 6.
Mulhare MJ, Therrien P-J. 1993. The natural degradation of #2 fuel oil in intertidal sand. In: Calabrese EJ, editor. Hydrocarbon contaminated soils and groundwater. Lewis Publishers. 467-478 p.
Mulhare MJ, Therrien P-J. 1997. Persistence of oil in beach sand following a catastrophic release of No. 2 fuel oil. Proceedings of the International Oil Spill Conference. American Petroleum Institute; Fort Lauderdale (FL). 409-413 p.
NASEM (National Academies of Sciences E, and Medicine),. 2020. The use of dispersants in marine oil spill response. Washington (DC): The National Academies Press. 340 p.
Neff JM, Hillman RE, Carr RS, Buhl RL, Lahey JI. 1987. Histopathologic and biochemical responses in Arctic marine bivalve molluscs exposed to experimentally spilled oil. Arctic. 40(Supp. 1):220-229.
Neff JM, Sharp MS, McCulloch WL. 1981. Impact of the Esso Bayway oil spill on salt marsh macro fauna. Proceedings of the 1981 International Oil Spill Conference. Atlanta (GA). 413-418 p.
Negri AP, Brinkman DL, Flores F, Botté ES, Jones RJ, Webster NS. 2016. Acute ecotoxicology of natural oil and gas condensate to coral reef larvae. Scientific Reports. 6:21153.
Nielsen J, Sandoval C, Donohoe R, Hampton S, Henkel L, Marek J. 2018. Western snowy plover reproductive effects associated with the Refugio Beach oil spill. Long Beach (CA): Poster for SETAC North America 39th Annual Meeting 2018.
Nikitik CCS, Robinson AW. 2003. Patterns in benthic populations in the Milford Haven waterway following the 'Sea Empress' oil spill with special reference to amphipods. Marine Pollution Bulletin. 46(9):1125-1141.
Nixon Z. 2018. Refugio Beach oil spill shoreline oil exposure quantification technical report. Sacramento (CA): California Department of Fish and Wildlife, Office of Spill Prevention and Response.
NOAA (National Oceanic and Atmospheric Administration). 1990. Incident News: Exxon Bayway Refinery.
NOAA (National Oceanic and Atmospheric Administration). 1991. Chevron El Segundo mooring. Incident News.
NOAA (National Oceanic and Atmospheric Administration). 1994. Guidelines and principles for social impact assessment. U.S. Department of Commerce; NOAA Technical Memo, NMFS-F/SPO-16. 29 p.
NOAA (National Oceanic and Atmospheric Administration). 1996. Final M/V *World Prodigy* oil spill restoration plan and environmental assessment, Narragansett Bay, Rhode Island. Gloucester (MA): National Oceanic and Atmospheric Administration, NMFS (National Marine Fisheries Service) Office of Habitat Protection Restoration Center. 59 p.

NOAA (National Oceanic and Atmospheric Administration). 1999. Restoration plan and environmental assessment for the January 19, 1996 *North Cape* oil spill. National Oceanic and Atmospheric Administration. 59 p.
NOAA (National Oceanic and Atmospheric Administration). 2002. Final restoration plan and environmental assessment for the *M/V Kuroshima* oil spill, Summer Bay, Unalaska, Alaska. Seattle (WA): National Oceanic and Atmospheric Administration. 172 p.
NOAA (National Oceanic and Atmospheric Administration). 2010a. Characteristics of coastal habitats: choosing spill response alternatives. Seattle (WA): American Petroleum Institute, National Oceanic and Atmospheric Administration, U.S. Coast Guard, and U.S. Environmental Protection Agency. 78 p.
NOAA (National Oceanic and Atmospheric Administration). 2010b. Oil spills in coral reefs, planning and response considerations. Washington (DC): National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration. 84 p.
NOAA (National Oceanic and Atmospheric Administration). 2012. The toxicity of oil: what's the big deal? Seattle (WA): NOAA Office of Response and Restoration.
NOAA (National Oceanic and Atmospheric Administration). 2013a. Risk assessment for potentially polluting wrecks in U.S. waters. Silver Spring (MD): National Oceanic and Atmospheric Administration. 126+appendices.
NOAA (National Oceanic and Atmospheric Administration). 2013b. Shoreline oiling maps from NOAA's Environmental Response Management Application (ERMA), 2010 and 2013.
NOAA (National Oceanic and Atmospheric Administration). 2016a. Chapter 4: *Deepwater Horizon* oil spill injury to natural resources. Final Programmatic Damage Assessment and Restoration Plan (PDARP) and Final Programmatic Environmental Impact Statement (PEIS). 685 p.
NOAA (National Oceanic and Atmospheric Administration). 2016b. Open water oil identification job aid for aerial observation. Seattle (WA): U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration Emergency Response Division. 54 p.
NOAA (National Oceanic and Atmospheric Administration). 2019. Small diesel spill (500-5,000 gallons) fact sheet. Seattle (WA): Office of Response and Restoration, NOAA. 2 p.
NOAA (National Oceanic and Atmospheric Administration), Rhode Island Department of Environmental Management, U.S. Department of the Interior, U.S. Fish and Wildlife Service. 1999. Restoration plan and environmental assessment for the January 19, 1996 *North Cape* oil spill. 169 p.
NOAA (National Oceanic and Atmospheric Administration), USFWS (U.S. Fish and Wildlife Service), Alaska Department of Law, Alaska Department of Natural Resources, Alaska Department of Environmental Conservation, Alaska Department of Fish and Game Anchorage AK. 2016. Natural resource damage assessment plan for the M/V Selendang Ayu oil spill. Anchorage (AK). 56 p.
NOEP (National Ocean Economics Program). 2020a. Coastal economy data. Center for the Blue Economy.
NOEP (National Ocean Economics Program). 2020b. Ocean economy data. Center for the Blue Economy.
Nordborg FM, Flores F, Brinkman DL, Agustí S, Negri AP. 2018. Phototoxic effects of two common marine fuels on the settlement success of the coral *Acropora tenuis*. Scientific Reports. 8(1):1-12.
Nordborg FM, Jones RJ, Oelgemoller M, Negri AP. 2020. The effects of ultraviolet radiation and climate on oil toxicity to coral reef organisms - a review. Science of the Total Environment. 720:137486.
Northern Economics. 1990. Economic impacts of the SS Glacier Bay oil spill. US Minerals Management Service, Alaska Outer Continental Shelf Region, Anchorage.

NPS (National Park Service). 1992. Nominating historic vessels and shipwrecks to the national register of historic places. Washington (DC): U.S. Government Printing Office. National Register Bulletin 20. 24 p.
NPS (National Park Service). 1997. How to apply the national register criteria for evaluation. Washington (DC): U.S. Government Printing Office. National Register Bulletin 15.
NPS (National Park Service). 1998. Guidelines for evaluating and documenting historic aviation properties. Washington (DC): U.S. Government Printing Office. National Register Bulletin 43.
NRC (National Research Council). 1983. Polycyclic aromatic hydrocarbons from natural and stationary anthropogenic sources and their atmospheric concentrations. Washington (DC): The National Academies Press. 38 p.
NRC (National Research Council). 2003. Oil in the sea III: inputs, fates, and effects. Washington (DC): The National Academies Press. 265 p.
NRC (National Research Council). 2005. Oil spill dispersants: efficacy and effects. Executive summary. National Academies Press. 396 p.
O'Hara PD, Morandin LA. 2010. Effects of sheens associated with offshore oil and gas development on the feather microstructure of pelagic seabirds. Marine Pollution Bulletin. 60(5):672-678.
Oregon Parks and Recreation Department. 2013. Statewide Comprehensive Outdoor Recreation Plan. 667 p.
Oregon State University. 2016. Recreation use values database. Corvallis (OR).
Øritsland NA, Engelhard FR, Juck FA, Hurst RJ, Watts PD. 1981. Effects of crude oil on polar bears. Ottawa, Ontario (CA): Environmental Studies No. 24. Northern Environmental Protection Branch, Indian and Northern Affairs. 268 p.
Orós J, Montesdeoca N, Camacho M, Arencibia A, Calabuig P. 2016. Causes of stranding and mortality, and final disposition of loggerhead sea turtles (*Caretta caretta*) admitted to a wildlife rehabilitation center in Gran Canaria Island, Spain (1998-2014): a long-term retrospective study. PLoS ONE. 11(2): e0149398.
Orth R, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck Jr. KL, Hughes AR, Kendrick GA, Kenworthy J, Olyarnik S, et al. 2006. A global crisis for seagrass ecosystems. BioScience. 56(12):987-996.
Ortmann AC, Anders J, Shelton N, Gong L, Moss AG, Condon RH. 2012. Dispersed oil disrupts microbial pathways in pelagic food webs. PLoS ONE. 7(7):e42548.
OSAT (Operational Science Advisory Team). 2010. Summary report for sub-sea and sub-surface oil and dispersant detection: sampling and monitoring. New Orleans (LA): United States Coast Guard. 131 p.
Osland MJ, Enwright N, Day RH, Doyle TW. 2013. Winter climate change and coastal wetland foundation species: salt marshes vs. mangrove forests in the southeastern United States. Global Change Biology. 19(5):1482-1494.
Overmans S, Nordborg M, Diaz-Rua R, Brinkman DL, Negri AP, Agusti S. 2018. Phototoxic effects of PAH and UVA exposure on molecular responses and developmental success in coral larvae. Aquatic Toxicology. 198:165-174.
Owens E, Parker Hall H, Mauseth G, Graham A, Allard T, Engles J, Lehmann S, Penland S, Reimer P, Whitney J. 2005. Shoreline and surveillance surveys on the M/V Selendang Ayu spill response, Unalaska Island, Alaska. Proceedings of the 28th Arctic Marine Oilspill Program (AMOP) Technical Seminar. Canada: Environment Canada; Ottawa (ON). 509-525 p.
Owens E, Teal A, Haase P. 1991. Berm relocation during the 1990 shoreline cleanup program following the "Exxon Valdez" spill. Proceedings of the 14th Arctic and Marine Oilspill Programme (AMOP) Technical Seminar. Environment Canada; Ottawa (ON). 607-630 p.

Owens EH, Sergy G, Prince R. 2002. The fate of stranded oil at the BIOS site twenty years after the experiment. Proceedings of the Arctic and Marine Oil Spill Program Technical Seminar. Environment Canada; Ottawa (ON). 1-12 p.

Page GW, Carter HR, Ford RG. 1990. Numbers of seabirds killed or debilitated in the 1986 Apex Houston oil spill in Central California. Studies in Avian Biology. 14:164-174.

Paine RT. 1966. Food web complexity and species diversity. The American Naturalist. 100(910):65-75.

Palinkas LA, Downs MA, Petterson JS, Russell J. 1993. Social, cultural, and psychological impacts of the *Exxon Valdez* oil spill. Human Organization. 52(1):1-13.

Parsons G, Firestone J. 2018. Atlantic offshore wind energy development: values and implications for recreation and tourism. Sterling (VA): U.S. Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM 2018-013. 52 p.

Parsons GR, Chen Z, Hidrue MK, Standing N, Lilley J. 2013. Valuing beach width for recreational use: combining revealed and stated preference data. Marine Resource Economics. 28(3):221-241.

Parsons GR, Kang AK, Leggett CG, Boyle KJ. 2009. Valuing beach closures on the Padre Island National Seashore. Marine Resource Economics. 24:213-235.

Paruk JD, Adams EM, Uher-Koch H, Kovach KA, Long IV D, Perkins C, Schoch N, Evers DC. 2016. Polycyclic aromatic hydrocarbons in blood related to lower body mass in common loons. Science of the Total Environment. 565:360-368.

Paruk JD, Long D, Perkins C, East A, Sigel BJ, Evers DC. 2020. Polycyclic aromatic hydrocarbons detected in common loons (*Gavia immer*) wintering off coastal Louisiana. Waterbirds 37 (Special Publication 1). 85-93.

Passow U, Ziervogel K. 2016. Marine snow sedimented oil released during the Deepwater Horizon spill. Oceanography. 29(3):118-125.

Passow U, Ziervogel K, Asper V, Diercks A. 2012. Marine snow formation in the aftermath of the Deepwater Horizon oil spill in the Gulf of Mexico. Environmental Research Letters. 7:035301.

Paxton AB, Peterson CH, Taylor JC, Adler AM, Pickering EA, Silliman BR. 2019. Artificial reefs facilitate tropical fish at their range edge. Communications Biology. 2:168.

Peacock EE, Hampson G, Nelson R, Xu, L, Frysinger GS, Gaines RB, Farrington J, Tripp BW, Reddy CM. 2007. The 1974 spill of the Bouchard 65 oil barge: petroleum hydrocarbons persist in Winsor Cove salt marsh sediments. Marine Pollution Bulletin. 54:214-225.

Pearson CE, James SR, Jr, Krivor MC, El Darragi SD, Cunningham L. 2003. Refining and revising the Gulf of Mexico Outer Continental Shelf region high-probability model for historic shipwrecks: final report. New Orleans (LA): U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region. OCS Study MMS 2003-060.

Pearson WH, Miller SE, Blaylock JW. 1981. Detection of the water-soluble fraction of crude oil by the blue crab, *Callinectes sapidus*. Marine Environmental Research. 5:3-11.

Peckol P, Levings SC, Garrity SD. 1990. Kelp response following the *World Prodigy* oil spill. Marine Pollution Bulletin. 21(10):473-476.

Petersen J, Nelson DM, Marcella T, Michel J, Atkinson M, Boring C, Szathmary L, Weaver J. 2019. Environmental sensitivity index guidelines version 4.0. NOAA Technical Memorandum NOS OR&R 52. Seattle (WA): NOAA Office of Response and Restoration. 228 p.

Peterson CH, Able KW, DeJong CF, Piehler MF, Simenstad CA, Zedler JB. 2008. Practical proxies for tidal marsh ecosystem service: application to injury and restoration. Advances in Marine Biology. 54:221-266.

Peterson CH, McDonald LL, Green RH, Erickson WP. 2001. Sampling design begets conclusions: the statistical basis for detection of injury to and recovery of shoreline communities after the Exxon Valdez oil spill. Marine Ecology Progress Series. 210:255-283.

Petroleum Equipment Institute. 2014. Volatile organic compound (VOC). https://www.pei.org/print/wiki/volatile-organic-compound.

Pezeshki S, DeLaune R. 2015. United States Gulf of Mexico coastal marsh vegetation responses and sensitivities to oil spill: a review. Environments. 2(4):586-607.

Phillott AD, Parmenter J. 2001. Influence of diminished respiratory surface area on survival of sea turtle embryos. Journal of Experimental Zoology. (289):317-321.

Picou JS, Gill DA. 1996. The *Exxon Valdez* oil spill and chronic psychological stress. Proceedings of the American Fisheries Society Symposium 18. 879-893 p.

Pilcher NJ. 2000. The green turtle, *Chelonia mydas*, in the Saudi Arabian Gulf. Chelonian Conservation and Biology. 3:730-734.

Pilson MEQ. 1990. Chemical and biological observations after the *World Prodigy* oil spill. Proceedings of the Oil Spills: Management and Legislative Implications; Newport (RI). American Society of Civil Engineers. 274-290 p.

Plotkin PT, Wicksten MK, Amos AF. 1993. Feeding ecology of the loggerhead sea turtle *Caretta caretta* in the Northwestern Gulf of Mexico. Marine Biology. 115(1):6.

Poncet F, Ragot R, Tintilier F. 2007. The Erika oil spill impact on terrestrial vegetation: main results of five-year monitoring program. Proceedings of the Thirteenth AMOP Technical Seminar; Ottawa (ON). Environment Canada. 301-317 p.

Powell JA. 1978. Evidence of carnivory in manatees (*Trichechus manatus*). Journal of Mammalogy. 59(2):442.

Powers SP, Hernandez FJ, Condon RH, Drymon JM, Free CM. 2013. Novel pathways for injury from offshore oil spills: direct, sublethal and indirect effects of the Deepwater Horizon oil spill on pelagic sargassum communities. PLoS ONE. 8(9):e74802.

Prosser AJ. 2004. Faunal community change following mangrove dieback in Moreton Bay, Australia. A research report submitted to the Department of Botany, School of Life Sciences, University of Queensland, in the partial fulfilment of the requirements of the Bachelor of Science – Honours Degree. 58 p.

Raimondi P, Bell C, Ammann K, Gaddam R, Lohse D, Douglas M, George M, Fletcher N, Anderson L, Minder M. 2019. Assessment of potential impacts to rocky intertidal community following the Refugio Beach oil spill, Santa Barbara County. Prepared for the California Department of Fish and Wildlife: Office of Spill Prevention and Response. 166 p.

Reddy CM, Eglinton TI, Hounshell A, White HK, Xu L, Gaines RB, Frysinger GS. 2002. The *West Falmouth* oil spill after thirty years: the persistence of petroleum hydrocarbons in marsh sediments. Environmental Science & Technology. 36(22):4754-4760.

Reddy CM, Quinn JG. 1999. GC-MS analysis of total petroleum hydrocarbons and polycyclic aromatic hydrocarbons in seawater samples after the *North Cape* oil spill. Marine Pollution Bulletin. 38(2):126-135.

Rees MA, Huey SM, Sorset S. 2019. An assessment of the effects of an oil spill on coastal archaeological sites in Louisiana. New Orleans (LA): U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region. OCS Study BOEM 2019-025. 307 p.

Refugio Beach Oil Spill Trustees. 2020. Refugio Beach oil spill draft damage assessment and restoration plan/environmental assessment. Prepared by the California Department of Fish and Wildlife, California State Lands Commission, California Department of Parks and Recreation, Regents of the University of California, U.S. Department of the Interior, U .S. Fish and Wildlife Service, and National Oceanic and Atmospheric Administration. 173 p.

Reger DR, Corbett D, Steffian A, Saltonstall P, Birkedal T, Yarborough LF. 2000. Archaeological index site monitoring: final report, *Exxon Valdez* oil spill restoration project (restoration project 99007A). Anchorage (AK): Alaska Department of Natural Resources. 55 p.

Reich DA, Etkin DS, Rowe G, Zamorski S. 2016. Modeling oil spill trajectories in Baffin Bay and Lancaster Sound. WWF-Canada Oil Spill Modeling Report. 241 p.

Reilly TJ. 1998. *Julie N* preassessment data report. Cambridge (MA): Prepared for Maine Department of Environmental Protection, Maine Department of Conservation, Maine Department of Inland Fisheries and Wildlife, Maine Department of Marine Resources, National Oceanic and Atmospheric Administration, U.S. Department of Interior. 101 p.

Reiter GA. 1981. Cold weather response F/V *Ryuyo Maru No. 2* St. Paul, Pribilof Islands, Alaska. Proceedings of the International Oil Spill Conference. American Petroleum Institute. 227-231 p.

Renegar DA, Schuler P, Turner N, Dodge R, Riegl B, Knap A, Bera G, Jézéquel R, Benggio B. 2017. TROPICS field study (Panama), 32-year site visit: observations and conclusions for near shore dispersant use NEBA and tradeoffs. Proceedings of the International Oil Spill Conference. American Petroleum Institute. 1-20 p.

Research Planning Inc (RPI). 2005. Preassessment data report the MP-69 / hurricane Ivan oil discharges Mississippi River delta, Louisiana. Silver Spring (MD): Prepared for: Damage Assessment Center, National Oceanic and Atmospheric Administration. 36 p.

Reuscher MG, Baguley JG, Conrad-Forrest N, Cooksey C, Hyland JL, Lewis C, Montagna PA, Ricker RW, Rohal M, Washburn T. 2017. Temporal patterns of Deepwater Horizon impacts on the benthic infauna of the northern Gulf of Mexico continental slope. PLoS ONE. 12(6):e0179923.

Rice SD, Thomas RE, Carls MG, Heintz RA, Wertheimer AC, Murphy ML, Short JW, Moles A. 2001. Impacts to pink salmon following the Exxon Valdez oil spill: persistence, toxicity, sensitivity, and controversy. Reviews in Fisheries Science. 9(3):165-211.

Ritchie LA, Gill DA. 2008. The *Selendang Ayu* shipwreck and oil spill: considering threats and fears of a worst-case scenario. Sociological Inquiry. 78(2):184-206.

Romero IC, Toro-Farmer G, Diercks AR, Schwing P, Muller-Karger F, Murawski S, Hollander DJ. 2017. Large-scale deposition of weathered oil in the Gulf of Mexico following a deep-water oil spill. Environmental Pollution. 228:179-189.

Roth A-MF, Baltz DM. 2009. Short-term effects of an oil spill on marsh-edge fishes and decapod crustaceans. Estuaries and Coasts. 32(3):565-572.

Saintilan N, Rogers K, McKee K. 2009. Salt-marsh-mangrove interactions in Australia and the Americas, chapter 31. Coastal wetlands: an integrated ecosystem approach. Elsevier. 855-883 p.

Sanders HL. 1978. *Florida* oil spill impact on the Buzzards Bay benthic fauna: West Falmouth. Journal of the Fisheries Research Board of Canada. 35(5):717-730.

Sanders HL, Grassle JF, Hampson GR, Morse LS, Garner-Price S, Jones CC. 1980. Anatomy of an oil spill: long-term effects from the grounding of the barge *Florida* off West Falmouth, Massachusetts. Journal of Marine Research. 38:265-380.

Santos LCM, Cunha-Lignon M, Schaeffer-Novelli Y, Cintrón-Molero G. 2012. Long-term effects of oil pollution in mangrove forests (Baixada Santista, Southeast Brazil) detected using a GIS-based multitemporal analysis of aerial photographs. Brazilian Journal of Oceanography. 60(2):159-170.

Sassaman KE, McFadden PS, Monés MP, Palmiotto A, Randell AR. 2014. North Gulf coastal archaeology of the here and now. In: Wallis NJ, editor. New Histories of Pre-Columbian Florida. Gainesville (FL): University Press of Florida. 143-161 p.

Schiel DR, Foster MS. 2015. The biology and ecology of giant kelp forests. Oakland (CA): University of California Press. 416 p.

Schmidt CW. 2010. Between the devil and the deep blue sea: dispersants in the Gulf of Mexico. Environmental Health Perspectives. 118:a338-a344.

Schrier E. 1978. Cleanup efficiency and Biological effects of a Fuel oil spill in cold weather: the 1977 Bouchard No. 65 Oil spill in Buzzards Bay, Massachusetts. Cincinnati (OH): Industrial Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency. 188 p.

Seegar WS, Yates MA, Doney GE, Perkins C, Giovanni M. 2015. Migrating tundra peregrine falcons accumulate polycyclic aromatic hydrocarbons along Gulf of Mexico following Deepwater Horizon oil spill. Ecotoxicology. 24(5):10.

Shapiro K, Khanna S, Ustin S. 2016. Vegetation impact and recovery from oil-induced stress on three ecologically distinct wetland sites in the Gulf of Mexico. Journal of Marine Science and Engineering. 4(2):1-19.

Sharma P, Schiewer S. 2016. Assessment of crude oil biodegradation in arctic seashore sediments: effects of temperature, salinity, and crude oil concentration. Environmental Science and Pollution Research. 23(15).

Sharp B, Cody M, Turner R, Rice S, Spies R, Wolfe D, Wright B. 1996. Effects of the Exxon Valdez oil spill on black oystercatchers. In: Rice SD, Spies RB, Wolfe DA, Wright BA, editors. Proceedings of the Exxon Valdez Oil Spill Symposium: American Fisheries Society Symposium 18. 748-758 p.

Shigenaka G. 2001. Toxicity of oil to reef-building corals: a spill response perspective. NOAA Technical Memorandum NOS OR&R 8, Office of Response and Restoration National Oceanic and Atmospheric Administration, Seattle (WA). 95 p.

Shigenaka G, Milton S, Lutz P. 2010. Oil and sea turtles: biology, planning, and response. National Oceanic and Atmospheric Administration, NOAA's National Ocean Service, Office of Response and Restoration. 116 p.

Short FT, Burdick DM. 1995. Mesocosm experiments quantify the effects of eutrophication on eelgrass, *Zostera marina*. Limnology and Oceanography. 40(4):740-749.

Simpson RD, Smith SDA, Pople AR. 1995. The effects of a spillage of diesel fuel on a rocky shore in the sub-Antarctic region (Macquarie Island). Marine Pollution Bulletin. 31(4):367-371.

Siniff DB, Williams TD, Johnson AM, Garshelis DL. 1982. Experiments on the response of sea otters, *Enhydra lutris* to oil contamination. Biological Conservation. 23:261-272.

Skwarecki B. 2014. Sea turtles' lost years transatlantic journey mapped for first time. Scientific American: https://www.scientificamerican.com/article/sea-turtles-lost-years-transatlantic-journey-mapped-for-first-time/.

Smith SD, Simpson RD. 1995. Effects of the '*Nella Dan*' oil spill on the fauna of *Durvillaea antarctica* holdfasts. Marine Ecology Progress Series. 121:73-89.

Soto LA, Botello AV, Licea-Durán S, Lizárraga-Partida ML, Yáñez-Arancibia A. 2014. The environmental legacy of the Ixtoc-I oil spill in Campeche Sound, southwestern Gulf of Mexico. Frontiers in Marine Science. 1(57):1-9.

Sousa WP, Mitchell BJ. 1999. The effect of seed predators on plant distributions: is there a general pattern in mangroves? Oikos. 86(1):55-66.

St Aubin DJ. 1990. Physiologic and toxic effects on pinnipeds. In: Geraci J, editor. Sea mammals and oil: confronting the risks. New York (NY): Academic Press. 103-127 p.

St Aubin DJ, Lounsbury V. 1990. Oil effects of manatees: evaluating the risks. In: Geraci J, St Aubin D, editors. Sea mammals and oil: confronting the risks. New York (NY): Academic Press. 241-251 p.

Stacy BA. 2012. Summary of findings for sea turtles documented by directed captures, stranding response, and incidental captures under response operations during the BP *Deepwater Horizon* (Mississippi Canyon 252) oil spill. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service. DWH-AR0149670. 254 p.

Stacy NI, Field CL, Staggs L, MacLean RA, Stacy BA, Keene J, Cacela D, Pelton C, Cray C, Kelley M, et al. 2017. Clinicopathological findings in sea turtles assessed during the Deepwater Horizon oil spill response. Endangered Species Research. 33:25-37.

State of Louisiana. 2005. Final damage assessment and restoration plan and environmental assessment: equinox oil company crude oil discharge, Lake Grand Ecaille, LA. Prepared by: Louisiana Oil

Spill Coordinator's Office/Office of the Governor, Louisiana Department of Environmental Quality, Louisiana Department of Natural Resources, Louisiana Department of Wildlife and Fisheries, National Oceanic and Atmospheric Administration, United States Fish and Wildlife Service. 98 p.

Stewardship Centre of British Columbia. 2004. Coastal shore stewardship: a guide for planners, builders and developers on Canada's pacific coast. Vancouver (BC): The Stewardship Series. 96 p.

Stirling HP. 1977. Effects of a spill of marine diesel oil on the rocky shore fauna of Lamma Island, Hong Kong. Environmental Pollution. 12:93-117.

Stoffyn-Egli P, Lee K. 2002. Formation and characterization of oil–mineral aggregates. Spill Science & Technology Bulletin. 8(1):31-44.

Stout JP. 1984. The ecology of irregularly flooded salt marshes of the northeastern Gulf of Mexico: a community profile. Washington (DC): National Coastal Ecosystems Team, Division of Biological Services, Research and Development, U.S. Fish and Wildlife Service. 98 p.

Stutts M. 2014. National register of historic places GIS dataset. U.S. Department of the Interior NPS.

Suchanek TH. 1993. Oil impacts on marine invertebrate populations and communities. American Zoologist. 33(6):510-523.

Sullivan L, Brosnan T, Rowles TK, Schwacke L, Simeone C, Collier TK. 2019. Guidelines for assessing exposure and impacts of oil spills on marine mammals. Silver Springs (MD): U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service. NOAA Technical Memorandum NMFS-OPR-62. 92 p.

Sumaila UR, Cisneros-Montemayor AM, Dyck A, Huang L, Cheung W, Jacquet J, Kleisner K, Lam V, Mccrea-Strub A, Swartz W, et al. 2012. Impact of the Deepwater Horizon well blowout on the economics of US gulf fisheries. Canadian Journal of Fisheries and Aquatic Sciences. 69:499-510.

Tandon A. 2018. First aid to cultural heritage in times of crisis. Rome: International Centre for the Study of the Preservation and Restoration of Cultural Property (ICCROM).

Tang D, Jing S, Zhou L, Wang S, Singh RP, Pan G. 2019. Ecological response of phytoplankton to the oil spills in the oceans. Geomatics, Natural Hazards and Risk. 10(1):853-872.

Tansel B, Fuentes C, Sanchez M, Predoi K, Acevedo M. 2011. Persistence profile of polyaromatic hydrocarbons in shallow and deep Gulf waters and sediments: effect of water temperature and sediment-water partitioning characteristics. Marine Pollution Bulletin. 62(12):2659-2665.

Tarr MA, Zito P, Overton EB, Olson GM, Adhikari PL. 2016. Weathering of oil spilled in the marine environment. Oceanography. 29(3):126-135.

Teal JM, Farrington JW, Burns KA, Stegeman JJ, Tripp BW. 1992. The *West Falmouth* oil spill after 20 years: fate of fuel oil compounds and effects on animals. Marine Pollution Bulletin. 24(12).

Tenera Environmental. 2020. Changes in the condition of surfgrass (*Phyllospadix torreyi*) and macroalgae following the Refugio oil spill, Santa Barbara County. San Luis Obispo (CA): Prepared for California Department of Fish and Wildlife by Tenera Environmental. Refugio Oil Spill Surfgrass Study 1. Document ESLO2020-01. 25 p.

Texaco Oil Spills Natural Resources Trustees. 2004. Final restoration plan and environmental assessment for the Texaco oil spills into Fidalgo Bay, Anacortes, Washington in 1991 and 1992. Olympia (WA). 60 p.

Texas General Land Office, Texas Natural Resource Conservation Commission, Texas Parks and Wildlife Department, U.S. Department of the Interior (USDOT), U.S. Fish & Wildlife Service (USFWS), U.S. Department of Commerce, National Oceanic and Atmospheric Administration (NOAA). 1997. Restoration plan/environmental assessment for the *Apex Barges* oil spill, July 28, 1990. 30 p.

Thayer GW, Kenworthy WJ, Fonseca MS. 1984. Ecology of eelgrass meadows of the Atlantic Coast: a community profile. Virginia University, Charlottesville (VA): U.S. Fish and Wildlife Service,

National Marine Fisheries Service, Beaufort, NC. Beaufort Lab. Department of Environmental Sciences.
Timmermann K, Banta GT, Klinge L, Andersen O. 2011. Effects of bioturbation on the fate of oil in coastal sandy sediments–an in situ experiment. Chemosphere. 82(10):1358-1366.
Tiner R, Burke D. 1995. Wetlands of Maryland. Hadley (MA): US Fish and Wildlife Service and Annapolis (MD): Department of Natural Resources Cooperative Publication. 195+appendices.
Torch/Platform Irene Oil Spill Natural Resource Trustees. 2005. Final preliminary analysis of injuries: abalone and rocky intertidal habitat. Prepared by the Natural Resource Trustee Agencies for the Torch/Platform Irene Oil Spill. California Department of Fish and Game, California State Lands Commission, National Oceanic and Atmospheric Administration, U.S. Fish and Wildlife Service, U.S. Department of Defense, Vandenberg Air Force Base. 8 p.
Torch/Platform Irene Trustee Council. 2007. Torch/Platform *Irene* oil spill final restoration plan and environmental assessment. Prepared by Torch/Platform Irene Trustee Council: United States Fish and Wildlife Service, California Department of Fish and Game, United States Department of Air Force, Vandenberg Air Force Base, California State Lands Commission with Assistance from Santa Barbara County Planning and Development Department. 220 p.
Tornfelt EE, Burwell M. 1992. Shipwrecks of the Alaskan shelf and shore. Anchorage (AK): U.S. Department of the Interior, Minerals Management Service, Alaska OCS Region.
TRC Environmental Corporation. 2012. Inventory and analysis of archaeological site occurrence on the Atlantic Outer Continental Shelf. New Orleans (LA): U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region. OCS Study BOEM 2012-008. 324 p.
Troisi G, Borjesson L, Bexton S, Robinson I. 2007. Biomarkers of polycyclic aromatic hydrocarbon (PAH)-associated hemolytic anemia in oiled wildlife. Environmental Research. 105(3):324-329.
Turner NR, Renegar DA. 2017. Petroleum hydrocarbon toxicity to corals: a review. Marine Pollution Bulletin. 119(2):1-16.
U.S. Department of Energy Office of Biological and Environmental Research. 2008. Oceanic food web. https://commons.wikimedia.org/wiki/File:Oceanic_Food_Web.jpg.
UNESCO. 2003. Convention for the safeguarding of the intangible cultural heritage. Paris (FR).
USFWS (U.S. Fish and Wildlife Service). 1997. The impact of a ship grounding and associated fuel spill at Rose Atoll National Wildlife Refuge, American Samoa. Honolulu (HI): Pacific Islands Ecoregion. 60 p.
USFWS (U.S. Fish and Wildlife). 2007. Sea turtles: global species at risk. https://www.fws.gov/home/feature/2007/seaturtles2.pdf.
USFWS (U.S. Fish and Wildlife). 2020a. Beach mice. https://www.fws.gov/panamacity/beachmice.html.
USFWS (U.S. Fish and Wildlife). 2020b. West Indian manatee (*Trichechus manatus*). https://www.fws.gov/southeast/wildlife/mammals/manatee/.
USGS (U.S. Geological Service). 2015. M079 North American Atlantic & Gulf coastal salt marsh macrogroup. https://www1.usgs.gov/csas/nvcs/nvcsGetUnitDetails?elementGlobalId=860714.
USGS (U.S. Geological Service). 2016. G535 *Carex subspathacea - Dupontia fisheri* Salt Marsh Group. https://www1.usgs.gov/csas/nvcs/nvcsGetUnitDetails?elementGlobalId=860714.
Valentine DL. 2019. Benthic oiling from the 19 May 2015 line 901 rupture at Refugio: a compilation of evidence. Goleta (CA): Valentine Scientific and Consulting Services, Inc. 14 p.
Van Horne YO, Chief K, Charley PH, Begay M-G, Lothrop N, Bell ML, Canales RA, Teufel-Shone NI, Beamer PI. 2021. Impacts to Diné activities with the San Juan River after the Gold King Mine Spill. Journal of Exposure Science and Environmental Epidemiology.1-15.
Veltre DW, Veltre MJ. 1982. Resource utilization in Unalaska, Aleutian Islands, Alaska. Technical paper# 58. Alaska Department of Fish and Game, Division of Subsistence. Contract 82-0790. 137 p.

Vonk SM, Hollander DJ, Murk AJ. 2015. Was the extreme and wide-spread marine oil-snow sedimentation and flocculent accumulation (MOSSFA) event during the Deepwater Horizon blow-out unique? Marine Pollution Bulletin. 100(1):5-12.
Wade TL, Jackson TJ, McDonald TJ, Wilkinson DL, Brooks JM. 1993. Oysters as biomonitors of the Apex Barge oil spill. Proceedings of the International Oil Spill Conference: American Petroleum Institute. 313-317 p.
Walker DA, Webber PJ, Everett KR, Brown J. 1978. Effects of crude and diesel oil spills on plant communities at Prudhoe Bay, Alaska, and the derivation of oil spill sensitivity maps. Arctic. 31(3):242-259.
Wallace BP, Stacy BA, Cuevas E, Holyoake C, Lara PH, Marcondes ACJ, Miller JD, Nijkamp H, Pilcher NJ, Robinson I, et al. 2020. Oil spills and sea turtles: documented effects and considerations for response and assessment efforts. Endangered Species Research. 41:17-37.
Wang W, Li X, Wang M. 2019. Propagule dispersal determines mangrove zonation at intertidal and estuarine scales. Forests. 10(3):24.
Wang Y, Lee K, Liu D, Guo J, Han Q, Liu X, Zhang J. 2020. Environmental impact and recovery of the Bohai Sea following the 2011 oil spill. Environmental Pollution. 263:114343.
Ward CP, Overton EB. 2020. How the 2010 Deepwater Horizon spill reshaped our understanding of crude oil photochemical weather at sea: a past, present, and future perspective. Environmental Science: Processes & Impacts. 25(5):1125-1138.
Ward GA, Baca B, Cyriacks W, Dodge RE, Knap A. 2003. Continuing long-term studies of the TROPICS Panama oil and dispersed oil spill sites. Proceedings of the International Oil Spill Conference (IOSC). American Petroleum Institute; Washington (DC). 259-267 p.
Washington SCORP (State Recreation and Conservation Office). 2013. Outdoor recreation in Washington: the 2013 state comprehensive outdoor recreation plan. Olympia (WA): Washington State Recreation and Conservation Office. 175 p.
Wernberg T, Krumhansl K, Filbee-Dexter K, Pedersen MF. 2019. Status and trends for the world's kelp forests. World seas: an environmental evaluation. Elsevier. 57-78 p.
Widbom B, Oviatt CA. 1994. The 'World Prodigy' oil spill in Narragansett Bay, Rhode Island: acute effects on macrobenthic crustacean populations. Hydrobiologia. 291(2):115-124.
Wiegert RG, Freeman B. 1990. Tidal salt marshes of the southeast Atlantic coast: a community profile. Washington (DC): U.S. Department of the Interior, Fish and Wildlife Service. 79 p.
Wilhelm SI, Robertson GJ, Ryan PC, Schneider DC. 2007. Comparing an estimate of seabirds at risk to a mortality estimate from the November 2004 *Terra Nova* FPSO oil spill. Marine Pollution Bulletin. 54(5):537-544.
Wilkin SM, Rowles TK, Stratton E, Adimey N, Field CL, Wissmann S, Shigenaka G, Fougères E, Mase B, Southeast Region Stranding Network, et al. 2017. Marine mammal response operations during the *Deepwater Horizon* oil spill. Endangered Species Research. 33:107−118.
Witherington B, Hirama S, Hardy R. 2012. Young sea turtles of the pelagic Sargassum-dominated drift community: habitat use, population density, and threats. Marine Ecology Progress Series. 463:1-22.
Witherington BE. 1994. Flotsam, jetsam, post-hatchling loggerheads, and the advecting surface smorgasbord. In: Bjorndal KA, Bolten AB, Johnson DA, Eliazar PJ, editors. Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation NOAA Tech Memo NMFS-SEFSC-351; Hilton Head (SC). 166-168 p.
Witting D, Donohoe R, Sullivan L, Ricker R, Anulacion B, Ylitalo G, Martin K. 2018. Impact of the Refugio Beach oil spill on two surf zone fish: Surfperch (*Embiotocidae*) and California grunion (*Leuresthestenuis*). Sacramento (CA): Poster for SETAC North America 29th Annual Meeting. 15 p.

Woodin SA, Nyblade CF, Chia F-S. 1972. Effect of diesel oil spill on invertebrates. Marine Pollution Bulletin. 3(9):139-143.
Wormald AP. 1976. Effects of a spill of marine diesel oil on the meiofauna of a sandy beach at Picnic Bay, Hong Kong. Environmental Pollution. 11(2):117-130.
Yamato O, Goto I, Maede Y. 1996. Hemolytic anemia in wild seaducks caused by marine oil pollution. Journal of Wildlife Diseases. 32(2):381-384.
Yan B, Passow U, Chanton J, Nothig E-M, Asper V, Sweet J, Pitiranggon M, Diercks A, Pak DK. 2016. Sustained deposition of contaminants from the *Deepwater Horizon* spill. Proceedings of the National Academy of Sciences, 113, no 24 (2016): E3332-E3340.
Yilmaz N, Isinibilir M. 2018. Interactions between zooplankton and oil spills: lessons learned from global accidents and a proposal for zooplankton monitoring. In: Unlu S, Alpar B, Ozturk B, editors. Oil spill along the Turkish Straits sea area; accidents, environmental pollution, socio-economic impacts and protection. Istanbul (TK): Turkish Marine Research Foundation. 238-243 p.
Yim UH, Kim M, Ha SY, Kim S, Shim WJ. 2012. Oil spill environmental forensics: the *Hebei Spirit* oil spill case. Environmental Science & Technology. 46:6431-6437.
Zedler JB, Bonin CL, Larkin DJ, Varty A. 2008. Salt marshes. Encyclopedia of Ecology. Oxford (GB): Elsevier. 3132-3141 p.
Zelenke B, O'Connor C, Barker C, Beegle-Krause CJ, Eclipse L. 2012. General NOAA operating modeling environment (GNOME) technical documentation. U.S. Dept. of Commerce, NOAA Technical Memorandum NOS OR&R 40. Seattle (WA): Emergency Response Division, NOAA. 105 p.
Zenetos A, Hatzianestis J, Lantzouni M, Simboura M, Sklivagou E, Arvanitakis G. 2004. The Eurobulker oil spill: mid-term changes of some ecosystem indicators. Marine Pollution Bulletin. 48(1-2):122-131.
Zengel S, Bernik BM, Rutherford N, Nixon Z, Michel J. 2015. Heavily oiled salt marsh following the Deepwater Horizon oil spill, ecological comparisons of shoreline cleanup treatments and recovery. PLoS ONE. 10(7):1-27.
Zengel S, Michel J. 2013. *Deepwater Horizon* oil spill: salt marsh oiling conditions, treatment testing, and treatment history in Northern Barataria Bay, Louisiana (Interim Report October 2011). Seattle (WA): NOAA. Technical Memorandum. 82 p.
Zhou L, Tang D, Wang S, Pan G. 2014. Satellite observations: oil spills impact on phytoplankton in Bohai Sea. Proceedings of the Dragon 3 Mid-Term Results Symposium; Chengdu (CN). 11 p.
Ziccardi M. 2015. Expert assistance-Deepwater Horizon oil spill bird injury. Davis (CA): DWH Birds NRDA Technical Working Group Report. Prepared for the Department of the Interior U.S. Fish and Wildlife Service.
Zieman JC, Orth R, Phillips RC, Thayer G, Thorhaug A. 1984. Effects of oil on seagrass ecosystems. In: Cairns J, Jr., Buikema A, Jr., editors. Restoration of habitats impacted by oil spills. Boston (MA): Butterworth Publishers. 37-64 p.

Appendix A: List of Spills Included in the Oil Spill Effects Literature Study of Median-Range Volume Spills (500–20,000 bbl)

1. *Agip Abruzzo*, Ligurian Sea, Italy, April 1991, 2,000 tons (~13,500 bbl) Iranian Light crude oil and an unknown amount of IFO 380.
2. *American Trader*, Huntington Beach, CA, 1990, 9,919 bbl Alaska North Slope crude oil.
3. *Anitra,* Delaware River, 1996, 952 bbl Nigerian light crude oil.
4. *Antonio Gramsci*, Baltic Sea, 1987, 4,200 bbl of crude oil in Gulf of Finland (onto sea ice).
5. Apex barges collision with the M/V *Shinoussa*, Galveston Bay, TX, 1990, 16,700 bbl partially refined crude oil.
6. *Apex Houston,* Offshore Marin, San Francisco, San Mateo, Santa Cruz, and Monterey Counties, 1986, 615 bbl crude oil.
7. *ARCO Anchorage*, Port Angeles, WA, 1985, 5,690 bbl Alaska North Slope crude oil.
8. Baffin Island Oil Spill (BIOS), Canadian Arctic, 1981, 280 bbl sweet medium crude oil (Venezuelan Lagomedi) experimental field oiling: 1) oil only and 2) oil+dispersant.
9. *Bahia Paraiso*, January 1989, Arthur Harbor, Antarctica, 3,760 bbl diesel fuel arctic.
10. Barataria Bay, LA, April 2005, 600 bbl South Louisiana crude oil.
11. Bayou Perot, 2007, 8,500 bbl of >45 API gravity condensate crude oil.
12. Bohai Sea, China, 2011, 3,200 bbl crude oil and drilling mud.
13. *Bouchard 65*, 1974, Buzzards Bay, MA, 600 bbl No. 2 fuel oil.
14. *Bouchard 65*, January 1977, Buzzards Bay, MA, 2,000 bbl No. 2 fuel oil (in ice).
15. *Bouchard-120*, 2003, Buzzards Bay, MA, 2,333 bbl No. 6 fuel oil.
16. BP MP-80 Delta 20 pipeline spill, 2004, Hurricane Ivan, LA, 7,058 bbl South Louisiana crude oil.
17. *Bunga Kelana*, 2010, Singapore, 15,000 bbl crude oil.
18. *Cosco Busan*, 2007, San Francisco Bay, CA, 1,285 bbl heavy fuel oil.
19. Eagle Creek, Queen Charlotte Islands, Canada, 1984, 1,000 bbl diesel and 250 bbl gasoline.
20. *El Omar*, 1988, Milford Haven, Wales, 100 tons (~670 bbl) light Iranian crude oil.
21. El Segundo, Santa Monica Bay, CA, 1991, 500 bbl diesel-like oil from offshore Chevron pipeline.
22. Equinox well blowout, 1998, Lake Grande Ecaille, LA, 1,535 bbl South Louisiana crude oil.
23. *Esso Bayway*, Texas, 1979, 900 tons (~6,000 bbl) Light Arabian crude oil.

24. *Estrella Pampeana*, 1999, Ria de la Plata, Argentina, 15,700 bbl crude oil.
25. *Eurobulker,* 2000, Aegean Sea, Greece, 700 tons (~4,850 bbl) crude oil (though ITOPF website says it was a mix of fuel oil/diesel oil).
26. Exxon Bayway Refinery, Arthur Kill, NY, 1990, 13,715 bbl No. 2 fuel oil.
27. Fidalgo Bay, WA, 1991, 714 bbl Alaska North Slope crude oil from a Texaco Refinery.
28. *Florida*, West Falmouth, MA, 1969, 4,385 bbl No. 2 fuel oil.
29. *Garbis*, Florida Keys, 1975, 1,500-3,000 bbl crude oil emulsion.
30. *Glacier Bay*, July 1987, Cook Inlet, AK, 3,100 bbl Alaska North Slope crude oil.
31. Greenhill well blowout, September 1992, Timbalier Bay, LA, average of 60 bbl per hour of crude oil for 13 days, though much of the oil burned.
32. *Jessica*, Galapagos Islands, Ecuador, 2001, 2,800 bbl diesel and 2,160 bbl bunker fuel.
33. *Julie N*, Portland, Maine, 1996, 2,058 bbl home heating fuel and 2,219 bbl IFO 380.
34. *Kuroshima*, Unalaska, AK, 1997, 928 bbl heavy fuel oil.
35. Lake Barre, LA, 1997, 6,561 bbl South Louisiana crude oil from Texaco pipeline.
36. Lamma Island, Hong Kong, 1973, 2,000-3,000 tons (~14,000-21,000 bbl) heavy marine diesel.
37. *Laura d'Amato*, Sydney Harbor, Australia, August 1999, 1,750 bbl Murban light crude oil.
38. Long Island Sound, March 1972, 1,905 bbl No. 2 fuel oil.
39. McGrath Beach/Berry Petroleum, CA, 1993, 2,075 bbl crude oil.
40. Moreton Bay, Australia, 2003, 11,900 bbl crude oil.
41. Mosquito Bay, LA (Williams Pipeline), 2001, 2,380-3,000 bbl condensate.
42. Nakika MP-69 pipeline Hurricane Ivan oil spill, September 2004, Louisiana, 4,528 bbl South Louisiana crude oil.
43. *Nella Dan*, Macquarie Island, southwest of New Zealand, 1987, 1,690 bbl diesel.
44. *North Cape*, South Kingstown, RI,s 1996, 19,700 bbl, home heating oil
45. Refugio Beach (All American Pipeline), CA, 2017, 500 bbl, Monterey crude oil.
46. Rose Atoll (*Jin Shiang Fa*), 1997, American Samoa, 2,380 bbl diesel and lube oil.
47. *RTC-380*, Long Island Sound, NY/CT, 1992, 524 bbl diesel.
48. *Ryuyo Maru No. 2*, Nov 1979, St Paul Island, Bering Sea, AK, 6,190 bbl No. 2 fuel oil.
49. Sao Paulo pipeline spill, Brazil, 1983, 16,250 bbl Brazil crude oil.
50. *Sefir*, 1981, Baltic Sea, ~2,800 bbl No. 1 fuel oil, diesel, leaked for 6+ weeks.
51. *Selendang Ayu*, 2004, Unalaska, AK, 8,300 bbl of intermediate fuel oil and marine diesel.

52. Tebar V refinery spill, Sao Paulo, Brazil, 19,000 bbl crude oil.
53. *Tenyo Maru*, 32 km northwest of Cape Flattery, WA, 6,500 bbl intermediate fuel oil, 2,166 bbl diesel, and unknown amounts of lube oil and fish oil.
54. Terra Nova FPSO blowout, 2004, Grand Banks, Canada, 1,000 bbl Hibernia crude oil.
55. Texaco March Point Refinery, 1971, Guemes Island, WA, 4,700 bbl diesel oil.
56. Torch platform, 1997, Offshore Santa Barbara, CA, 163 bbl of an emulsion of California crude oil and produced water, plus 21 bbl of diesel and 19 bbl of anti-corrosion chemicals.
57. TROPICS (TRopical Oil Pollution Investigations in Coastal Systems) field oiling study, 1984, Panama, 6 bbl Prudhoe Bay crude oil.
58. Uniacke G-72, 1984, blowout off Sable Island, Nova Scotia, 1,500 bbl gas condensate.
59. *Universe Leader*, Bantry Bay, Ireland, 1974, 650,000 gal Kuwait crude oil.
60. UNOCAL, Avila Beach, CA, 1992, 600 bbl San Joaquin Valley crude oil from UNOCAL tank farm.
61. *World Encouragement*, Botany Bay, Australia 1979, 100 tons (~660 bbl) Arabian crude oil.
62. *World Prodigy*, Narragansett Bay, RI, 1989, 6,900 bbl home heating oil.

Department of the Interior (DOI)

The Department of the Interior protects and manages the Nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors the Nation's trust responsibilities or special commitments to American Indians, Alaska Natives, and affiliated island communities.

Bureau of Ocean Energy Management (BOEM)

The mission of the Bureau of Ocean Energy Management is to manage development of U.S. Outer Continental Shelf energy and mineral resources in an environmentally and economically responsible way.

BOEM Environmental Studies Program

The mission of the Environmental Studies Program is to provide the information needed to predict, assess, and manage impacts from offshore energy and marine mineral exploration, development, and production activities on human, marine, and coastal environments. The proposal, selection, research, review, collaboration, production, and dissemination of each of BOEM's Environmental Studies follows the DOI Code of Scientific and Scholarly Conduct, in support of a culture of scientific and professional integrity, as set out in the DOI Departmental Manual (305 DM 3).