From: Emily Jeffers [mailto:ejeffers@biologicaldiversity.org]
Sent: Tuesday, March 21, 2023 3:19 PM
To: CityClerk <CityClerk@longbeach.gov>
Subject: CBD Sources Part 3.2

-EXTERNAL-

Please see attached sources for our comment letter.

Analysis of Hydrocarbons in Samples Provided from the

Cruise of the R/V WEATHERBIRD II, May 23-26, 2010

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Summary

The National Oceanic and Atmospheric Administration (NOAA) undertook, through a certified testing laboratory, an independent analysis of 25 water samples provided from the cruise of the R/V WEATHERBIRD II during its mission to sample for hydrocarbons associated with the Mississippi Canyon 252 (MC-252) incident. One liter samples were provided in amber bottles from six to 12 depths at three discrete sampling locations: Station 01 was 142 nautical miles southeast of the MC-252 incident; Station 07 was 45 n. miles northeast of the incident, and the Station labeled "slick1" was 40 n. miles northeast.

Concentrations of total petroleum hydrocarbons (TPH) were less than 0.5 mg/L (parts per million) at all stations and depths. TPH concentrations in these 25 samples ranged from 0.085 to 0.480 mg/L (parts per million), with a median value of 0.203 mg/L (mean 0.223 mg/L). Concentrations of the individual 16 EPA priority pollutants[1] ranged from below reporting limits (< 10-13 ng/L or parts per trillion) to 79.2 ng/L (naphthalene, DSH Slick1-09). Five samples had reportable concentrations of these 16 polycyclic aromatic hydrocarbons (PAHs), with Total concentrations for the 16 priority pollutant PAHs (Total PAH_{16} or $TPAH_{16}$) ranging from 17.7 to 79.2 ng/L. PAH concentrations were estimated below the reporting limits (but above the minimum detection limits – MDLs).[2] For $TPAH_{16}$, PAHs were detected above the MDLs in all samples, ranging from 9 to 112 ng/L with a median value for all 25 samples of 26 ng/L.

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[1] EPA Priority Pollutants: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, dibenz[*a,h*]anthracene, benzo[*g,h,i*]perylene, and indeno[*1,2,3-cd*]pyrene.

[2] Note that reporting limit is defined as the lowest level to which the analytical instrumentation is calibrated. Although instrumentation has a defined "lower limit", it is still possible to detect compounds below this lowest

A comparison of TPH and $TPAH_{16}$ concentrations among the three stations (see Table 1, Figures 2 and 3) suggests that all three locations have similar depth distributions of TPH and $TPAH_{16}$ concentrations. Further, comparisons of the individual PAH_{16} concentrations to eco-toxicological benchmarks pulled together from the literature (e.g., NOAA SQuiRTs Tables), illustrate that the PAH concentrations measured in these 25 water samples are well below these eco-toxicological benchmarks.

Despite low overall concentrations several samples had sufficient concentrations of petroleum biomarkers that allowed for a preliminary assessment with the MC 252 source oil:

a. At the station labeled “slick1”, surface samples had hydrocarbons present[3] and based on the distribution of diagnostic petroleum compounds these hydrocarbons are CONSISTENT with an MC 252 source.

b. At Station DHS-01 at 100 meters depth, and at 300 meters depth, hydrocarbons were detected, but based on the distribution of diagnostic petroleum compounds these hydrocarbons are INCONSISTENT with an MC 252 source

c. At station DSH-07, samples from the surface, 50, and 400 meters depth appear to have trace oil present, but concentrations are too low to allow for source determination.

d. The petroleum concentrations in the remaining samples are too low for correlation.

Report

A recent joint USF/NOAA cruise was recently conducted by the R/V Weatherbird II. One of the objectives of the cruise was to evaluate if reflecting layers identified by acoustics were related to subsea petroleum. Of immediate concern was the ability to identify and potentially track deep subsurface

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calibration level, but still meet the criteria for determining if a compound is actually present. If these criteria are met as determined by seasoned analysts, these compound(s) are confirmed to be present and the concentration is reported but qualified with a “J” to denote an estimated value.

The method detection limit (MDL) is defined in *40CFR part 136*. Laboratories determine a MDL for each compound analyzed by processing a series of laboratory-spiked replicates through the entire sample extraction and analytical process. Statistics – as defined in 40CFR part 136 - are then applied to the results in order to mathematically determine the MDL for each compound.

[3] See Table 1 for sample hydrocarbons concentrations.

plumes that may be derived from the dispersed MC 252 oil. To accomplish this, the crew conducted fluorometry, salinity, temperature, dissolved oxygen, and conducted acoustic analyses, in addition to collecting water samples for chemical identification of the signals derived from their onboard instrumentation. Seawater samples were collected by scientist from the University of South Florida (USF) at each sampling station, and 130 one liter splits were provided to NOAA for independent analysis. USF scientists identified the 25 highest priority samples from three stations (Figure 1). Station DSH-01 was located 142 nautical miles southeast of the MC-252 incident; Station DHS-07 was 45 n. miles northeast of the Deepwater Horizon MC 252 incident, and the Station labeled "slick1" was 40 n. miles northeast of the location of the Deepwater Horizon MC 252 site. NOAA sent these samples to Alpha Analytical for quantification of TPH and PAHs using EPA SW846 for TPH and a modified method 8270c for PAHs.

In light of the Deepwater Horizon MC252 (DWH-MC252) release, including the unprecedented use of surface and subsurface applied dispersants, it's important for NOAA to understand the distribution and magnitude of any sub surface dispersed oil derived from this release. The measurement of TPH and PAH concentrations provide a quantitative evaluation of the amount of any hydrocarbons in these samples. Additionally, NOAA is interested in understanding if hydrocarbons found at depth are from the DWH-MC252 release or from natural or other anthropogenic sources of oil. To accomplish the source identification, results from the Gas Chromatography/Mass Spectrometry (GCMS) analysis were evaluated by experienced petroleum geochemists who developed their conclusions based on review of the m/z 85, 191, and 217 ion traces[4]. An important consideration with these analyses is to understand that they require a minimum amount of hydrocarbons in the samples to ensure that the diagnostic hydrocarbon signal is strong enough to measure.

Results and Discussion

TPH

Results for TPH are presented in Table 1 and Figure 2. TPH concentrations for all water samples are fairly low; ranging from 0.085 to 0.48 mg/L (parts per million) with a median value of 0.203 mg/L. As illustrated in Figure 1, there is a general decrease in TPH concentrations with increasing depth at all three stations. For DHS-01, the TPH concentration decreases from 0.35 mg/L at the surface to 0.14 mg/L at 500 m. For DSH-07, the TPH concentrations generally decrease with depth, with an elevated concentration of 0.40 mg/L at a depth of 300 m. At station Slick1, TPH concentrations are highest at a depth of 50 m (0.48 mg/L), and generally constant with depth from 100 m to 400 m.

PAHs

Total PAH concentrations for all 16 EPA priority pollutants are presented in Table 2. Concentrations of individual PAHs were measured above the reporting limits in only 5 of the 25 samples. In these samples, $TPAH_{16}$ range from 17.7 to 18.7 ng/L. For 4 of the 5 samples, the $TPAH_{16}$ concentrations are supported

[4] The mass to charge ratios for 85 is for n-alkanes and isoprenoids, 191 is the ion used for terpanes, and 217 is for steranes and diasteranes. Which are diagnostic of petroleum.

by one or two, two or three-ring PAH compounds. Only one of the samples analyzed has reportable concentrations of the EPA priority pollutant PAHs in the high molecular weight range (i.e., 4 to 6 ring PAHs).

Considering the potential for subsurface dispersed oil and the presence of surface oil and mousse at stations Slick1 and DHS 07, the TPH and PAH concentrations measured in these samples are relatively low. As noted above, the TPH and $TPAH_{16}$ concentration profiles from all three stations are similar, even though station DHS-01 is far to the south (see figure 1) and surface oil was found at stations Slick1 and DSH07. Comparison of the maximum individual PAH concentrations (or estimated concentrations where it fell below the reporting limit) to benchmark exposure concentrations from the literature (e.g., NOAA SQuiRTs) supports the contention that these hydrocarbon concentrations are low.

Source

Evaluation of the mass spectra from these analyses indicates that for most of the samples, the hydrocarbons present are at concentrations too low to discern a pattern. However, despite low overall concentrations several samples had sufficient concentrations of diagnostic petroleum hydrocarbons that allowed for a preliminary assessment with the MC 252 source oil:

a. At the station labeled "slick1", surface samples had trace hydrocarbons present[5] and based on the distribution of diagnostic petroleum compounds these hydrocarbons are CONSISTENT with an MC 252 source (Fig. 4).

b. At Station DHS-01 at 100 meters depth, and at 300 meters depth, trace hydrocarbons were detected, but based on diagnostic petroleum compounds these hydrocarbons are INCONSISTENT with an MC 252 source.

c. At station DSH-07, samples from the surface, 50, and 400 meters depth appear to have trace oil present, but concentrations were too low to allow for source determination.

d. Concentrations of diagnostic petroleum compounds in the remaining samples were too low for correlation.

[5] See Table 1 for sample hydrocarbons concentrations.

Interim conclusions

Hydrocarbons are present in a number of the water samples collected from all three stations. TPH concentrations measured in these samples are low, with none being above 0.5 mg/L. Individual PAH concentrations are also low, being in the ng/L or parts per trillion range. Only 5 of the 25 samples had reportable levels of any of the 16 EPA priority pollutant PAHs, and of these, the highest $TPAH_{16}$ value was 79 ng/L (Station DHS-07, 200 m depth). The next highest $TPAH_{16}$ value was 49 ng/L (slick1, 350 m depth).

These data do not show a substantial difference between the hydrocarbon concentrations or distributions measured in samples from station DHS-01, 142 nautical miles southeast of the Deepwater Horizon MC 252 well head and stations DHS-07 and Slick1 located 45 and 40 nautical miles (respectively) to the northeast of the well site. The Slick1 station does have an elevated TPH concentration at 50 m, while station DHS-07 has an elevated TPH concentration at 300 m depth. Neither of these depths corresponds with an elevated $TPAH_{16}$ concentration.

Mass spectral analysis of the m/z 85, 191, and 217 mass spectra suggest the surface sample from the Slick1 station contains hydrocarbons consistent with a Deepwater Horizon MC252 source. Hydrocarbons are also present in samples from station DHS-01 at 100 m and 300 m depths that are inconsistent with a Deepwater Horizon MC 252 source. Finally, results from station DHS-07 indicate that trace hydrocarbons are present in water samples from the surface, 50 m, and 400 meters depth; but that the concentrations are too low to allow for source determination.

While this data set does not conclusively demonstrate that hydrocarbons found in waters collected from northeast and southwest of the Deepwater Horizon MC 252 well site are derived from a specific source, it does demonstrate that the concentrations of hydrocarbons in these waters are generally low. These data further indicate that concentrations of TPH and PAH_{16} from waters collected from stations 142 n. miles southeast of the wellhead and from stations 40 and 45 n. miles northeast of the wellhead are not substantially different.

Additional work is needed to better understand the fate and transport of hydrocarbons within the deeper waters of the Northern Gulf of Mexico. As well, a more complete and robust understanding of the fate and transport of hydrocarbons at depths will be critical to evaluating and correlating hydrocarbons found throughout the water column with the varying sources of hydrocarbons, both natural and anthropogenic, that are present in the waters of the northern Gulf of Mexico.

Table 1. Analysis of WEATHERBIRD II Data from three stations

Station	Lat.	Long.	Date	Sample Name	Depth (m)	Total Petroleum Hydrocarbons (C9-C44) mg/L	RL[1] mg/L	TPAH-16[2] ng/L	RL ng/L	LMW[3] TPAH-16 ng/L	HMW[4] TPAH-16 ng/L
Slick 1	29 13.82	87 49.85	5/26/10	-0	0	0.306	0.00109	20.5	10.9	ND	20.5
Slick 1	29 13.82	87 49.85	5/26/10	-1	50	0.480	0.00123		12.3	ND	ND
Slick 1	29 13.82	87 49.85	5/26/10	-2	100	0.114	0.00114		11.4	ND	ND
Slick 1	29 13.82	87 49.85	5/26/10	-3	150	0.184	0.00122		12.2	ND	ND
Slick 1	29 13.82	87 49.85	5/26/10	-4	200	0.206	0.00118		11.8	ND	ND
Slick 1	29 13.82	87 49.85	5/26/10	-5	250	0.232	0.00118		11.8	ND	ND
Slick 1	29 13.82	87 49.85	5/26/10	-7	350	0.203	0.00118	48.68	11.8	48.68	ND
Slick 1	29 13.82	87 49.85	5/26/10	-8	400	0.180	0.00120		12.0	ND	ND
Slick 1	29 13.82	87 49.85	5/26/10	-9	425	0.235	0.00112	17.7	11.2	17.7	ND
01	27 28.67	86 06.50	5/23/10	-0	0	0.349	0.00118		11.8	ND	ND
01	27 28.67	86 06.50	5/23/10	-10	100	0.259	0.00116		11.6	ND	ND
01	27 28.67	86 06.50	5/23/10	-9	200	0.257	0.00108	79.17	10.8	79.17	ND
01	27 28.67	86 06.50	5/23/10	-8	300	0.163	0.00119		11.9	ND	ND
01	27 28.67	86 06.50	5/23/10	-7	400	0.112	0.00119		11.9	ND	ND
01	27 28.67	86 06.50	5/23/10	-6	500	0.140	0.00118		11.8	ND	ND
07	29 15.20	87 44.07	5/25/10	-0	0	0.425	0.00114	36.31	11.4	36.31	ND
07	29 15.20	87 44.07	5/25/10	-1	50	0.174	0.00127		12.7	ND	ND
07	29 15.20	87 44.07	5/25/10	-2	100	0.237	0.00128		12.8	ND	ND
07	29 15.20	87 44.07	5/25/10	-3	150	0.189	0.00114		11.4	ND	ND
07	29 15.20	87 44.07	5/25/10	-4	200	0.153	0.00122		12.2	ND	ND
07	29 15.20	87 44.07	5/25/10	-5	250	0.210	0.00122	28.2	12.2	28.2	ND
07	29 15.20	87 44.07	5/25/10	-6	300	0.404	0.00119		11.9	ND	ND
07	29 15.20	87 44.07	5/25/10	-7	350	0.085	0.00119		11.9	ND	ND
07	29 15.20	87 44.07	5/25/10	-8	400	0.141	0.00111		11.1	ND	ND
07	29 15.20	87 44.07	5/25/10	-9	423	0.146	0.00120		12.0	ND	ND

Notes: 1 – Reporting Limit, 2 – Summation of 16 EPA Priority Pollutants, 3 – LMW – 2+3 Ring PAHs, 4 – HMW – 4+5+6 ring PAHs

Table 2. Statistics of Total Petroleum Hydrocarbons (TPHs) in 25 samples obtained from three locations where WEATHERBIRD II collected water.

Station	Count	Median	Standard Deviation	Minimum	Maximum
All Stations Combined	25	0.203	0.10	0.085	0.480
Station DHS-01	6	0.210	0.09	0.112	0.349
Station DHS-07	10	0.182	0.11	0.085	0.425
“Slick1”	9	0.206	0.10	0.114	0.480

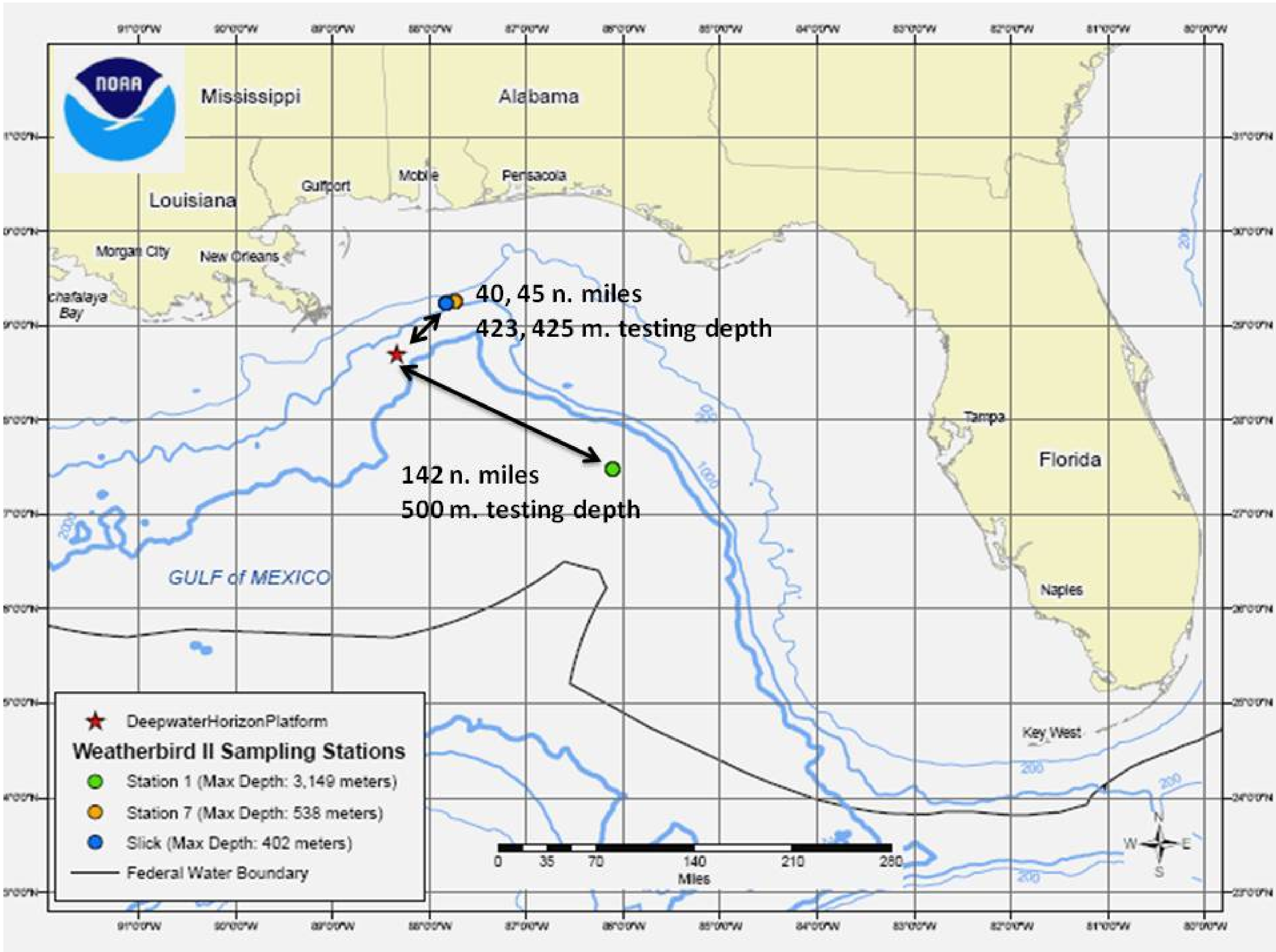


Figure 1 Sampling locations tested by NOAA from the R/V WEATHERBIRD II cruise.

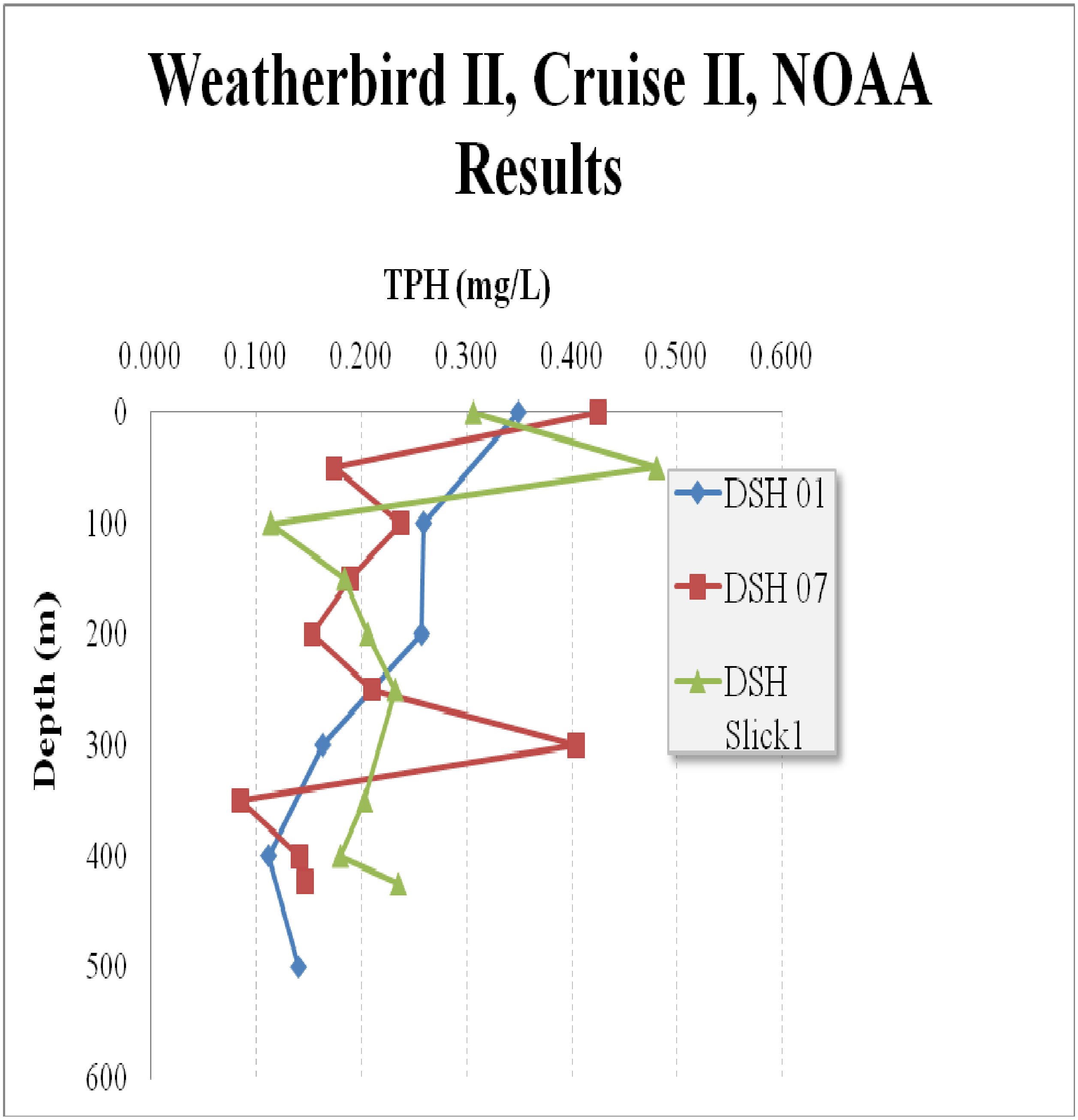


Figure 2. TPH concentrations at depth for three WEATHERBIRD II stations

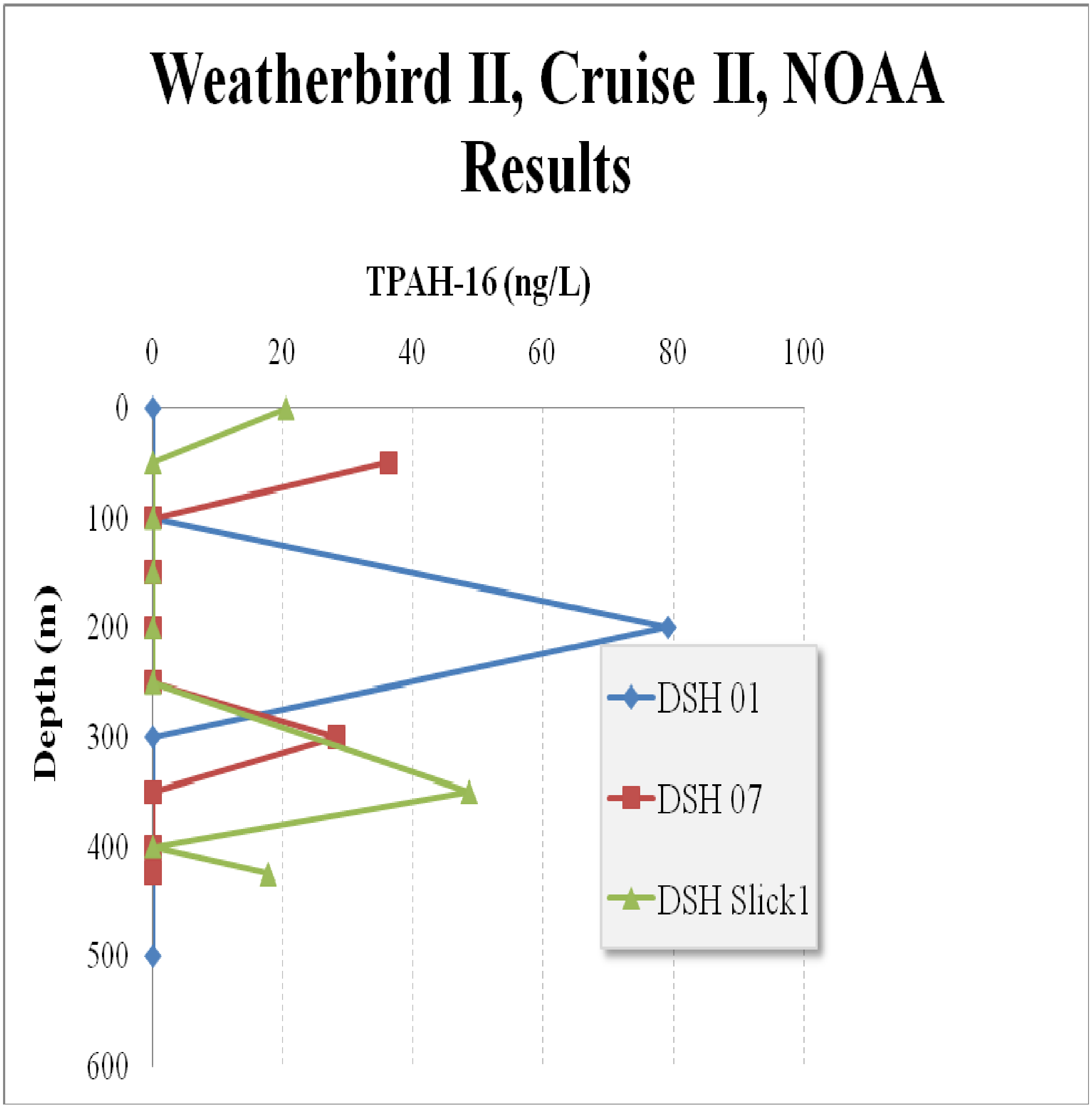
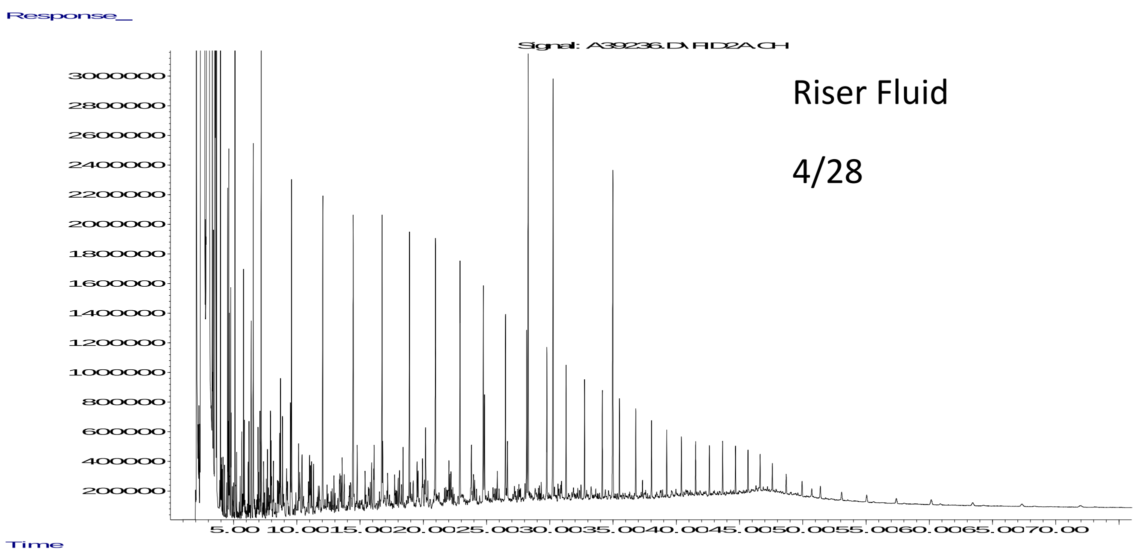
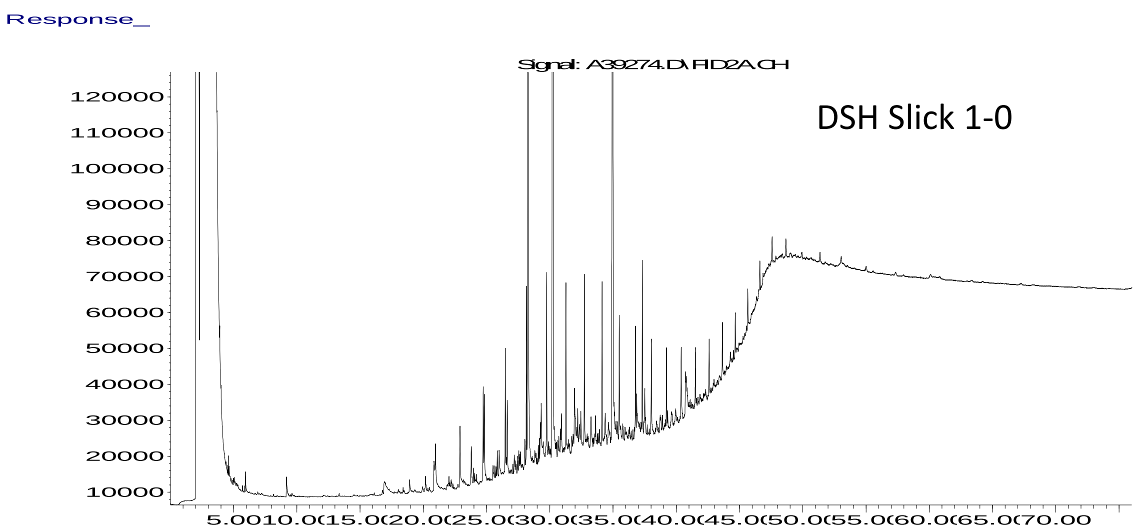


Figure 3. Total PAH compounds measured at depth in three WEATHERBIRD II stations

Figure 4. Chromatographic Comparisons of Source Oil, Slick 1-0 (Surface), and DSH 01-10 (100 m)





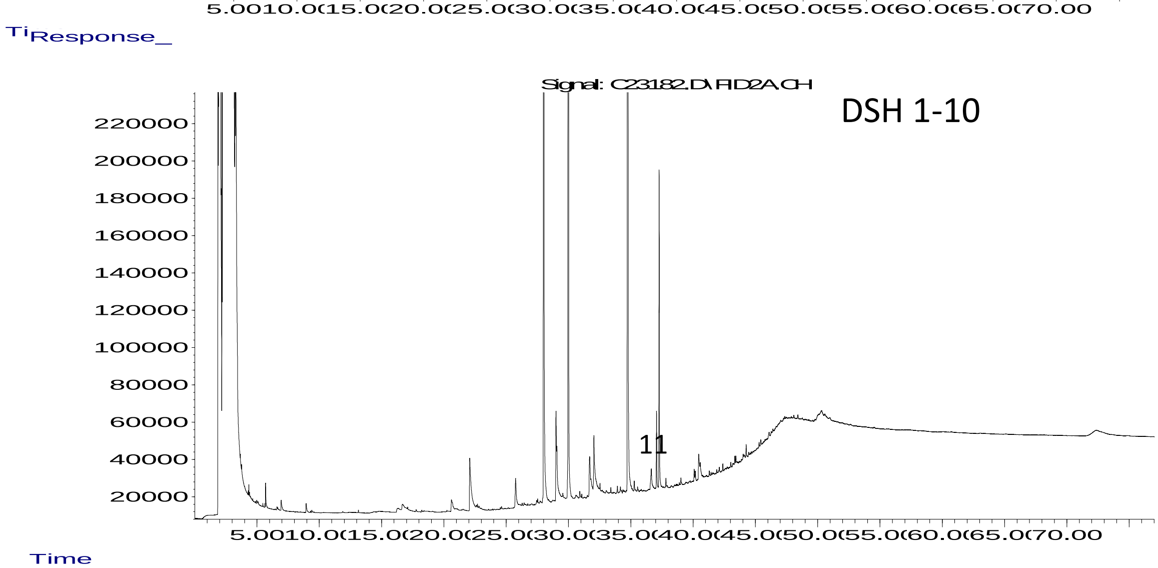
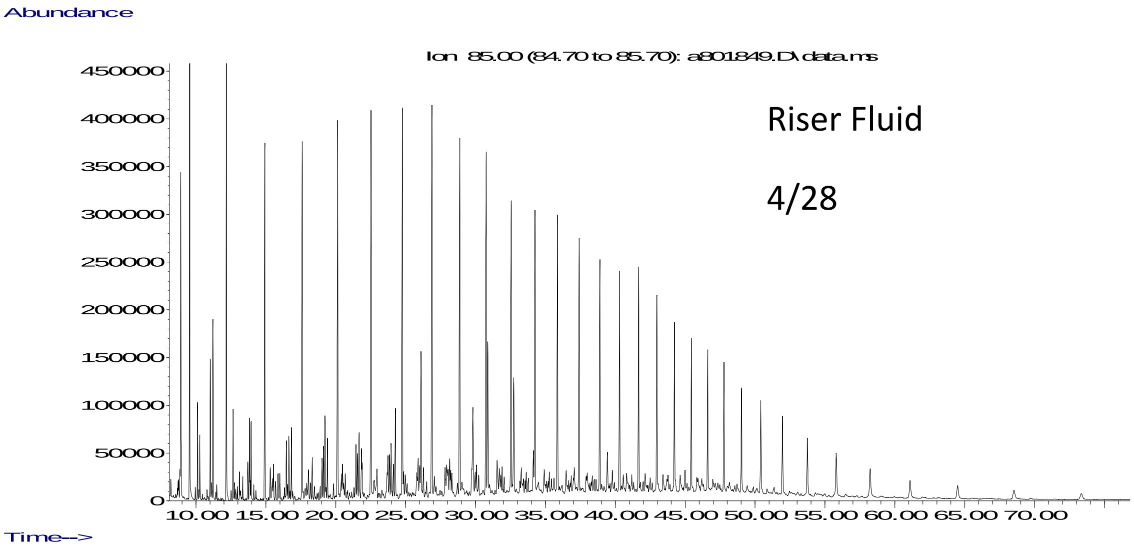
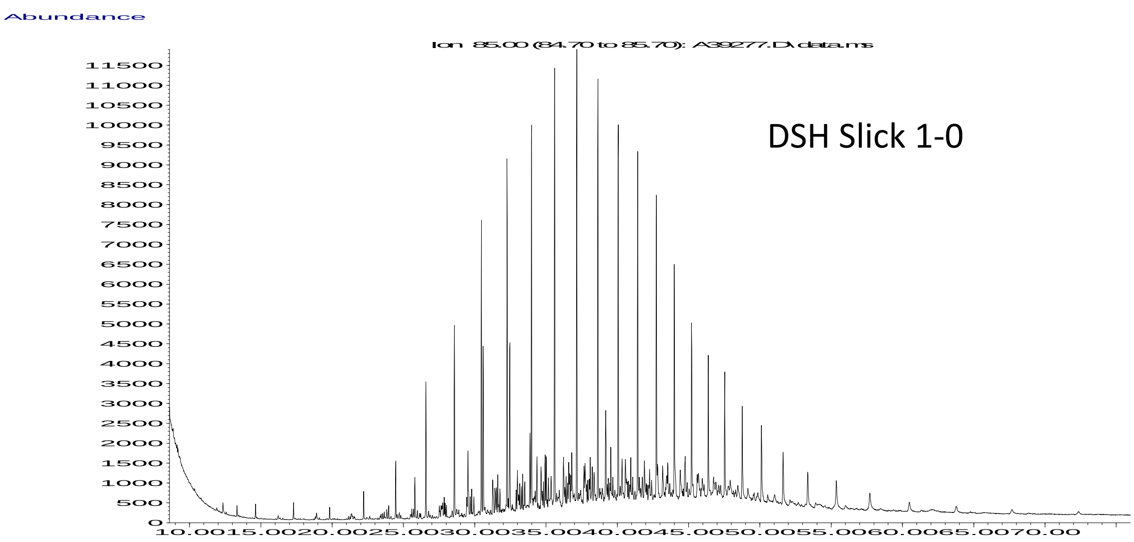


Fig 4 (cont.) m/z 85





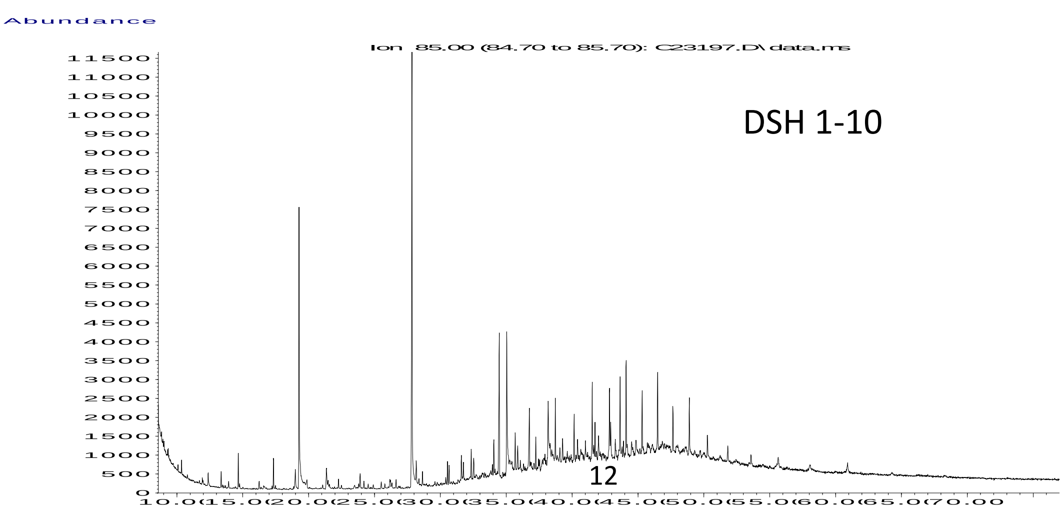
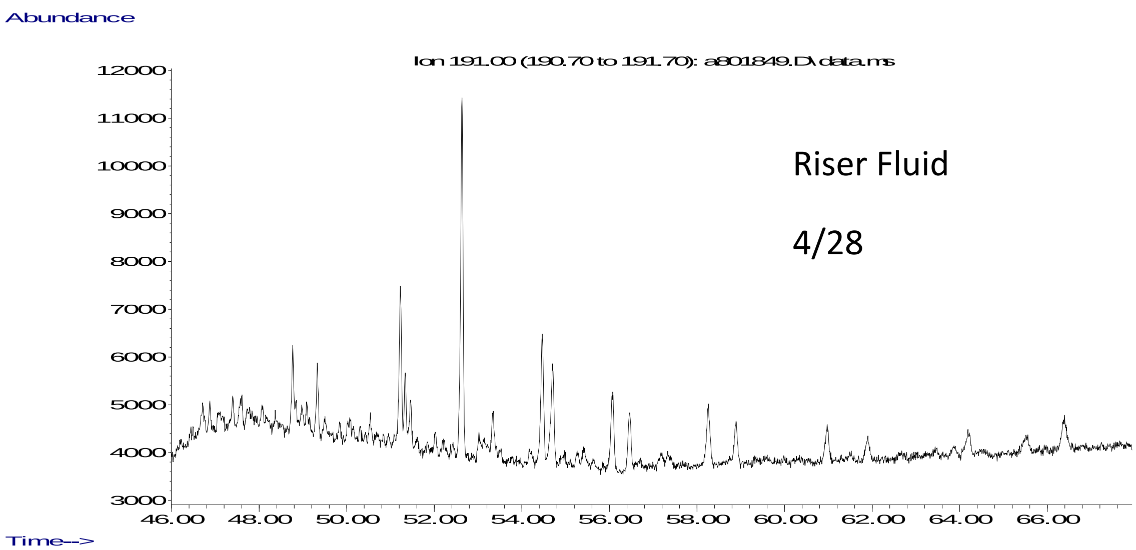
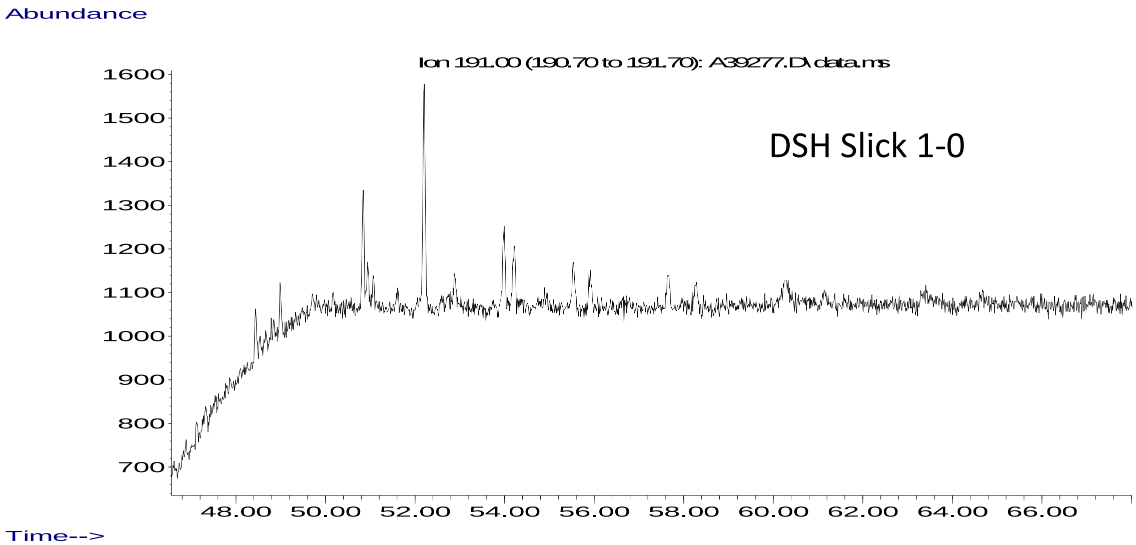


Fig.4 (cont.) m/z 191





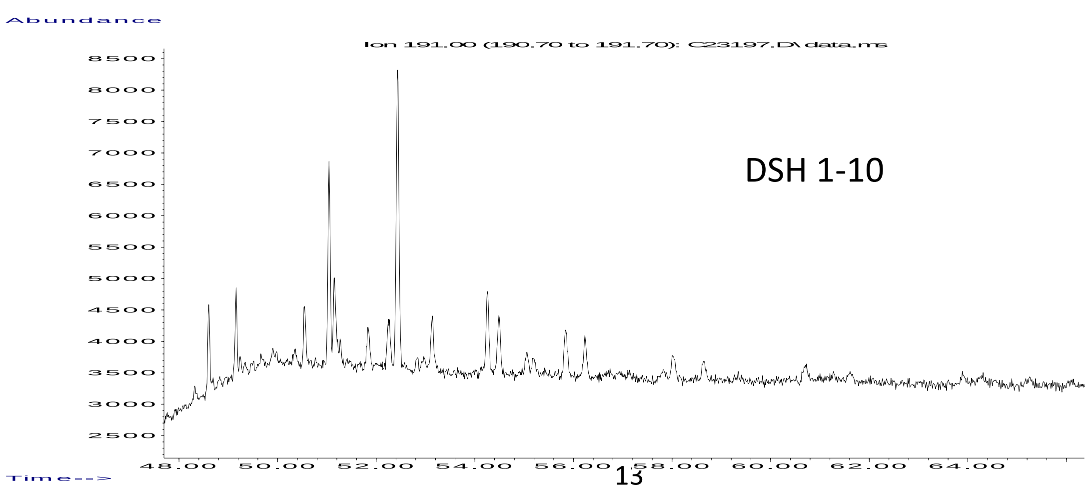
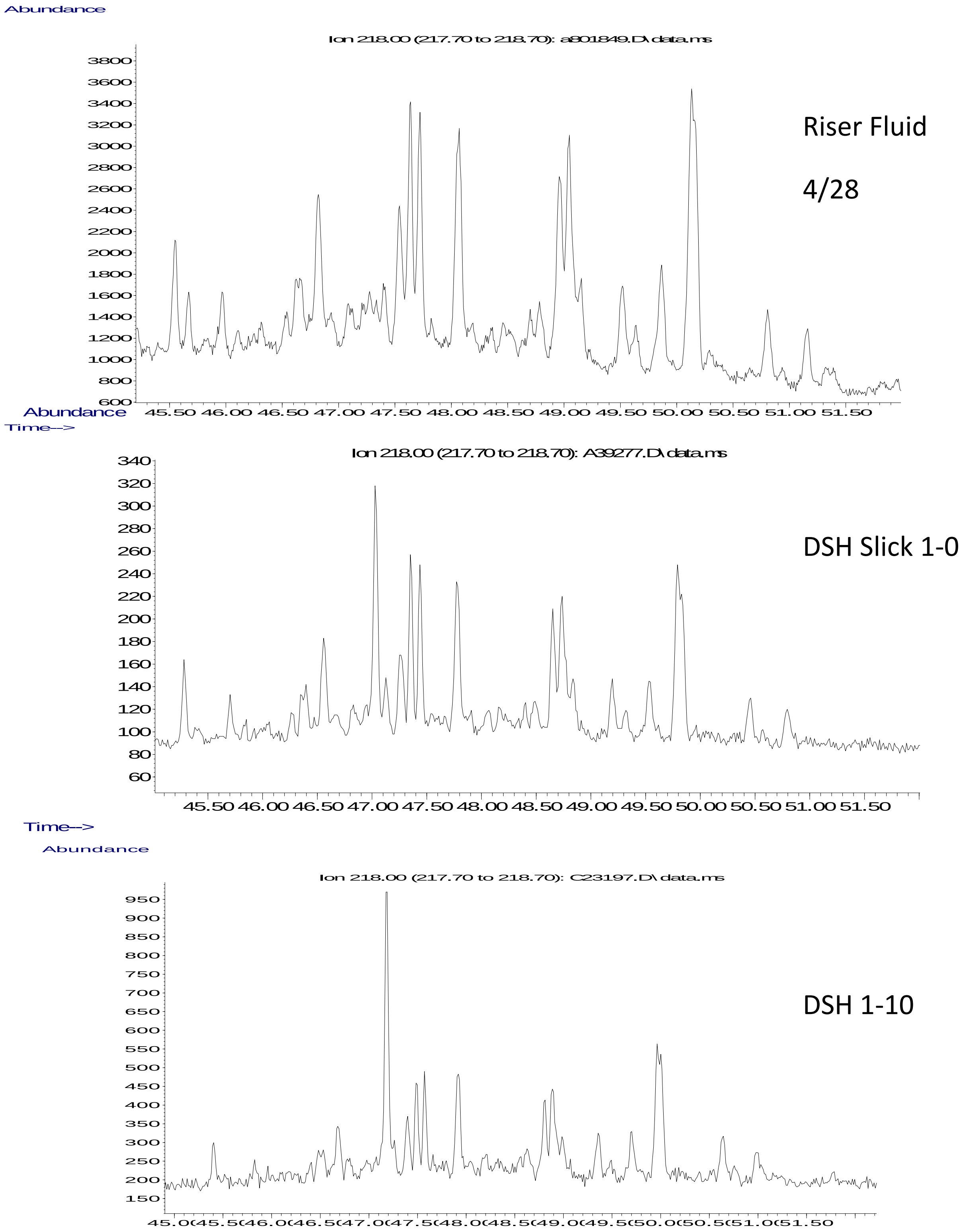


Fig. 4 (cont.) m/z 217



Oil Spills in Marshes

PLANNING & RESPONSE CONSIDERATIONS

September 2013

DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Ocean Service · Office of Response and Restoration

AMERICAN PETROLEUM INSTITUTE

Oil Spills in Marshes

PLANNING & RESPONSE CONSIDERATIONS

September 2013

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DEPARTMENT OF COMMERCE
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INTRODUCTION

This report is intended to assist those who work in spill response and planning where fresh and salt marshes are at risk of oil spills. By understanding the basics of the ecology of marshes and learning from past oil spills in marshes, we can better plan for, protect, and make appropriate decisions for how to respond to future oil spills. Along coastal areas, marshes occur in intertidal to supratidal zones, and the marsh fringe is often contaminated by spills on water. In many areas of the country, pipelines cross under, through, or adjacent to marshes, making them at risk of interior oiling.

Marshes provide many important ecological services and functions and are habitat to many species. When an oil spill affects these habitats, impacts can be severe; however, impacts from inappropriate response methods can increase these impacts and slow overall recovery.

This report is intended to be a technical "job-aid" for spill response scientists. Our goal was to summarize as much of the scientific literature and experience at past spills in a format that balances between too much detail and too many generalizations. Every spill is a unique combination of conditions–oil type, amount of oil, location of oiling, extent of oiling on the soils and vegetation, vegetation types, time of year, presence of species of concern, degree of exposure to natural removal processes, etc. Responders have to evaluate all of these factors and make a decision on the best course of action, *quickly*. We don't have the ready answer for how to respond for every spill. However, we hope that we have provided the reader with practical and useful information gleaned from a large number of studies to help them make informed decisions.

We have organized the topics by chapter, with all the references provided at the end of each chapter. Chapter 1, *Marsh Ecology*, provides an overview of marshes and their associated communities. Chapter 2, *Oil Toxicity and Effects on Marshes*, provides information on oil types and summarizes what we know about how oil affects marsh vegetation. In Chapter 3, *Response*, we discuss what is known on the effectiveness and effects of the different response options appropriate for marshes. Lastly, Chapter 4, *Case Studies*, includes four of the important case studies from which we have learned so much.

We acknowledge associates who reviewed drafts of this report, in alphabetic order: Rene Bernier, Robert Castle, Robyn Conmy, Rebecca Hoff, Jim Jeansonne, Alan Mearns, Irv Mendelssohn, Ed Owens, Heather Parker-Hall, Gary Shigenaka, Ruth Yender, and Scott Zengel. Wendy Early and Joe Holmes of Research Planning, Inc. prepared the text, graphics, and bibliographic database. The American Petroleum Institute funded the work by Research Planning, Inc. NOAA provided funding for its staff.

CHAPTER 1. MARSH ECOLOGY

Key Points

- Marshes are wetlands dominated by emergent herbaceous vegetation that are regularly, frequently, or continually flooded.
- Marshes are highly productive ecosystems that support a complex food chain of plants, microbes, and animals.
- Marshes vary widely in type of vegetation, soils, inundation frequency, salt tolerance, and seasonality.

What are Marshes?

The word "marsh" describes a wide range of habitats. In general, marshes are wetlands that are dominated by herbaceous (in contrast to woody), "emergent" vegetation where the vegetation is erect and extends above the water or very wet soils. There are many different types of marshes, ranging from freshwater to saltwater, but all are inundated with water for extended periods of time or on a regular basis. Marshes can be coastal or inland, connected to a water body or isolated, and are generally fed by surface water, although many are also fed by groundwater. Marsh plants have adaptations that allow them to grow in waterlogged soils; vegetation growing in salt water has adaptations to deal with salt stress.

Marshes support a rich and diverse flora and fauna, serving as important nesting, breeding, spawning, rearing, and feeding habitats for many species of birds, mammals, reptiles, amphibians, fish, shellfish, and other invertebrates. They also provide many ecological services, including primary production, food web support, nutrient recycling, water filtration, sediment and storm water retention, shoreline stabilization, storm-surge protection, and soil development. Plates 1 and 2 show representative plant and animal species in marshes.

Plate 1: Representative marsh plants. All images reproduced with permission, with rights reserved. A: Green arrow arum (R.A. Howard, Smithsonian Institute). B: Pacific silverweed (Arthur Haines). C: Smooth cordgrass stem with salt meadow cordgrass behind (Sandy Richard). D: Salt crystals on smooth cordgrass stem (Sandy Richard). E: Virginia glasswort (Sandy Richard). F: Wild rice (Eli Sagor).

Plate 2: Representative marsh fauna. All images reproduced with permission, with rights reserved. A: Blue crab (Brian Henderson). B: Light-footed clapper rail (Nick Chill). C: Juvenile chinook salmon (NOAA). D: Hine's emerald dragonfly (P. Burton/USFWS). E: Ruddy ducks (Tom Koerner/USFWS). F: Salt marsh harvest mouse (Judy Irving). G: Gulf killifish (Dr. Stephen "Ash" Bullard). H: Whooping crane (Mehgan Murphy).

Types of Marshes

Freshwater Non-Tidal Marshes

Freshwater, non-tidal marshes are common, widespread, and diverse. They are similar in that they are dominated by grasses and sedges, but otherwise differ in their geologic origins, hydrology, and size (Mitsch and Gosselink 1986). They are often found in poorly drained depressions or basins, near streams, rivers, ponds, and lakes, in oxbows, on floodplains, on deltas, and at the base of steep slopes (Fretwell et al. 1996). Freshwater marshes can be permanently or periodically flooded with inches to feet of water, and some may dry out completely on a seasonal or periodic basis. Water levels are controlled both directly and indirectly by precipitation, with many marshes intercepting flood waters from lakes and rivers, surface runoff, or groundwater (Fretwell et al. 1996).

Freshwater, non-tidal marshes are found throughout the United States and Canada and include prairie potholes, wet meadows, wet prairies, playas, and vernal pools. Prairie potholes are numerous, shallow depressions associated with the formerly glaciated landscape of central North America, particularly Iowa, Wisconsin, Minnesota, and North and South Dakota (van Der Valk and Pederson 2003). Wet meadows and wet prairies are grasslands with very wet soils but without standing water most of the year that are common to the Midwest and southeastern United States. Playas are circular, shallow depressions that are typically found in the southwestern United States, particularly in northern Texas and New Mexico (Mitsch and Gosselink 1986; Tiner et al. 2002). Vernal pools are small, seasonally flooded wetlands that dry up completely in the summer and are found throughout the United States, but occur in the highest numbers on the Pacific coast (Zedler 2003). The Florida Everglades contain the largest single freshwater marsh system in the United States (Mitsch and Gosselink 1986). Although each of these systems has unique features, they share characteristic soils, vegetation, and wildlife.

Soils in freshwater non-tidal marshes are typically alkaline, highly organic, mineral soils of sand, silt, and clay with high concentrations of calcium. Nutrient levels in the soils are high, resulting in highly active bacterial communities that rapidly decompose vegetative litter and fix nitrogen (Mitsch and Gosselink 1986). They vary in exposure to physical processes such as water currents and waves.

Although geographically and geologically diverse, freshwater non-tidal marshes are dominated by similar types of grasses, sedges, rushes, and other water-adapted plants. Dominant grasses include common reed (*Phragmites australis*), prairie cordgrass (*Spartina pectinata*), wild rice (*Zizania aquatic*), and maidencane (*Panicum hemitomon*). Typical sedges include *Carex* spp., *Cladium* spp., and the bulrushes (*Scirpus* spp.). Other common plants include various rushes (*Juncus* spp.), cattails (*Typha* spp.), arrowhead (*Sagittaria* spp.), pickerelweed (*Pontederia cordata*), and horsetail (*Equisetum* spp.) (Mitsch and Gosselink 1986)[1].

These marshes provide important habitat for migrating, breeding, and overwintering birds. According to Smith et al. (1964), in Tiner et al. (2002), over half of North America's waterfowl are produced in the prairie pothole region in an average year, while playas provide overwintering grounds for between 1 to 3 million birds, or greater than 90% of the region's waterfowl. Numerous species of reptiles and amphibians also depend on these habitats to breed and for refuge, as do many mammals, including muskrats (*Ondatra zibethicus)*, weasels (*Mustela frenata* and *M. nivalis*), mink (*Mustela vison*), and raccoons (*Procyon lotor*) (Haukos and Smith 1992).

Tidally Influenced Marshes

Tidally influenced marshes represent a salinity continuum from freshwater to fully marine waters with several different salinity regimes in between. For the purposes of this document, tidally influenced marshes will be divided into tidal freshwater marshes and saltwater marshes.

Tidal Freshwater Marshes

Tidal freshwater marshes occur close enough to the coast to undergo daily changes in water levels driven by tides, but whose waters are fresh, with salinity less than 0.5 parts per thousand (ppt). They occur in the uppermost portion of the estuarine zone. Tidal freshwater marshes can experience significant tidal ranges, often of a greater amplitude than those tides experienced at the mouth of the river due to constriction of the water as it moves inland (Mitsch and Gosselink 1986; Odum 1988).

Tidal freshwater marshes can be found on the Atlantic, Pacific, and Gulf coasts of North America, and are usually associated with large river systems (Leck et al. 2009; Mitsch and Gosselink 1986; Odum 1988). They are most extensive on the middle and southeast Atlantic coasts, northern Gulf of Mexico coast, and in Alaska. On the west coast, generally steep topography and mountains limit the size and drainage of the estuaries, leaving few areas with broad drowned river basins that permit the development of extensive freshwater systems. Consequently, the only extensive tidal freshwater marshes are found in San Francisco Bay Delta, Columbia River, and Puget Sound (Leck et al. 2009).

[1] All plant names are from the USDA Plant Database (2013).

Sediments in tidal freshwater marshes typically contain clay, silt, and fine organic matter with minor amounts of sand that have been deposited from upriver and terrestrial sources (Odum et al. 1984). The amount of organic material varies greatly, with Atlantic and Gulf coast sediments containing between 10 to 40% organic matter, and west coast sediments ranging from 5% to around 60% (Thom et al. 2002; Josselyn 1983).

Tidal freshwater marshes are characterized by salt-intolerant plant species, typically a diverse community of emergent grasses, sedges, rushes, and herbaceous flowering plants. Typical plants in Atlantic coast tidal freshwater marshes include wild rice, cattails, and green arrow arum (*Peltandra virginica*), as well as pickerelweed, and broadleaf arrowhead (*Sagittaria latifolia*). On the Pacific coast, typical plant species include mountain rush (*Juncus arcticus*), Pacific silverweed (*Argentina egedii*), hardstem bulrush (*Schoenoplectus acutus*), and cattails. Tidal freshwater marsh plant communities are highly influenced by flooding duration, changes in salinity and/or precipitation, and changes in elevation as well as other factors, and vary seasonally, between years, and over longer time frames (Leck et al. 2009). The marsh fringe can be exposed to riverine and tidal currents and some wave action, whereas the inner marsh is very sheltered.

Because tidal freshwater marshes contain such a wide diversity of habitats and plant communities, they support many species of birds, mammals, reptiles, amphibians, fish, and invertebrates. More birds use tidal freshwater marshes for breeding, nesting, rearing, and feeding than any other type of marsh. Likewise, numerous species of fish use these marshes as breeding, spawning, and nursery grounds, ranging from year round residents like sunfishes, minnows, and catfish, to anadromous fish such as salmon, herring, and shad (Mitsch and Gosselink 1986).

Tidal Saltwater Marshes

There are several types of tidal saltwater marshes, including salt, brackish, and intermediate marshes. They are defined by their average salinity. Salt marshes are regularly flooded by salt water, while brackish and intermediate marshes experience irregular tidal flooding. The varying tidal regime influences the composition of the plant community found within each. For the purposes of this document, these specific types of saltwater marshes will be referred to collectively as salt marshes.

Salt marshes are tidally influenced and experience salinities ranging from 0.5 ppt up to seawater (≥30 ppt). The salinity gradient is nearly continuous from the ocean to the head of the saltwater intrusion into the estuary, until the saltwater signature is drowned by the inflow of freshwater. Tidal ranges in salt marshes are from less than 0.5 meters (m) on the Gulf Coast, to 2-3 m on the East Coast, and in some areas of the West coast, greater than 3 m (Pennings and Bertness 2001; Seliskar and Gallagher 1983). Salt marshes have many adaptations to tolerate salt stress, as listed in Table 1-1.

Table 1-1. Adaptations of salt marsh plants to salt stress (modified from Tiner 1999).

Adaptation Type	Examples
Morphological	Salt secretion glands (to eliminate excess salt; see Plate 1D) Succulent stems and leaves (increased water retention to maintain internal salt balance Waxy leaf coatings (to minimize contact with sea water) Salt concentration in specialized hairs Reduced leaves (to minimize exposure to salt and evapotranspiration)
Physiological	Salt exclusion (reduced salt update by roots) High ion update (lowers osmotic potential of cell sap) Dilution of salts Accumulation of salt in cell vacuoles
Other	Salt stress avoidance (by occupying higher levels of salt marsh) Periodic shedding of salt-saturated organs

Salt marshes are found on all tidally influenced coasts of the United States, but the vast majority of the nation's salt marshes (97%) are located on the Atlantic and Gulf coasts. 58% of the nation's total salt marsh area is located on the Gulf Coast, while the middle and south Atlantic coast contains 37% of the nation's salt marsh area. Of the Gulf Coast states, Louisiana contains the most salt marsh habitat, with 42% of the nation's total, while South Carolina has the largest total area of salt marsh (>9%) of the Atlantic states. In total, the south Atlantic and Gulf coasts contain nearly 80% of the nation's salt marshes (Field 1991).

In contrast, the Pacific coast (excluding Alaska) has few large saltwater tidal habitats, contributing only 3% of the nation's salt marshes. Of the 3%, 75% of those salt marshes are located in California (Field 1991). As described earlier for tidal freshwater marshes, on the west coast, mountains limit the location and size of the estuaries, with estuaries and lagoon constituting less than 20% of the shoreline (Macdonald 1977).

Salt marsh sediments vary widely in their composition and are determined by the sediment source and tidal current patterns. Sediments may be river silt, organic material, or sand and clay originating from marine sources. Large variations in sediment organic content across regions and within individual marshes can occur as a result of different rates of production and below-ground decomposition (Odum 1988; Zedler and Callaway 2001). The organic content of the sediment, in addition to the elevation and drainage, are more important than the source of mineral sediment in determining marsh productivity (Mitsch and Gosselink 1986).

Salt marshes are characterized by salt-tolerant flowering plants, including salt-tolerant grasses, rushes, and sedges. In salt marshes of the entire east coast and much of the Gulf coast, smooth cordgrass (*Spartina alterniflora*) is the most dominant species. In some Gulf coast marshes, needlegrass rush (*Juncus roemerianus*) is dominant. Other species common in east and Gulf coast salt marshes include salt meadow cordgrass (*Spartina patens*), saltgrass (*Distichlis spicata*), Virginia glasswort (*Salicornia depressa*), and turtleweed (*Batis maritima*) (Mitsch and Gosselink 1986; Odum 1988; Wiegert and Freeman 1990; Zedler and Callaway 2001).

On the Pacific coast, smooth cordgrass is a non-native, invasive species. In the California marshes, California cordgrass (*Spartina foliosa*), pickleweed (*Salicornia* spp.), saltgrass, and turtleweed are common species (Macdonald 1977; Zedler 1982). The plant communities of Oregon, Washington, and Alaska share some species in common with the California marshes, including Virginia glasswort and saltgrass, but have no California cordgrass or turtleweed (Zedler 1982). Alkaligrass (*Puccinellia* spp.) and extensive stands of sedges (*Carex* spp., *Scirpus validus, Scirpus americanus*) and rushes are common (Macdonald 1977; Seliskar and Gallagher 1983).

Salt marsh species and forms differ depending on the frequency and duration of flooding, as shown in Figure 1-1. The lower, regularly flooded zone ("low marsh") is usually dominated by one species, such as cordgrass along the Atlantic and Gulf coasts. On the Pacific coast, the low marsh may be dominated by nearly monotypic stands of Lyngbye's sedge (*Carex lyngbyei*), the northwest analogue to the cordgrass marshes of the Atlantic and Gulf coasts. Or, as depicted in Figure 1-1, it may host a mixed community of plants that includes saltgrass, marsh jaumea (*Jaumea carnosa*), and pickleweed, among others (Seliskar and Gallagher 1983). The higher, irregularly flooded zone ("high marsh") has more diverse vegetation because the plants have less inundation stress and fewer fluctuations in salinity and temperature than the plants in the low marsh. The salt marsh fringe is exposed to tidal currents and wave action, whereas the inner marsh is sheltered from these processes.

Salt marshes are some of the most productive ecosystems in the world, typically exceeding the production of the most successful agricultural activities. These highly productive habitats support abundant invertebrates, fish, and wildlife, and produce large quantities of organic material that play an important role in the marsh food web. They are important feeding, breeding, nesting, and rearing

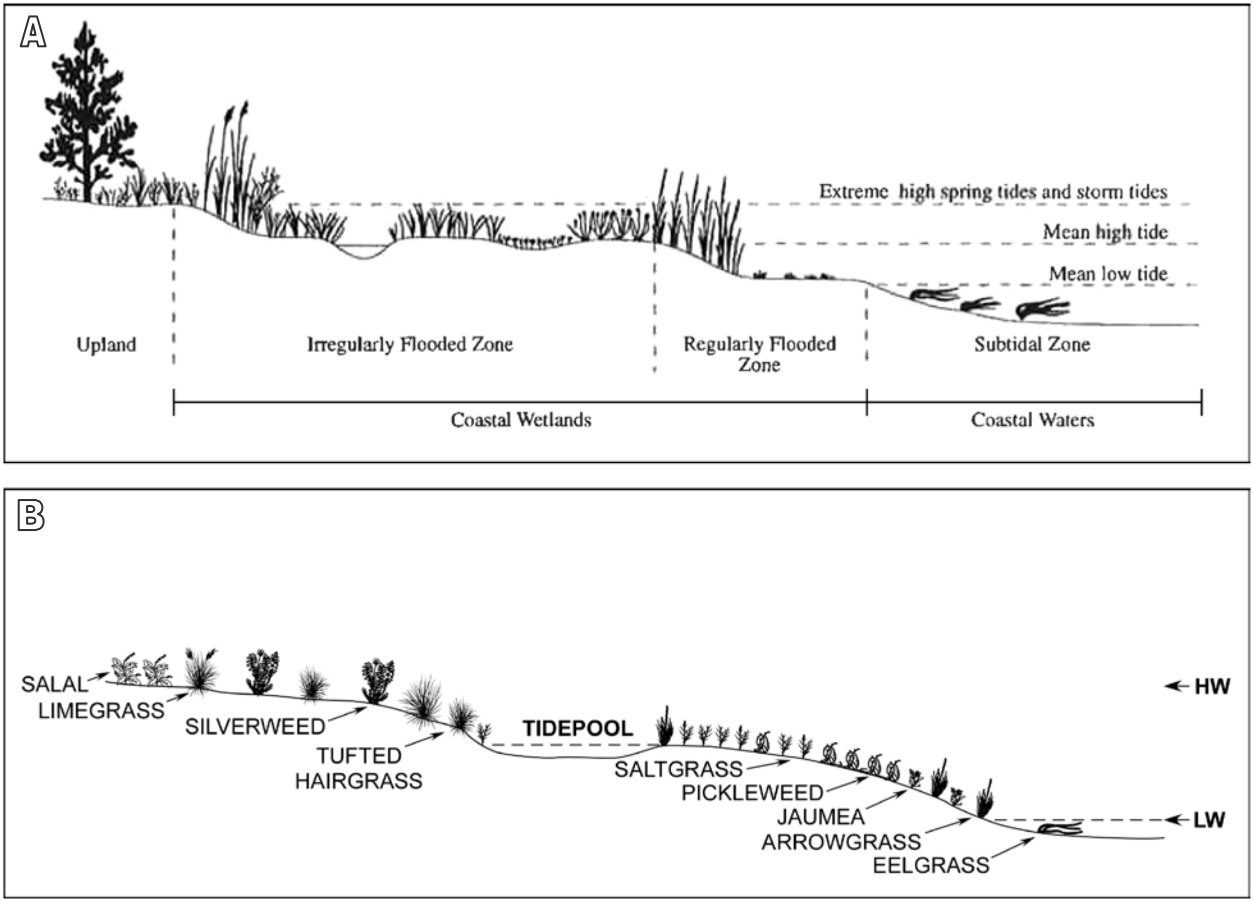

Figure 1-1. Tidal salt marsh zonation. A: Mid-Atlantic salt marshes based on frequency of tidal flooding. The low marsh is flooded at least one daily; the high marsh is flooded less often (from Tiner and Burke 1995). B: Typical zonation of marsh plants in a Pacific Northwest tidal salt marsh. The lateral extent of the zones depends on the slope and may range from a few meters to hundreds of meters (from Seliskar and Gallagher 1983).

habitat for numerous fish, mammals, invertebrates (e.g., crabs, shrimp, insects), and birds, including migratory waterfowl. Salt marshes are particularly valuable habitat as nurseries for commercial and recreationally important fish and shellfish species, especially for native and at-risk species (Gewant and Bollens 2012).

General Life History Information

Annuals vs. Perennials

Annuals are plants that complete their entire life cycle within a year. They germinate, flower, produce flowers, and die within one year. All of their roots, stems, and leaves die annually.

Perennials live for two or more years, overwintering and producing flowers and seeds from the same rootstock. In some perennials, the leaves, stems, and flowers die back in the fall or winter, and the plant regrows in the spring from the rootstock. In other perennials, the plant retains its aboveground structures year round. Perennials can reproduce by seeds, but have evolved a variety of vegetative cloning strategies, including the production of bulbs, tubers, woody crowns, and rhizomes (thick parts of plants that grow horizontally under or on the ground and send out roots and shoots). Vegetative cloning strategies such as rhizome growth allow the development of dense, single-species stands of vegetation as seen in the smooth cordgrass-dominated salt marshes of the east and Gulf coasts.

Seasonality

As discussed earlier, annual plants complete their entire life cycle in one year or less. Some summer annuals sprout, flower, seed, and die in less than one month. Other annual plants may take several months to complete their life cycle. Their seeds persist until the environmental conditions are right for germination, thus starting a new generation. Annual plants come in two forms: summer and winter. Summer annuals germinate and die in a single season (spring, summer, or fall). Winter annuals germinate in the fall or winter, bloom in the winter or early spring, and then die once they set seeds. The seeds of annuals are the sole source of the next year's growth.

Perennial plants, on the other hand, live through multiple seasons and years. In warm climates, perennials may grow year round, while in climates with pronounced seasonality, growth is limited to the growing season. In these instances, the perennials enter a period of dormancy with associated senescence (die back) of the aboveground vegetation. Other perennials may not be truly dormant, but just stop or slow growth if the temperatures are too low or there isn't enough light. In these instances, once the environmental conditions are correct, the plant resumes growth. Most vegetative growth of plants in the tidal marshes of the east and Gulf coasts occurs from March to November (Eleuterius 1990). However, *S. alterniflora*, the dominant plant in east and Gulf coast saltwater tidal marshes, grows year round, but more slowly in the winter months (Gosselink 1984).

Most of the plants found in freshwater non-tidal and tidal habitats are a mix of annuals and perennials. These marshes exhibit pronounced seasonality with changes in plant community dominance as the seasons progress. As the annuals flower and die, their seeds are dispersed to lie dormant until the environmental conditions are right for germination, thus starting a new generation. Salt marshes, on the other hand, are dominated by perennial plants, which have adapted to handle the more extreme environment created by high or fluctuating salinities and varying flooding regimes.

Fauna

Marshes support a rich and diverse assortment of animals. The high productivity, diverse habitat structure, and flood regimes of these transitional areas between terrestrial and aquatic habitats attract and support numerous invertebrates, fish, amphibians, reptiles, mammals, and birds. Marshes are critically important habitats for migratory and resident bird species including numerous ducks, wading birds, and shorebirds, and are used by nearly one-third of North American birds for shelter, resting, feeding, nesting, breeding, and rearing habitat (Fretwell et al. 1996 in Stewart, 1996). Nearly two-thirds of the continental United States' waterfowl reproduce in the prairie pothole marshes of the Midwest. In addition, tidally influenced marshes function as the nursery grounds for numerous species important for and as recreational and commercial fisheries including shrimp, crabs, and wide variety of fish species. Freshwater marshes also provide refuge, spawning, and rearing habitat to a variety of amphibians and reptiles including the American alligator, and numerous species of turtle, snakes, and frogs. Common mammals that either live in marshes or visit frequently include muskrats, otters, minks, and raccoons.

Marshes are home to numerous threatened and/or endangered species. In fact, some estimates are that greater than 40% of the nations endangered and threatened species rely directly or indirectly on wetlands for survival (Department of Environmental Conservation, Vermont 2011; Environmental Law Institute 2011). Although the term wetlands encompasses more than just marshes, this statistic illustrates the importance of these habitat types. Examples of threatened and endangered species that rely on marsh habitats include the Everglades snail kite, Lower Keys marsh rabbit, wood stork, chinook salmon, salt marsh harvest mouse, light-footed clapper rail, Yuma clapper rail, Hine's emerald dragonfly, and the whooping crane.

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CHAPTER 2. OIL TOXICITY AND EFFECTS ON MARSHES

Key Points

- Oil type is one of the major factors determining the degree and type of impacts on marshes.
- Lighter oils are more acutely toxic than heavier oils; however, when spilled offshore, light oils are seldom cause extensive damage because they spread into thin slicks.
- Heavy refined oils and most crude oils affect marshes through physical smothering of both leaves and soils. The oil weathering and emulsification prior to landfall reduces the initial toxicity of the oil.
- The extent of oiling on the vegetation is a key factor. If only parts of the leaves are oiled, often the marshes recover quickly, within one growing season.
- Exposure to waves and currents that speed oil removal is another key factor. Other factors include degree of contamination of the soils, time of year, and different sensitivities among plant species.

Oil Groups

Oils can be divided into five groups as shown in Table 2-1 based on their general behavior, persistence, and properties. Each group is defined by a range in specific gravity, defined as the ratio of the mass of the oil to the mass of freshwater, for the same volume and at the same temperature. If the specific gravity of the oil is less than the specific gravity for the receiving water (freshwater = 1.00 at 4°C; seawater = 1.03 at 4°C), it will float on the water surface. API gravity[2] is another property that is often reported and can be used to characterize an oil's behavior.

Factors Affecting the Impacts of Oil on Marsh Vegetation

Oil Type

The type of oil spilled influences the potential type and degree of impacts to marshes because of differences in behavior, persistence, and toxicity. In this section, case histories and summaries are provided to indicate the likely impacts from spills of: 1) light refined products (mostly Group 2 oils because Group 1 oils usually evaporate quickly); 2) light to medium crude oils (mostly Group 3 oils); and 3) heavy crude oil and refined products (Group 4 oils).

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[2] API = (141.5/specific gravity) - 131.5. An API of 10 is equal to a specific gravity of 1.00; an API of 45 is equal to a specific gravity of 0.80. Note that API gravity has an inverse relationship with specific gravity.

Table 2-1. Oil groups and their characteristics.

Group 1: Gasoline products • Specific gravity is less than 0.80; API gravity >45 • Very volatile and highly flammable • Evaporate and dissolve rapidly (in a matter of hours) • Narrow cut fraction with no residues • Low viscosity; spread rapidly into thin sheens • Will penetrate substrates but are not sticky • High acute toxicity to animals and plants
Group 2: Diesel-like Products 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 can have residue after evaporation is complete • Low to moderate viscosity; spreads rapidly into thin slicks; not likely to form stable emulsions • Are more bioavailable than lighter oils (in part because they persist longer), so are more likely to affect animals in water and sediments
Group 3: Medium Crude Oils and Intermediate Products • 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 and plants in water and sediments • Can form stable emulsions and cause long-term effects via smothering or coating
Group 4: Heavy Crude Oils and Residual Products • Specific gravity of 0.95-1.00; API gravity of 10-17.5 • Very little product loss by evaporation or dissolution • Very viscous to semi-solid; may be heated during transport • Can form stable emulsions and become even more viscous • Tend to break into tarballs quickly • Low acute toxicity to 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 on or in sediments
Group 5: Sinking Oils • Specific gravity of >1.00; API gravity <10 • Very little product loss by evaporation or dissolution • Very viscous to semi-solid; may be heated during transport or blended with a diluent that can evaporate once spilled • Low acute toxicity to biota (though may have some toxicity if blended with a lighter, more - toxic diluent) • Penetration into substrates will be limited at first, but can increase over time • Can cause long-term effects via smothering or coating, and as residues on or in sediments

Light Refined Oil Products

Light refined products, such as jet fuel, kerosene, No. 2 fuel oil, home heating oil, and diesel, have been shown to have the highest acute toxic effects on marsh vegetation. Appendix A is a summary of the results of spill studies and field/greenhouse experiments of light refined products on marshes. These types of oil have low viscosity and high rates of loss by evaporation and dispersion into the water column under even low-to-moderate wave energy. When spilled on open water, they usually spread into thin slicks and sheens and often do not persist long enough to cause significant shoreline oiling. As noted in the case studies discussed below, those spills that did result in extensive plant mortality and long-term impacts involved large volumes released to sheltered waterbodies, resulting in heavy oiling of marsh habitats.

In all the tables in the Appendices, the last column shows what the study results reported as years to "recovery," which usually meant vegetative growth (mostly aboveground biomass or stem density) that is comparable to unoiled vegetation. It should be noted that this definition of recovery is incomplete because it is based on just one metric of marsh services and functions. Very few studies considered other metrics, particularly animals living in the marsh.

The 185,000 gallons of No. 2 fuel oil from the T/B *Florida* in 1969 in Buzzards Bay, Massachusetts is one of the most famous spills in the literature, partially because many plants and animals were killed, but also because it was close to the Woods Hole Oceanographic Institute where many then- or now-famous scientists became involved in studies of the spill for nearly 40 years. Thus, it is discussed in detail as one of the case studies included in Chapter 4.

In 1974, there was another spill in Buzzards Bay of 3.17 million gallons of No. 2 fuel oil from the T/B *Bouchard 65* that affected a different marsh and has also been well studied. Three years later, Hampson and Moul (1978) documented complete mortality in heavily oiled marshes and significant erosion of the marsh edge. The number of infaunal species was reduced by 92%. By 1991, Hampson (2000) reported that the salt marsh vegetation had slowly recovered, but the peat substrate had been permanently eroded, leaving only a sand and gravel beach.

Burger (1994) and chapters therein summarized the impacts of a release of 567,000 gallons of No. 2 fuel oil from a pipeline at the Exxon Bayway refinery into the Arthur Kill on 1-2 January 1990. By the first summer, they documented that 7.6 hectares (ha) of mostly *S. alterniflora* had been killed (15% of the affected area), and 2.8 ha were oiled but recovering. There was also high mortality (>67%) of ribbed mussels (*Geukensia demissa*) close to the spill source, and fiddler crab (*Uca* spp.) mortality and sublethal effects were noted. In 1993, after three growing seasons, there was no recovery of most of the dead vegetation (Burger 1994).

Many field and greenhouse experiments where marsh plants were exposed to No. 2 fuel oil (see Appendix A for details) have found that:

- No. 2 fuel oil can be highly toxic to salt marsh vegetation and more toxic than other types of oil under similar exposure conditions.
- The severity of impacts was directly related to the amount of plant covered by the oil. Studies by Booker (1987) supported the hypothesis that oil exposure affected cell membrane permeability, which would reduce tissue viability through an impaired ability to maintain chemical balances and metabolism in the cells.
- There was a dose-response relationship between the degree of oil in the marsh soils and impacts to plants.
- Both direct physical damage to contacted tissues plus translocation of toxic components of the oil from stems to the root system caused death or a reduction in the ability of the root system to regenerate shoots.

However, not all spills of light refined products result in high mortality of vegetation. NOAA responds to many spills of diesel from fishing vessels, where most of the oil quickly spreads into thin slicks and is dispersed or evaporated, such that shoreline oiling is light and rapidly removed by natural processes. The April 2004 Kinder Morgan pipeline spill in a diked marsh in San Francisco Bay, California did not penetrate into the clayey soils along the channel banks, so there was mortality of fish and invertebrates but little plant mortality.

Interpreting the Oil Loading in Field and Greenhouse Experiments
Most experiments report the oil loading in terms of the number of liters per square meter (L/m^2) of oil applied to the surface of the treatment area (field plot or potted plant). Converting this dose to an oil thickness is complicated because of the variable surface area of the vegetation. However, ignoring the surface area of the vegetation, the thicknesses of different doses are:

1 L/m^2 = 0.1 cm   4 L/m^2 = 0.4 cm   8 L/m^2 = 0.8 cm   24 L/m^2 = 2.4 cm (1 inch)

Shoreline Cleanup Assessment Technique (SCAT) thickness terms:
Cover = <0.1 cm   Coat = >0.1 cm to <1 cm   Thick = >1 cm

In summary:

- Light refined products such as No. 2 fuel oil, diesel, kerosene, and jet fuels do have high acute toxicity to marsh plants and associated communities, and there is a strong dose-response relationship.
- Spill events where large amounts of these kinds of oils get transported into and contained within marshes will likely result in plant and fauna mortality.
- Where the rhizomes die (rather than just the vegetation dying back), recovery depends on regrowth from plants outside the oiled area; thus spills affecting large areas may not recover quickly.
- Spills in confined waterways, where the oil is not able to spread out and strands on the shoreline quickly, have the highest risk of impact.
- Offshore spills, small spills, and those where the oil is dispersed by wave action before stranding onshore have a lower risk of impacting sensitive marsh habitats and associated communities.

Light to Medium Crude Oils

Light to medium crude oils can range widely in terms of their fate and effects on marshes, depending on their chemical composition and the degree of weathering prior to stranding on the marsh. Appendix B lists representative spills and experiments to demonstrate the range of impacts under different conditions. There have been several summaries of the literature on the impacts of crude oil on the marshes of U.S. Gulf Coast (Pezeshki et al. 2000; DeLaune et al. 2003; DeLaune and Wright 2011).

Cowell (1969) was the first to note the differences due to weathering of oil at sea on the effects of two large spills of light Kuwait crude in 1967 on U.K. marshes: the spill from the *Chryssi P. Goulandris* that stranded within hours after the release caused much higher mortality of plants and animals than the spill from the *Torrey Canyon* that stranded after eight days of weathering at sea. This effect was also evident at the *Deepwater Horizon* oil spill where oil was released at the seafloor, rose through approximately 1,500 m of water, was treated by dispersants both subsea and on the surface, and had to be transported by wind and currents for 80-300 kilometers (km) through warm Gulf of Mexico waters to reach the shoreline. Those marshes with a thick layer of oil on the marsh vegetation and substrate died; those with moderate oiling appeared to be recovering (Lin and Mendelssohn 2012; pers. observation of the authors; see case history in Chapter 4).

Crude oil releases from pipelines directly into marshes undergo limited weathering processes and thus tend to result in higher mortality and longer recovery times. A spill of 12,600 gallons of Louisiana crude into a brackish marsh in Louisiana in April 1985 caused nearly complete mortality of about 20 ha, and recovery of the vegetation took four years (Mendelssohn et al. 1993; Hester and Mendelssohn 2000). This amount of oil, if evenly spread throughout the 20 ha, would be at a loading of 0.28 liters/square meter (L/m^2), which is much lower than what is normally found to be toxic to plants based on greenhouse experiments (compare with greenhouse studies in Appendix B). Yet, there was extensive mortality, likely because of a lack of chemical weathering before the oil came in contact with the marsh and minimal physical removal processes.

When reviewing the results of the greenhouse and field experiments, it is very important to understand if the oil was weathered prior to oiling and how the oil was applied–because it varies widely. This information is briefly summarized in the various tables in the appendices, but a full understanding can only be gained from review of the methods of each study. These studies also varied in terms of the water level above the plants during oil exposure, the amount of oil applied to the vegetation (or not), and month of exposure, all of which influence how plants respond to oiling.

In summary:

- Crude oils can have both acute, short-term toxicity if relatively fresh oil comes in contact with the plants and if most of the plant surface is covered by the oil, but recovery often occurs quickly. These effects are reduced when oil weathers/emulsifies prior to stranding.
- Crude oils can also cause physical smothering, as discussed in the next section on heavy oils.
- It is difficult to summarize the impacts of crude oil spills on marshes because of the range of spill conditions and the importance of other factors.

- Most of the factors controlling the initial impacts and recovery rates from exposure to crude oils are discussed later in this chapter.

Heavy Crude Oils and Refined Oil Products

Heavy crude oils (including crude oils derived from tar sands) and heavy refined oil products, such as heavy fuel oil, Bunker C, No. 6 fuel oil, and intermediate fuel oils (IFO) 180 and 380, are thought to affect marsh vegetation primarily via physical effects from coating and smothering of the vegetation and/or soil surface because they generally have low amounts of acutely toxic compounds. Twelve studies of these kinds of spills were identified (summarized in Appendix C), and some of the key points are discussed below.

The February 1970 spill of nearly 3 million gallons of Bunker C oil from the T/V *Arrow* in Chedabucto Bay, Nova Scotia, heavily oiled a sheltered lagoon containing *S. alterniflora* marshes and mud flats. No cleanup was conducted, thus there was chronic re-oiling over time. There was high mortality of the vegetation and periwinkles (*Littorina littorea*), which took over six years to recover (Thomas 1978). Soft-shell clams (*Mya arenaria*) in the adjacent tidal flat showed initial high mortality. This spill showed that chronic re-oiling and persistence of heavy oil accumulations can have long-term impacts to marsh vegetation and fauna.

The T/V *Golden Robin* spill of Bunker C fuel oil in New Brunswick showed that aggressive manual and mechanical treatment (see Appendix C), even of heavily oiled marshes, can result in slower recovery compared to natural recovery or light treatment (Vandermeulen and Jotcham 1986). Aggressive treatment increased the amount and persistence of oil in the soils. This lesson was learned again during the Bunker C spill from the M/V *Westwood* in British Columbia, where Challenger et al. (2008) documented extensive vegetation damage and increased soil contamination in areas where aggressive oil and soil removal and trampling occurred (at the insistence of local stakeholders), compared to untreated or carefully treated areas.

The barge *STC-101* spill of No. 6 fuel oil in Chesapeake Bay (Hershener and Moore 1977) was one of several studies that showed an increase in net productivity of oiled vegetation. Other spills in marshes that showed a net increase in biomass from light oiling included *Phragmites* (Lin et al. 1999) and *S. alterniflora* (Krebs and Tanner 1981; Li et al. 1990). Although the mechanism by which oil stimulates plant growth is uncertain, Lin et al. (1999) hypothesized that oil in marsh soils may increase microbial N-fixation or shift competitive interactions among species.

Hershener and Moore (1977) found 100% mortality of marsh periwinkles (*Littorina irrorata*) in the heavily oiled marsh and 80% reduction in abundance in the oiled marsh after two growing seasons. Periwinkle recovery is tied to vegetative recovery; juveniles are only able to settle and survive where there are stalks to climb and leaves in which to hide. Thus, penetration into and heavy contamination of marsh soils in a sheltered setting can result in impacts to salt marsh vegetation and communities for years.

There are few studies of the impacts of heavy refined oils in freshwater environments. Burk (1977) studied a heavy fuel oil spill in a freshwater marsh in February (see Appendix C), documenting high mortality of annual species and impacts that lasted at least four years. Perennial species were less affected. Alexander et al. (1981) found that oiled/cut *Typha* along the St. Lawrence River grew taller but didn't flower the first year after the spill, but had normal growth and flowering by the second growing season. Study of the spill of Bunker C into Lake Wabamun in Alberta for two growing seasons indicated that oil exposure during the late growing season in August 2005 and the winter senescent period did not cause large-scale effects on the summer regrowth in 2006 and 2007 for the reed-bed communities, except for some treated sites (Wernick et al. 2009). Spills in freshwater environments, where water-level fluctuations are seasonal rather than daily, have a lower risk of contamination of the marsh soils, unless the oil sinks. Thus there is potential for quick recovery rates, particularly in rivers that have the benefit of continuous water flow to speed natural removal processes. Large lakes can have significant wave energy; small ponds generally do not.

There have been several field or greenhouse oiling experiments using heavy fuel oil. Alexander and Webb (1985) included a No. 6 fuel oil in their field oiling experiments that were mentioned previously and summarized in Appendices A-C. There were slight impacts to vegetation for the 1.5 L/m^2 partial and 2 L/m^2 entire plant applications in May, but only for months 1 and 5 after oiling. By month 12, the oiled plants were no different than the unoiled controls.

Based on the published studies and personal observations at many spills of heavy refined products in marshes, long-term impacts (>2 years) are likely to occur for the following conditions:

1) There is chronic re-oiling;
2) The marsh soils are heavily oiled, either by thick layers on the surface or penetration into the soil;
3) The oil strands very quickly after spillage, thus there is relatively little weathering;
4) The entire plant surface is covered with oil during the growing season; or
5) There has been aggressive treatment that causes damage to roots and mixes oil into the soils.

Relatively short recovery periods (1-2 growing seasons) are likely to occur when:

1) Oiling degree is light;
2) Oiling occurs in the fall or winter when the plants are in senescence;
3) The oil undergoes extensive weathering or emulsification prior to stranding;
4) There is little to no contamination of the marsh soils; or
5) The oiled areas are exposed to waves or currents that speed natural removal rates.

In the next sections, the other factors influencing the degree of impact of oiling of marsh vegetation are summarized.

Extent of Contamination of the Vegetation

As discussed in the previous section, the extent of oil on the vegetation is an important factor in determining the initial impact on vegetation. Although we know that there are important differences between field spills and greenhouse experiments, the greenhouse studies do provide good control to demonstrate this effect. Review of Appendices A-C shows that:

1) When the entire plant and the soil surface is covered with 1.5-2 L/m^2 of light refined oil, there is usually 100% mortality of the aboveground vegetation and sometime high mortality of the entire plant;
2) Similar coverage and loading by heavy refined oils and crude oils in greenhouse experiments result only in a slight decrease in aboveground biomass for a few months; and
3) At spills where at least the upper one-third of the aboveground vegetation remains unoiled, the plants tend to have high survival rates.

Thus, there is a general dose-response relationship in terms of the degree of oiling of the vegetation, with emphasis on the leaves versus the stems. The leaves are responsible for respiration, transfer of oxygen to the roots, photosynthesis, and, in some cases, salt extrusion. Light oils exert a chemical toxicity, damaging the plant cells and their functions. Heavy oils are thought to exert a physical toxic effect through coating and smothering. Both mechanisms of toxicity are a function of the amount of oil coverage of the leaves.

Degree of Contamination of the Marsh Soils

One of the concerns about manual or mechanical treatment in oiled marshes is the risk of mixing oil into the marsh soils, which can increase the likelihood of further damage. Marsh plants have variable degrees of tolerance to oil in their soils. Greenhouse experiments allow for controlled comparisons of plant responses to various degrees of oiling. Figure 2-1 shows that there is a dose-response relationship for sprigs of *S. alterniflora* exposed to different amounts of No. 2 fuel oil mixed homogenously into marsh soils in pots for three months. Starting around 29 milligrams/gram (mg/g; 29,000 parts per million [ppm]), oil exposure starts to have detrimental effects on belowground biomass; aboveground biomass effects start at exposure to 57 mg/g. Lin and Mendelssohn (2008) also exposed *S. alterniflora* to weathered South Louisiana crude at six doses for 12 months, with various measures of plant health significantly lower at 320 mg/g and 640 mg/g. No plants survived exposure to 800 mg/g. These studies also support the conclusion that No. 2 fuel oil is more toxic to *S. alterniflora* than crude oil. Lin and Mendelssohn (2009) did similar studies with *Juncus roemerianus* exposed to weathered diesel for twelve months, with detrimental impacts to biomass occurring at 80 mg/g.

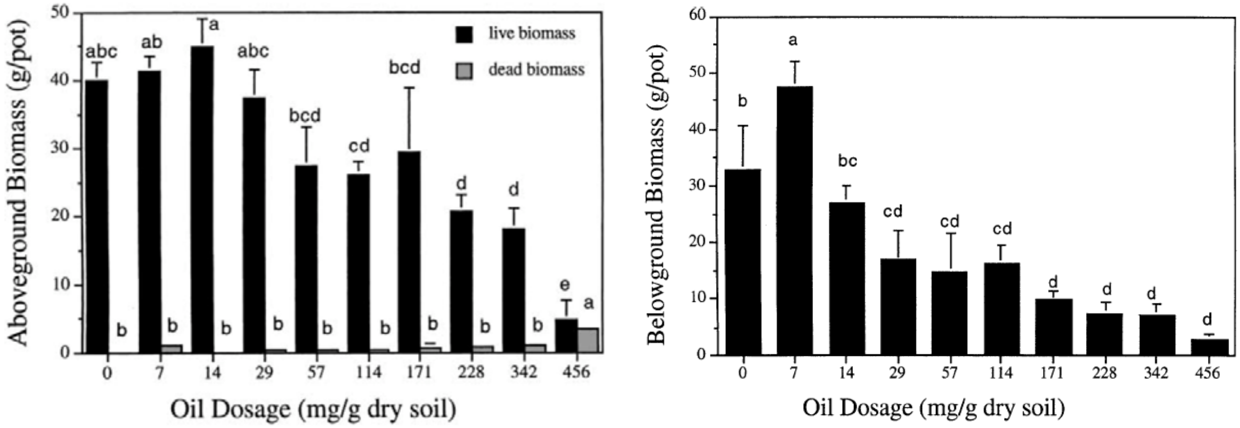

Figure 2-1. Effect of No. 2 fuel oil on the aboveground (left) and belowground (right) biomass of *S. alterniflora* three months after transplantation into soils mixed with different levels of oil. Values are means with standard errors (n=3). Means with the same letter are not significantly different. There is clearly a dose-response relationship (Lin et al. 2002b).

These thresholds of oil contamination from greenhouse experiments are higher than what is normally found in the field. Levels of No. 2 fuel oil in marsh soils after the *Florida* spill in Buzzards Bay, which caused such extensive plant mortality, were 0.45-0.59 mg/g right after the spill and 0.76-1.80 mg/g three months later (Sanders et al. 1980). At the *Bouchard 65* spill of No. 2 fuel oil in 1974 in Buzzards Bay, which also caused extensive marsh mortality and significant erosion, soil concentrations measured right after the spill were 11.4 and 20.6 mg/g in the top 6 centimeters (cm) (Teal et al. 1978). At the Exxon Bayway spill of No. 2 fuel oil in the Arthur Kill, New York, initial oil concentrations in the soils where marshes were killed were 6.4 mg/g right after the spill, 15-66 mg/g three years later, and 2.4-22 mg/g five years later in areas still denuded of vegetation (Bergen et al. 2000).

For crude and heavy refined products, the results are more variable. When planting marsh sprigs in an oil-impacted marsh, No. 6 fuel oil in soils at concentrations less than 2 mg/g had no effect on *S. alterniflora*, 2-10 mg/g had increasing effects, and greater than 10 mg/g resulted in plant mortality (Krebs and Tanner 1981). A light crude oil in the soil greater than 10.5 mg/g reduced live stem density of *S. alterniflora* and led to long-term impacts (Alexander and Webb 1987). The application of up to 8 L/m^2 of S. Louisiana crude oil to field plots enclosed by metal cylinders did not adversely affect *S. alterniflora* after three months, though the TPH levels in the soils at the end of the study were 40 mg/g (DeLaune et al. 1979).

Four spills stand out in terms of the persistence of a thick layer of oil on the marsh surface that affected recovery of the vegetation: a small spill in 1969 in Wales where a 5-cm thick oil layer on the marsh surface was not removed and the vegetation took 15 years to recover (Baker et al. 1993); the 1974 T/V *Metula* where 5-10 cm of thick emulsified oil covered the marsh surface and recovery was estimated to take decades (Figure 2-2); the 1991 Gulf War oil spill in the Arabian Gulf where thick and deeply penetrated oil resulted in extensive mortality (Barth 2002; Research Planning Inc. 2003; Höpner and Al-Shaikh 2008); and the 2010 *Deepwater Horizon* where thick mousse several centimeters thick was under a layer of thick oiled vegetative mat (see case study in Chapter 4). In fact, it was the lessons learned from the three earlier spills that led to the decision to use intensive treatment methods for the marshes with thick oil residues from the *Deepwater Horizon* spill.

The differences between greenhouse experiments and spills might be related to how the oil penetrates the marsh soils during a spill. Spilled oil is not uniform in its distribution with depth; it often penetrates into root cavities and burrows, forming pockets of very high oil loading and areas of clean sediment, particularly for viscous oils. Depending on the soil type, oil properties, and oil behavior over time, plant tissues will be exposed to widely varying oil concentrations for similar oil loading on the surface. Collecting a representative sample of such variable oil exposures is difficult, thus the range in measurements of how much oil causes different effects.

Exposure to Currents and Waves

The degree of exposure of a shoreline to mechanical energy generated by waves and currents is a core concept in shoreline sensitivity and the persistence of stranded oil, as evidenced in the Environmental Sensitivity Index shoreline classification scale (NOAA 2010). The residence time of oil on a shoreline increases as the energy of waves and currents decrease. Though marshes occur in low energy environments, there are still relative differences among the physical settings that are important to consider in determining the rate of natural removal by physical processes. For example, the T/V *Metula* in the Strait of Magellan heavily oiled 5-10 ha of tidal salt marsh, with spring high tides stranding thick layers of oil on the high marsh surface. In this cold, arid climate, there are no physical processes to assist in oil removal, thus the oil is predicted to persist for decades (Figure 2-2). In contrast, the heavily oiled marsh along the Delaware River from the T/V *Grand Eagle*, exposed to strong riverine and tidal currents, and boat wakes, recovered within two years (Figure 2-3 A and B).

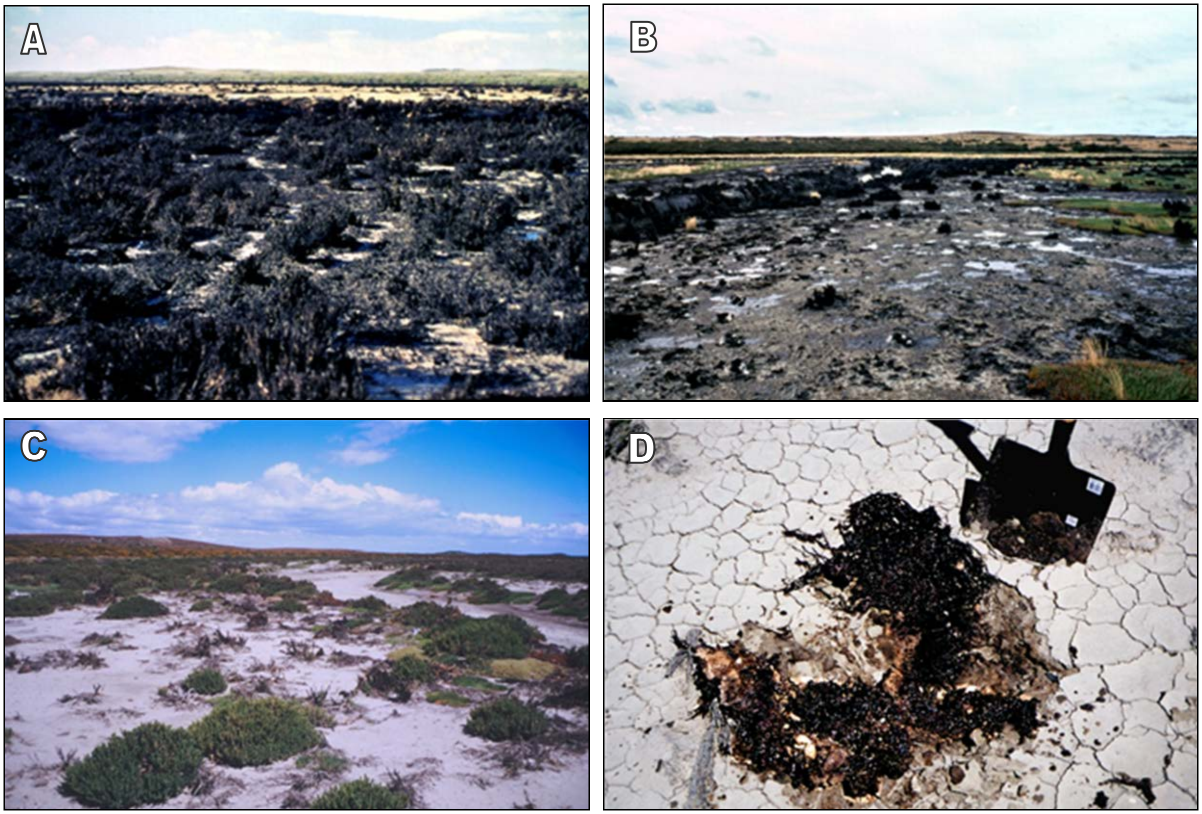

Figure 2-2. Examples of long-term persistent oiling in highly sheltered marshes. Punta Espora, Chile marsh that was heavily oiled as a result of the T/V *Metula* spill in 1974. A: Oiled marsh in January 1976. B: Same area in January 1981. The oil stranded on the high marsh platform where it is isolated from physical removal processes. The oil is expected to persist for many decades. C and D: In 1995, the marsh surface has been covered by a thin layer of silt; however, the thick layer of oil has peristed for 21 years. Photo credit: Erich Gundlach.

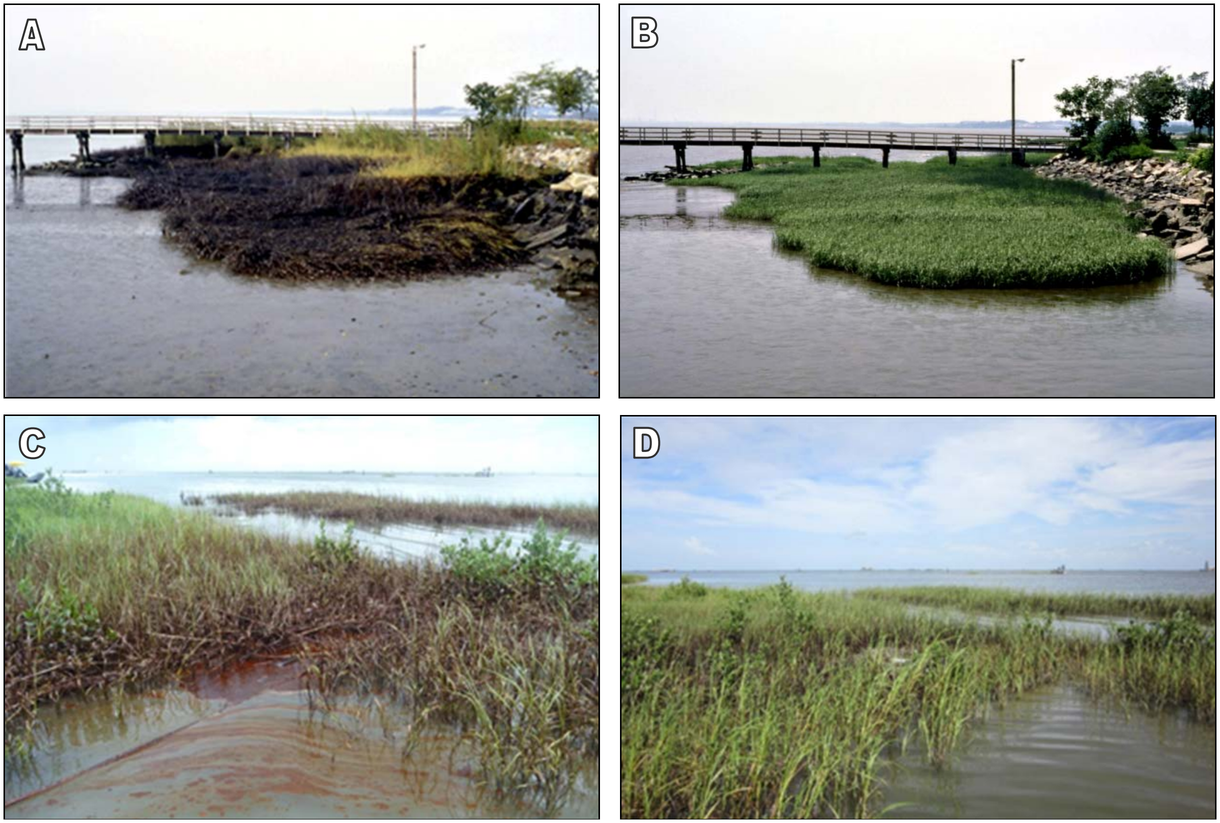

Figure 2-3. Examples of the role of natural removal processes. Relatively exposed marshes. Top Row: *Grand Eagle* spill in the Delaware River. A: 1984; B: 1986. Strong river currents and boat wakes were very effective at natural oil removal. Photo credit: Tom Ballou. Bottom Row: *Deepwater Horizon* oil spill. C: Moderately oiled Louisana salt marsh on 3 July 2010; D: Same area on 27 July 2010. Photo credit: Missy Kroninger.

There are many examples of the importance of waves and currents in speeding natural removal of oil on marshes. At the 2010 *Deepwater Horizon* oil spill, 796 km of marsh shoreline in Louisiana were oiled; however, shoreline treatment was approved for only 71 km, or 8.9% of oiled marshes and associated habitats (with the actual distance treated being much lower than this) (Michel et al. 2013). One year later, there were 200 km of oiled marsh remaining. The bottom row of photographs in Figure 2-3 shows one area in Louisiana where the oil was removed by wave action over a period of a few weeks.

Time of Year of the Spill

Observations during experimental and actual spills have shown that the time of year of oiling of marsh vegetation is an important factor in the potential for impacts and the rate of recovery. In fact, Baker (1971) was the first to report that oiling outside of the growing season was less damaging. Several researchers have suggested why seasonality is so important (Mendelssohn et al. 1995; Webb 1996; Pezeshki et al. 2000). When plants are growing, they are physiologically very active, thus if oiling interrupts these physiological functions, plant health can be affected. Damage to leaf stomata, either by coating by heavier oils or tissue damage by lighter oils, can reduce transpiration, which can lead to overheating and death of the aboveground vegetation. Oil coating can also reduce oxygen transport to the roots, which can kill the belowground vegetation. Oil can reduce photosynthetic rates, which can slow growth and affect plant survival.

In contrast, it is clear that marshes that are oiled at the start of or during dormancy, when the aboveground vegetation has naturally died back, have a much greater potential for recovery. It makes sense that oiling of senescent vegetation would have less physiological stress on the plant. Figure 2-4 shows a *S. alterniflora* marsh that was heavily oiled in late September 1996 during the T/V *Julie N* spill of an IFO 380, compared with the next summer. The vegetation fully recovered in one growing season, in spite of the very heavy oiling of the vegetation, with only passive recovery of oil using sorbents.

Species Sensitivity

Marsh plants vary in their sensitivity by species and even by ecotypes within species (Lin and Mendelssohn 1996; DeLaune et al. 2003). When exposed under similar greenhouse experiments, the following species can be ranked from least to most sensitive:

Least Sensitive →			***→ Most Sensitive***
Sagittaria lancifolia (bulltongue arrowhead) *Phragmites australis* (roseau cane/common reed) *Typha latifolia* (broadleaf cattail)	*Spartina alterniflora* (smooth cordgrass)	*Juncus roemerianus* (needlegrass rush)	*Spartina patens* (saltmeadow cordgrass)

Figure 2-4. Heavily oiled *S. patens* marsh during the T/V *Julie N* spill of an IFO 380 in Portland, Maine in October 1996 (A) and July 1997 (B), showing the importance of season in how plants respond to oil exposure. Oiling in fall, when the plants are in senescence, has the lowest potential for impacting the vegetation. Photo credit: Jacqueline Michel.

Sensitivity among species may be controlled by the depth and size of the rhizomes, with deeper rhizomes less likely to be exposed to oil on the surface and larger rhizomes having more food storage and ability to survive short-term effects on photosynthesis and other metabolic processes. It may also be a function of the properties of the soils the plants grow in. Oil tends to accumulate and persist in soils with high organic matter content, depending on the water levels when oil is present (that is, the oil has to come in contact with the soil surface). The size and number of stems may also be a factor, with smaller, more numerous stems per plant having the potential for a higher surface area of oiling. For example, *S. patens* can have ten times the number of stems per meter than *S. alterniflora*, which would provide a very large surface area for oil adherence.

It has generally been found that annuals are more sensitive than perennials. Annuals have to grow every year from seed, so they would be more susceptible than plants that regrow from an existing root network. However, if there is a nearby source of seeds, often the annuals are the first to recolonize a heavily oiled marsh. As the surface oil weathers, new seeds can germinate in the cracks in the oil layer. Once some vegetation takes root, it speeds the overall rate of recovery (see the case study of the *Amoco Cadiz* spill in Chapter 4). In contrast, perennial plants usually recover from the spread of roots from live plants around the impacted site, which can be relatively slow.

One result of the different sensitivities of plant species is that oiling can cause a temporary change in the composition of a marsh because of the dieback of the more sensitive species. However, eventually the normal species distribution returns, as long as other factors are not changed (such as a change in the elevation of the marsh). This effect has been seen at spills and in greenhouse experiments, mostly in brackish and freshwater marshes because they can have a more diverse mix of species present. Salt marshes are usually dominated by one species, or a distinct zonation of species, that can best compete given the salinity regime and tidal elevation.

Impacts of Oil on Marsh Fauna

There are few studies of the impacts of oil on the fauna associated with marshes. Many of the available studies focus on epifauna, such as intertidal crabs, periwinkle snails, and mussels. High rates of mortality for fiddler crabs have been documented after spills of light refined oils. At the *Florida* spill in Buzzards Bay, Krebs and Burns (1978) documented that it took more than seven years for fiddler crabs to recover because of the persistence of the toxic naphthalene aromatic compounds in the soils in which the crabs burrow and the juveniles recruit. High fiddler crab mortalities were also reported for the Exxon Bayway spill of No. 2 fuel oil in Arthur Kill (Burger 1994), a crude oil spill in Nigeria (Snowden and Ekweozor 1987), and a No. 6 fuel oil spill in New Jersey (Dibner 1978). Oil can affect crabs in several ways: 1) acute and chronic mortality from the toxic components of the oil; 2) physical smothering by heavier oils; and 3) creation of physical barriers to access to the marsh surface and subsurface sediments such as thick oil layers, viscous oils, and algal mats. Massive mortality of intertidal crabs occurred as a result of the largest marine oil spill in history, the Gulf War spill in the Arabian Gulf, and the crabs have been a key part of the overall recovery of intertidal communities because of their prodigious burrowing which speeds oil degradation (Barth 2007). In fact, the large restoration projects along the Saudi Arabian coast are focusing on removal of the physical barriers to crab recruitment (Hale et al. 2011).

Periwinkle snails are also very susceptible to oiling impacts because they are closely associated with the emergent vegetation in the marsh, typically *S. alterniflora*. While vertical movement up and down cordgrass stems for feeding, predator avoidance, and regulation of temperature and oxygen availability is frequent, marsh periwinkles rarely move laterally more than a few meters (Vaughn and Fisher 1992). Both oil spill and experimental spill studies have observed high mortality of periwinkles immediately after a spill, followed by gradual increase in numbers over months or years as the vegetation recovers (Hershener and Moore 1977; Hershener and Lake 1980; Lee et al. 1981; Conan et al. 1982; Clarke and Ward 1994; Pearce 1996; Zengel and Michel 2012; Zengel et al. 2013).

Ribbed mussels are important to the survival of *S. alterniflora*, particularly along waterways with heavy boat traffic and wakes, by binding the root mat together, effectively stabilizing the substrate and strengthening the plant and the entire marsh against physical disturbance and erosion (Bertness 1984). Ribbed mussels are also important filter feeders, playing a key role in the food web and in the cycling of carbon, nutrients, and minerals through the salt marsh ecosystem. Several of the spills listed in Appendices A-C include cases where high mortality of ribbed mussels was noted, particularly for light refined oils. They are also susceptible to smothering from oil or inability to recruit due to chronic toxicity.

Studies of resident Gulf killifish (*Fundulus grandis*) in marsh habitats and in laboratory studies with oiled sediments affected by the *Deepwater Horizon* oil spill (Whitehead et al. 2011; Dubansky et al. 2013) showed a wide range of sublethal responses, including development abnormalities in gills, liver, head kidney, and intestine of adult and larval fish, cardiovascular defects in embryonic fish, delayed hatching, overall reduced hatching success, smaller size at hatching, and edemas. These fish have small home ranges and high site fidelity, making them particularly sensitive to population-level impacts from persistent oil exposures.

Summary and Response Implications

The body of literature on oil toxicity and impacts to marshes is extensive and provides a range of results from which we can extract guidance to assist planning for or responding to oil spills.

- When a spill threatens a shoreline, marshes are likely to become oiled because they occur in the upper intertidal zone where the oil usually strands. The degree of impact is very closely correlated with the degree of oiling. Therefore, response actions that minimize the amount of oil that can reach the shoreline will reduce the degree of impact to these sensitive and productive habitats.
- Spills of light refined oils can result in high mortality of marsh vegetation and biota, but only where large amounts of oil strand on the shoreline, such as large spills in inland waterbodies, small spills in small waterbodies, or spills directly into marshes. In most offshore spills, the oil spreads, disperses, and evaporates to the point that the amount of oil that reaches the marsh is not enough to cause large-scale effects.
- Crude oils and heavy refined oils that coat the entire plant, and particularly the leaves, will have the greatest potential impacts. Oiling of only the stems often results in limited mortality. If only the aboveground vegetation is oiled, regrowth is likely during the next growing season, particularly for oiling of the marsh fringe where natural removal processes are relatively fast.

- Spills in the marsh interior are likely to result in thicker oil residues, higher impacts (partially because of the lack of weathering before contact with the marsh), and slower natural removal rates. Thus, these kinds of spills often require intensive removal actions.
- Impacts are more persistent when oil penetrates into the marsh soils. Persistence increases with deeper penetration, soils high in organic matter, and sites that are sheltered from natural removal processes.
- Vegetation recovery will occur quicker for spills of any type of oil during the non-growing season, compared to a spill during the growing season.
- Although there are some indications of different sensitivities among species, the specific spill conditions are the most important factors in determining impacts.
- Annuals are more likely to be affected compared to perennials; however, they often are the first to recruit to oiled sites.
- Thick oil layers on the marsh surface are known to cause long-term impacts to both vegetation and fauna; therefore, early removal actions can speed recovery, as long as they are well planned and are conducted with careful oversight.

There have been several summaries of the recovery rates for oiled marshes. Sell et al. (1995) compared the recovery rates of heavily oiled salt marshes for seventeen spills and field experiments, showing that sometimes treatment resulted in more rapid recovery, and sometimes treatment slowed recovery. Hoff (1995), in her paper on "The Fine Line between Help and Hindrance" summarized recovery rates for seventeen spills and field experiments (there were seven cases common to both summaries) made similar observations.

Figure 2-5 shows a plot of the estimated "years to recovery" for 33 spills and field experiments for lightly to heavily oiled marshes. Note that for the Gulf War oil spill, those marshes that showed little or no recovery as of 2009 were treated during an extensive restoration project being conducted from 2011 to 2014, thereby shortening what would have been even longer recovery periods for the upper marshes, which are composed of long-lived, slow-growing woody species.

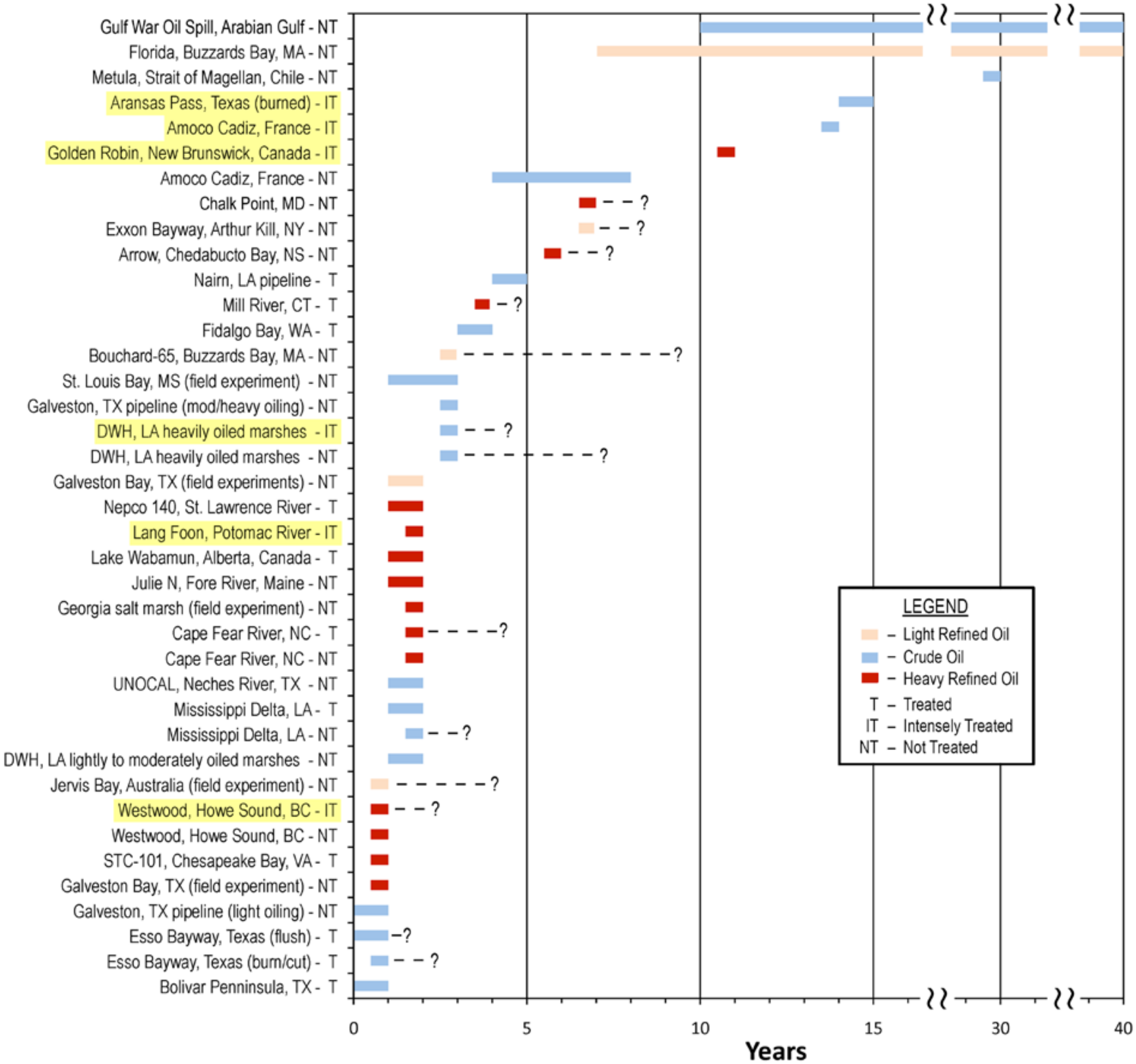

Figure 2-5. Years to recovery for spills and a few field experiments color-coded by oil group, from shortest to longest recovery. Yellow highlighting is used to identify those spills where intensive treatment was conducted. Dashes and question marks are used to represent potential time to recovery based on results of the most recent data.

The interpretations are similar to Sell et al. and Hoff, in that:

- Recovery is longest for spills with the following conditions:
    - Cold climate (e.g., *Metula*, *Arrow*, *Amoco Cadiz*)
    - Sheltered settings (e.g., *Metula*, *Arrow*, Gulf War, Nairn pipeline, Mill River)
    - Thick oil on the marsh surface (e.g., *Metula*, *Amoco Cadiz*, Gulf War)
    - Light refined products with heavy loading (e.g., *Florida*, *Bouchard-65*, Exxon Bayway)
    - Heavy fuel oils that formed persistent thick residues (*Arrow*)
    - Intensive treatment (e.g., Aransas Pass, *Amoco Cadiz*, *Golden Robin*)
- Recovery is shortest for spills with the following conditions:
    - Warm climate (e.g., many spills in Louisiana and Texas)
    - Light to heavy oiling of the vegetation only
    - Medium crude oils
    - Less-intensive treatment

It is interesting to note in Figure 2-5 that for most spills, recovery occurred within 1-2 growing seasons, even in the absence of any treatment. The decision to conduct treatment operations in oiled marshes needs to be based on the best understanding of the likely tradeoffs. Every spill is a unique combination of conditions that have to be evaluated to determine if and how much of the oil has to be removed, and the most effective removal methods. In Chapter 3, we discuss guidelines on appropriate removal methods.

For Further Reading

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CHAPTER 3. RESPONSE

Key Points

- Marshes are highly sensitive to oil and often are priority areas for protection.
- Winds and currents can carry spilled oil into marshes where the oil coats the soil surface, vegetation, and animals in the marsh.
- Dispersing or burning offshore can prevent or lessen impacts to salt marshes, though these response options are not often considered for use in freshwater environments where drinking water intakes are at risk. Also, dilution rates are slower, thus there would be concerns about impacts to aquatic resources such as fish.
- Spill containment and cleanup techniques need to be carefully evaluated for the specific spill conditions, to minimize any additional impacts to marsh environments and associated fauna and speed overall recovery post spill.
- Often, multiple response options should be used in combination or succession.
- At some point in time, all treatment methods will become less effective and can potentially cause additional damage.

As detailed in the previous chapter, marshes are particularly sensitive to oil and should be priority areas for protection. However, it is difficult to protect extensive marshes even under ideal conditions, and the rapid transport of oil onshore often results in oiling of these sensitive habitats. Any oil removed during on-water response will reduce the amount of oil potentially reaching the shoreline. On-water response options to minimize oiling of wetlands discussed here include mechanical containment and recovery, offshore dispersant application, and offshore *in situ* burning.

Once oil reaches a marsh, the impact of oiling varies by oil type, degree of oiling, wetland type, weather, water levels, degree of exposure to waves and currents, and time of year. Cleanup options should be evaluated to determine whether the ultimate benefits from the response action outweigh any additional impacts occurring during their implementation. This chapter summarizes what is known about the environmental tradeoffs with different treatment options.

On-Water Response Options to Prevent Marsh Oiling

Mechanical Recovery

Mechanical containment and collection of spilled oil on water using equipment such as booms and skimmers are primary initial cleanup methods used at many spills. Experience has shown, though, that mechanical recovery alone usually cannot adequately deal with offshore spills. Weather and sea conditions, the nature of the oil, and other factors may limit the effectiveness of mechanical recovery. Experience has shown that mechanical recovery rates greater than 20% are rare. In such cases, alternative open-water response techniques, such as dispersant application or *in situ* burning of oil on water, may significantly reduce the risk that oil will reach shore and impact marshes and other sensitive intertidal and nearshore habitats.

Offshore Dispersant Application

Chemical dispersants are products applied to oil on the water surface to enhance formation of smaller oil droplets that are more readily mixed into the water column and dispersed by turbulence and currents. During and since the *Deepwater Horizon* oil spill, dispersants have also been considered as a response action to reduce the amount of oil reaching the surface during a subsea release. Most oils physically disperse to some degree due to agitation created by wave action and ocean turbulence. Chemical dispersants enhance and speed up this natural dispersion process. Dispersing oil soon after release minimizes impacts to wildlife at the water surface (e.g., birds and marine mammals) and reduces the amount of floating oil that may reach sensitive nearshore and shoreline habitats. If applied appropriately offshore, chemical dispersants can be an effective tool for protecting marshes and the habitat they provide. Tradeoffs among other resources at risk, such as potential effects of temporarily higher concentrations of oil in the water column on pelagic organisms and sedimentation of oil in sensitive benthic habitats such as seagrasses and shellfish beds, should be considered before dispersant use. In freshwater environments, there are additional concerns about mixing oil into the water column that would increase the risk of contamination of water intakes and the slower mixing and dilution rates in lakes, thus increasing concerns about impacts to aquatic resources. Furthermore, most current dispersant formulations are not all that effective in freshwater. Therefore, use of chemical dispersants is less likely to be considered during spills in freshwater environments.

There have been few studies to mimic the effects on marshes from oil that is dispersed nearshore. Smith et al. (1984) conducted a field experiment of the effect of dispersed and undispersed South Louisiana crude oil on the growth of *S. alterniflora* and meiofauna in a uniform Louisiana salt marsh. The oil and the oil plus dispersant were applied to open water adjacent to the marsh and forced onto the marsh using a pump to create a “head” of water that simulated tidal conditions. Neither crude oil nor oil plus dispersant had any inhibitory or stimulatory effect on the growth of *S. alterniflora* or the meiofaunal communities, including the meiobenthos. Laboratory studies showed that both fresh and salt marsh vegetation is not sensitive to chemical dispersants (JD 2000 and Corexit 9500) at even high concentrations of exposures (>8,000 ppm) in the water column (Lin and Mendelssohn 2003, 2004). These studies also showed that the toxicity of both diesel and crude oil was reduced when simulating exposure of dispersed oil to *S. alterniflora* vegetation. Thus, under realistic exposure pathways (dispersed oil entering a marsh with the tides), it appears that marsh vegetation is not particularly sensitive, although the marsh fauna may be sensitive, depending on the dispersed oil concentrations.

Offshore *In Situ* Burning

in situ burning is a response technique in which spilled oil is burned in-place. When used appropriately, *in situ* burning offshore can remove large quantities of oil quickly and efficiently with minimal logistical support. Like dispersants, *in situ* burning of offshore spills can help minimize impacts to wildlife at the water surface and reduce the amount of oil that reaches sensitive nearshore and shoreline habitats. A potential disadvantage of open-water *in situ* burning is that a small percentage of the original oil volume may remain as a taffy-like residue after the burn. Floating residue can be collected, but residues that sink or escape collection and move inshore could potentially contaminate nearshore benthic habitats. Burning also can affect air quality.

Response Options for Oiled Marshes

When marshes are oiled, selection of the best response option(s) is very important. Table 3-1 is an updated version of the matrix for salt to brackish marshes from the NOAA (2010) Characteristic Coastal Habitats: Choosing Spill Response Alternatives. It ranks response options for shoreline cleanup in marshes for different oil types considering both the impact of the cleanup method and its effectiveness at oil removal.

In this section, the effectiveness and likely impacts of these response options are discussed. It is important to note that multiple response options may be used in combination or succession, depending on the oiling conditions.

Table 3-1. Recommendations for response options in oiled marshes by oil group (modified from NOAA 2010).

Oil Group Descriptions

I – Gasoline products
II – Diesel-like products and light crudes
III – Medium grade crudes and intermediate products
IV – Heavy crudes and residual products

The following categories are used to compare the relative environmental impact of each response method in the specific environment and habitat for each oil type. The codes in each table mean:

A = The least adverse habitat impact.
B = Some adverse habitat impact.
C = Significant adverse habitat impact.
D = The most adverse habitat impact.
I = Insufficient information – impact or effectiveness of the method could not be evaluated.
– = Not applicable.

Response Method	Oil Group I	II	III	IV
Natural Recovery	A	A	B	B
Barriers/Berms	B	B	B	B
Manual Oil Removal/Cleaning	D	C	B	B
Mechanical Oil Removal	D	D	C	C
Sorbents	–	A	A	B
Vacuum	–	B	B	B
Debris Removal	–	B	B	B
Sediment Reworking/Tilling	D	D	D	D
Vegetation Cutting/Removal	D	D	C	C
Flooding (deluge)	B	B	B	B
Low-pressure, Ambient-water Flushing	B	B	B	B
Shoreline Cleaning Agents	–	–	B	B
Nutrient Enrichment	–	B	B	C
Natural Microbe Seeding	–	I	I	I
In Situ Burning	–	B	B	B

Natural Recovery

There are many spills in marshes where the decision is made to allow natural recovery to proceed without any active cleanup, because active cleanup would cause more harm than benefit to the habitat and the animals using that habitat. Nearly all types of active cleanup will include some habitat damage or disturbance whether it is from the type of equipment used, the way it is used, or the mere presence of the cleanup workers disturbing wildlife or trampling the marsh. Typically, natural recovery is selected when:

- The spill is of a light oil that is expected to naturally evaporate and break down rapidly. The toxic effects of light refined products such as diesel and jet fuels occur quickly, and attempts to remove the oil could cause more damage.
- The impact area is small.
- The oil is mostly on the vegetation. As discussed in the section on oil impacts, it has been well documented that oil on vegetation will often weather to a non-sticky coating within weeks, and plants often survive even heavy coating.

- The vegetation is in its dormant season. The aboveground vegetation for many species naturally dies back in the fall/winter and new vegetation emerges in spring. Therefore, the oiled vegetation will be replaced, and the oil is removed from the marsh by this process as well.
- The oiled marsh is exposed to waves and/or currents that speed the rate of oil weathering and removal.
- Key animals are not at risk, such as threatened or endangered species.
- Active cleanup methods are determined to be causing too much damage or are no longer effective and thus are terminated.

This last point is important; responders should continually reevaluate the shoreline response to make sure that approved methods are being properly implemented and are still effective and needed. Oils change properties as they weather, and methods that were initially very effective can become less effective over time.

Figure 3-1 shows time-series photographs of a spill where natural recovery was found to be very effective. When natural recovery is the preferred response option, it is still important to take action to contain any oil that is released from the marsh and prevent oiling of adjacent areas. Possible response options are discussed below in the order listed in Table 3-1.

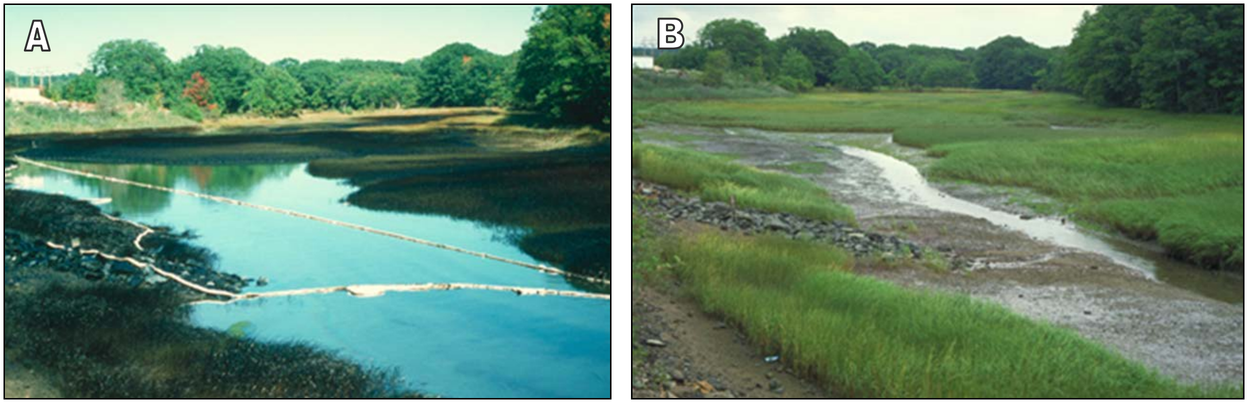

Figure 3-1. T/V *Julie N* spill of IFO 380 into the Fore River, Portland Maine where natural recovery was very effective. A: September 1996. B: July 1997, one year later. Photo credit: Jacqueline Michel.

Barrier Methods

Barriers such as boom or filter fences can be used in an attempt to keep oil from stranding in the marsh. Booms float on water, so they need to be anchored or staked so that they do not foul on the intertidal zone during low tide or on the vegetation at high tide. This often happens anyway, so even booming can cause damage; it certainly causes disturbance because of the constant need for maintenance and replacement. Booms are particularly difficult to keep in place along shorelines exposed to waves and currents, and they should be removed when a large storm is predicted to affect the area. During the *Deepwater Horizon* spill, hundreds of miles of hard boom and sorbents were stranded along hundreds of miles of shoreline by large waves from an offshore storm. It took months of work by many special boom-removal teams to retrieve the stranded boom (Figure 3-2), and there was a massive effort to locate and remove orphan anchors. SCAT teams still found boom stranded in the marsh in early 2013, nearly three years after the spill. Therefore, responders need to carefully evaluate the effectiveness of placement of boom along extensive areas of marsh shoreline, particularly where exposed to waves. Improper booming can cause significant damage.

Filter fences have been placed along the marsh edge, with variable success. Numerous stakes are necessary to keep them in place, and they often fail under wave action (Figure 3-3). Furthermore, they are very difficult to remove because the stakes get buried in mud, the cloth can get weighted down with mud, and debris tends to accumulate around them. Complete removal is important because the stakes can pose hazards to people and boats, particularly if the shoreline is eroding. Recording accurate GPS coordinates when such barriers are installed will aid in their location during removal actions. Based on experience during the *Deepwater Horizon* well, such protection measures are not likely to be effective and pose significant difficulties during removal.

Figure 3-2. Top row: boom stranded on salt marshes (left) and *Phragmites* marsh in Louisiana in July 2010 after the passage of two storms that generated waves and high water. Photo credit: Andy Graham. Bottom row: specialized boom removal teams removed the stranded boom using various techniques to minimize further damage to the marshes. Photo credit: *Deepwater Horizon* Response.


Figure 3-3. Shoreline barriers used during the *Deepwater Horizon* oil spill. Filter fences require many stakes. Usually there is not enough time to deploy this type of barrier after a spill, they have limited effectiveness, and they are difficult to remove. The hard boom has become stranded on the marsh. Photo credit: Helen Chapman (left); Thomas Minter (right).

Manual and Debris Removal

Manual removal involves the use of hand tools and manual labor to remove thick accumulations of viscous oil and oiled debris from the marsh surface. Depending on location, vehicles such as marsh buggies and all-terrain vehicles may be used to haul workers and wastes. All work in soft sediments and in vegetated areas needs to be conducted using walking boards (planks of wood) to prevent damage. Trampling is very hard to avoid and often causes long-lasting damage, mostly by driving the oil deep into the soils, and also by physically damaging the vegetation. There have been many spill responses in marshes where years later the main evidence of the spill is from the physical damage caused by foot traffic and vehicles used to transport workers and wastes. After a spill of Bunker C in a *Carex* marsh in British Columbia, where local stakeholders pushed for aggressive removal of the oil, Challenger et al. (2008) documented nearly complete vegetation mortality and increased and prolonged oil contamination of soils. However, with small teams, close supervision, and a clear understanding of the removal methods and adaptation over time, manual removal can be effective. During the *Deepwater Horizon* spill, most of the marsh cleanup was conducted manually by teams that removed very heavily oiled wrack and thick oil layers along 11 km of fringing marsh in Louisiana, with mainly positive results (see *Deepwater Horizon* case history).

Mechanical Removal

Mechanical removal is seldom used because of the potential for extensive damage to the marsh soils. It is usually considered only under very heavy oiling conditions when rapid removal is of priority or where soft substrates limit manual removal. Two recent examples are the 2000 Chalk Point spill in Maryland and the 2010 *Deepwater Horizon* spill in Louisiana. The Chalk Point spill released 126,000 gallons of a mixture of No. 6 and No. 2 fuel oils from a pipeline break in the interior of a brackish marsh. A network of trenches was dug to improve low-pressure flushing efforts (Figure 3-4). The trenches were backfilled with clean material and bare areas successfully re-planted (Gundlach et al. 2003). Mechanical methods used during the *Deepwater Horizon* response included barge- and airboat-based platforms with long-reach hydraulic arms coupled with attachments for rakes, grapples, vegetation cutting devices, and "squeegees" that involved only one spotter on the marsh to direct the operator on the boat. Even with close supervision, mechanical methods had a greater chance of causing impacts compared to manual crews. For more discussion of impacts associated with mechanical removal, see the *Deepwater Horizon* case history in Chapter 4.

Figure 3-4. The extensive network of trenches dug during the Chalk Point oil spill in April 2000 to increase effectiveness of flushing of the mixture of No. 6 and No. 2 fuel oil that was released inside the marsh from a pipeline break. Extensive replanting was conducted and was very successful (Gundlach et al. 2003). Photo credit: Jacqueline Michel.

Sorbents

Even when natural recovery is the selected option, sorbents are often deployed to recover any oil released from the area. Sorbents are composed of materials that either adsorb oil on the surface or absorb oil into the pores of the material. There are many types: natural organic substance (e.g., peat, wood, cotton, straw, shredded sugarcane process residual called bagasse), synthetic organic substance (e.g., polypropylene, polyurethane), inorganic mineral substance (e.g., clay, vermiculite, diatomite), or a mixture of the three. The material may also be treated with oleophilic (oil-loving) and hydrophobic (water-hating) compounds to improve performance. They come in various forms: round sausage “boom,” snare, sweeps, pads, rolls, loose particulates, pillows, and socks. In marshes, sorbents are often used in the following manner:

1) On water, sorbent “boom” is deployed to passively recover oil being mobilized by waves and currents from the marsh. Care is needed during placement and removal to minimize the damages and disturbances previously described for booms. Sorbents can generate excessive wastes so they should be removed when sheening reaches minimal amounts.
2) On the marsh surface, sorbent pads and snares can be used to pick up liquid or sticky oil. Figure 3-5 shows workers on walking boards (which can be planks of wood nailed together or sheets of plywood) using snares to recover thick oil from deep inside a marsh where there was no access for vacuum systems.
3) On the marsh surface and vegetation, loose organic sorbents can be spread on the surface and lightly raked into areas of liquid or sticky oil (making sure not to disturb the vegetation or marsh sediments) then removed for proper disposal (Figure 3-6). This application method requires cleanup crews to walk on the marsh surface, so walking boards are required.
4) On the marsh surface and vegetation, loose organic sorbents can be applied by hand or a small sprayer to provide a barrier to reduce the risk of oil exposure by wildlife in the marsh. For fringe oiling, the sorbents can be applied from shallow-draft boats, otherwise, walking boards will be required for foot traffic on the marsh surface.

Usually approval from the Regional Response Team is required for application of loose organic sorbents without removal.

Figure 3-5. Workers using snares on poles to remove thick oil floating on the water surface deep in the brackish marsh interior at the Chalk Point oil spill on the Patuxent River, Maryland in April 2000. Note the use of walking boards. Photo credit: Jacqueline Michel.

Figure 3-6. Use of loose organic sorbents during the *Deepwater Horizon* spill in Louisiana on 9 July 2011. A: Crews used potato rakes (lower left) to mix the sorbent into thick oil on the marsh surface then removed it. B: A final layer of sorbent was applied at the end of treatment, as a barrier to contact with wildlife. Photo Credit: Eric Schneider.

Vacuuming

Vacuuming can be used to remove pooled or thick oil accumulations on the marsh surface, in depressions, and floating in channels. Vacuum equipment ranges from small, portable units to large suction devices mounted on barges adjacent to the marsh edge. Vacuuming is most often appropriate to use early in the response for medium and heavy oils, when the oil is still liquid and floating on the water surface. Weathered or viscous oils have to be concentrated using booms and “fed” into the nozzle. Operationally, it is important to minimize vacuuming of water, because of limited storage capability and the water may have to be treated prior to discharge. The biggest limitations are usually logistical; that is, how to get the vacuum system to where the oil is in the marsh under variable tide and wave conditions and in shallow water. Land-based operations are limited by the distance over which the hoses can be laid out between the oil to be treated and the storage tank, though it can be hundreds of meters with use of booster pumps. Care will be required to minimize trampling of soils and vegetation during handling of hoses and actual vacuuming of the oil. Workers also need to be careful to not gouge the surface of the marsh, removing marsh soils and inadvertently changing the marsh elevation with potential subsequent adverse effects to marsh vegetative and fauna communities. Another issue is that the oil will continue to spread into thinner layers, reducing the effectiveness of vacuuming, thus rapid identification and removal of areas of pooled oil are essential.

Hoff et al. (1993) showed that careful use of vacuum and flushing by workers using walking boards removed the most oil and minimized damage to a *Salicornia virginica* marsh in Fidalgo Bay, Washington heavily oiled by a spill of Prudhoe Bay crude oil. By the second growing season, there was 100% plant cover in all but one small area.

Figure 3-7 shows the use of a small vacuum system to recover emulsified oil from a tidal channel during the 1997 Bayou Perot, Louisiana oil spill. Note the use of boom in a "tear-drop" configuration to concentrate the oil and minimize pickup of water. The oil was pumped into barrels on an airboat; when the barrels were full, another airboat brought an empty replacement and ferried the full barrel back to a barge in deeper water offshore.

Figure 3-7. Vacuuming of thick oil from the water surface in a marsh channel, Bayou Perot, Louisiana in February 1997. Photo credit: Jacqueline Michel.

During the *Deepwater Horizon* spill, crews used vessel-based vacuuming to remove the thick mousse adjacent to oiled vegetation in the most heavily oiled areas in Louisiana. Though this method removed a lot of mostly floating oil initially, when used later in the response on the marsh surface, the hard nozzle gouged the marsh surface, creating holes that allowed the mousse to slowly seep deeper into the sediments. Once it was determined to be no longer effective and was causing more harm than benefit, operations were terminated. This is an important point to be made: at some point in time, all treatment methods will become less effective and can potentially cause additional damage. Thus, it is important to monitor operations to make sure that each method is still effective.

Vegetation Cutting

Cutting of oiled vegetation is considered for several reasons:

- To reduce contact hazards with wildlife, particularly birds and small fur-bearing mammals associated with the marsh;
- To speed the recovery of the marsh;
- To gain access to oil trapped by vegetation on the marsh surface or in thick vegetation; and
- For aesthetic reasons in public areas of high visibility.

Cutting methods include weed trimmers, power hedge trimmers, and floating mechanical reed cutters.

Zengel and Michel (1996) reviewed 22 spills and experiments where cutting was used as a treatment method and generated a tabular summary of each study. Figure 3-8 shows time-series photography of some of these cases. Seven other studies have been identified since then:

- A field experiment in Brazil where both cut and uncut *S. alterniflora* marshes oiled with a medium fuel oil recovered within six months (Wolinski et al. 2011);
- A small-scale field test of cutting of *S. foliosa* oiled by an intermediate fuel oil in Humbolt Bay, California in October 1997 that was revisited one and two growing seasons later showing the cut areas were slightly impacted versus natural recovery (Lesh and Jocums 1999);
- Cut *Phragmites* (to gain access to the marsh interior) showed better recovery versus untreated areas in Louisiana in 1993 (Lin et al. 1999; and photographs in Figure 3-9);
- Cut bulrushes after the August 2005 spill of Bunker C into Lake Wabamun, Alberta, Canada recovered more slowly compared to uncut areas (Wernick et al. 2009);
- Various aggressive treatment of a *Carex* marsh following a August 2006 spill of Bunker C in British Columbia, Canada that showed cut-only areas were similar to untreated and control areas (Challenger et al. 2008);
- *Typha* that was cut during the June 1978 spill of Bunker C from the barge *Nepco-140* in the St. Lawrence River grew taller but didn't flower the first growing season, but was normal the second growing season (Alexander et al. 1981); and
- The operational raking and cutting of 11 km of heavily oiled salt marsh in Louisiana during the *Deepwater Horizon* oil spill, which will be discussed in more detail below.

The strongest justification for cutting is made for the protection of wildlife. However, there is usually no careful discussion as to whether a given oiled marsh poses a clear and present danger to wildlife, and for how long. Often oiled marshes are less of a threat by the time discussions of cutting take place; thus the perceived tradeoff of wildlife protection for marsh injury is unfairly weighted toward the former. Prior to the decision to cut oiled marsh vegetation, responders should involve experts in both marshes and the wildlife at risk to make a very balanced evaluation of the tradeoffs, including the exposure pathways from an oiled marsh to wildlife, the reduction of that exposure/risk over time, and methods of determining this risk in the field.

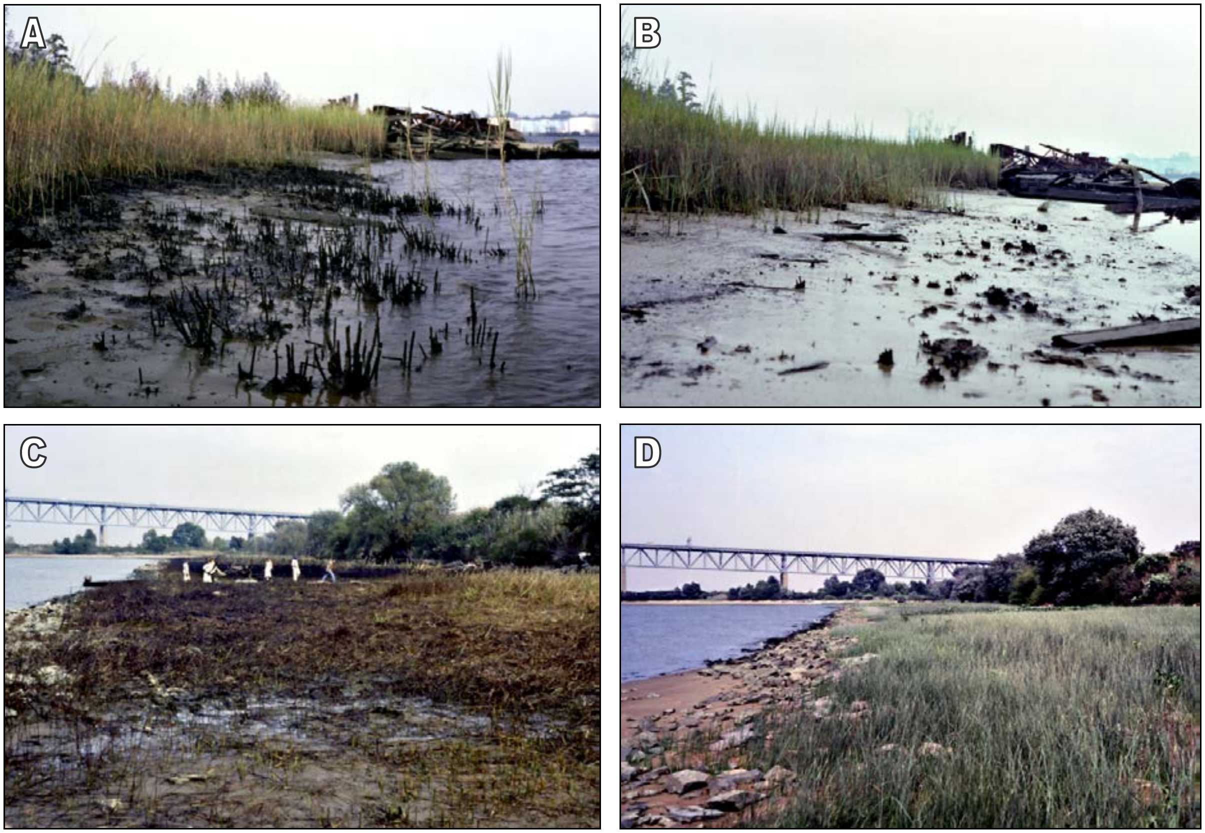

Figure 3-8. Vegetation cutting time series. Top Row: The Cape Fear River, North Carolina spill of a No. 6 fuel oil where the vegetation was cut in May 1985 (A). Two years later, the cut vegetation did not recover (B). Photo credit: Research Planning, Inc. Bottom Row: The *Grand Eagle* spill of a medium crude oil into the Delaware River in summer 1985 that was cut (C) but the vegetation recovered within two years (D). Photo credit: Tom Ballou.

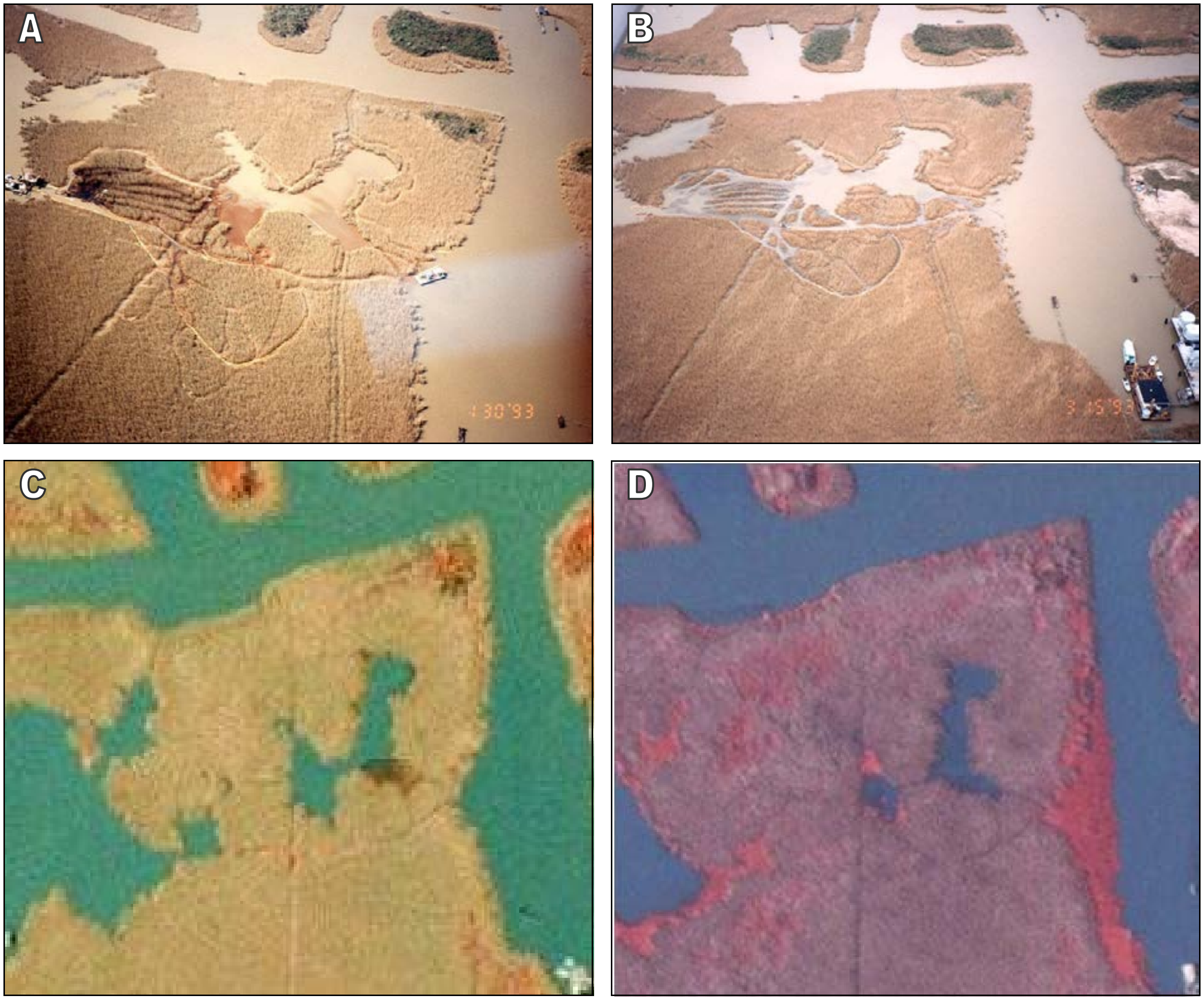

Figure 3-9. Time-series photographs of a spill in the Mississippi River birdsfoot delta, Louisiana in January 1993 where cutting of *Phragmites* was used to gain access to the interior where the oil was up to 7 cm thick. The oblique photographs were taken in 1993 before (A) and after cleanup (B); note the multiple paths cut to access the oil. The vertical images were taken five (C) and nine (D) years later, showing good vegetative recovery in five years. Photo credit: Dwight Bradshaw.

Table 3-2 is a summary of the studies on the effects of cutting of oiled marshes, updated from Zengel and Michel (1996) where they made a qualitative judgment on whether the effects were positive, showed no differences, or negative, based on the measured parameters and endpoints used in each study. One way to look at the results for all the cases in Table 3-2 is to consider the reasons for cutting and the potential consequences. If cutting is proposed to reduce the risk of continued oiling to wildlife or for aesthetic reasons, it is possible that 34% of the time, negative impacts to the vegetation could occur. If cutting is proposed to speed the recovery of the oiled vegetation, cutting is likely to be damaging or unnecessary for 66% of the time (sum of negative and no difference cases). Based on the 19 cases with data on direct comparisons, there is even a less likelihood that cutting will result in a positive effect on the vegetation, and cutting will do more harm or have no effect on vegetation recovery for 79% of the time. Because of these kinds of study results, cutting has not been used very often in recent times.

Table 3-2. Summary of the relative effects of oiled marsh cutting for all studies and those studies with direct comparisons with cut and uncut vegetation (updated from Zengel and Michel 1996).

Effect of Cutting	All Studies (# of cases)	All Studies (% of all cases)	Cut vs. Uncut Comparisons (# of cases)	Cut vs. Uncut Comparisons (% of cases)
Positive (+)	10	34	4	21
No Difference (=)	8	28	8	42
Negative (-)	11	38	7	37
Total	29	100	19	100

Some key observations on cutting of oiled marsh vegetation updated from Zengel and Michel (1996) include:

- The studies of marsh cutting that resulted in positive effects almost always included a heavy fuel oil or heavy crude oil. This would also apply to the *Deepwater Horizon* spill where the oil on the marsh platform was a thick, emulsified mousse that had properties similar to heavy oils.
- Most of the studies with positive effects were cases where the marsh was cut in fall or winter, when the plants are dormant and less likely to be stressed by both oil and vegetation removal. This effect was demonstrated by the experiments by Kiesling et al. (1988) where oiled vegetation cut in spring had lower recovery than those cut in winter.
- Cut vegetation that was submerged for a long period of time did not recover well, likely because the water layer would prevent oxygen transfer from air to the roots, which is essential for plant survival in water-logged, low-oxygen soils.

- Vegetation under salinity stress, such as water salinity that is higher or lower than normal, is more likely to have poor recovery after oiling and cutting.
- Physical damage from foot and vehicular traffic can cause additional damage to both the vegetation and the soils. Cleanup crews have to follow specific guidelines to minimize foot traffic during cutting, such as working only from boats, standing on firm (unoiled) substrates, or 100% use of walking boards.
- There is not enough information to state if there are any differences in recovery of cut vegetation among herbaceous (grassy) species.

Flooding and Low-Pressure Ambient-Temperature Flushing

Table 3-1 gives flooding and low-pressure, ambient-temperature flushing a grade of "B" for all oil types. The objective of these techniques is to flush floating oil that is trapped in the fringing marsh vegetation to open water for collection. Water pressure should not exceed 50 pounds per square inch (psi) to minimize sediment erosion. These techniques sound like they would be beneficial, mimicking the action of natural currents. In practice, however, pushing a liquid (oil) on a liquid surface (water) is hard, particularly because the water surface is flat. Large volumes of water are needed to be effective, requiring a lot of equipment and materials in terms of pumps, hoses, working platforms, recovery devices, etc. A nearby water source of the same salinity as in the treatment area is also necessary. One of the biggest challenges is to get "behind" the oil that is trapped in the vegetation so it can be flushed to open water where the oil can be contained with boom and recovered using vacuums, skimmers, or sorbents. Flushing operations have to consider tidal currents (flush on a falling tide) and wind (an onshore wind will push any released oil back onto the shoreline). Figure 3-10 shows the flushing operations at the Chalk Point spill site, demonstrating the complexity of the operations when the oil is in the marsh interior.

Figure 3-10. Intensive flushing operations along one of the trenches excavated at the Chalk Point, Maryland spill in April 2000. Flushing of oil from the marsh interior is very difficult. Photo credit: Jacqueline Michel.

Flushing can also be used to remove fluid oil stranded on the marsh surface. Figure 3-11 shows a barge-mounted flushing system developed during the *Deepwater Horizon* oil spill that was used to flush oil stranded on the marsh fringe in Louisiana. This approach allowed flushing to be directed from the landward side of the oiled band without placing equipment and crews on the marsh. The stranded oil was then flushed into the water where it could be collected. It worked well as long as the oil was liquid; however, the oil became too viscous to be mobilized by flushing over time.

Figure 3-11. The barge-mounted flushing system that had a long-reach mechanical boom with a spray bar attached, *Deepwater Horizon* oil spill. The angle and pressure of the water spray had to be adjusted to minimize sediment erosion. However, this technique was not effective because the oil was too viscous to be flushed. Photo credit: Scott Zengel.

Shoreline Cleaning Agents

Shoreline cleaning agents (also called surface washing agents) are products that contain surfactants, solvents, and/or other additives that work to remove oil from solid surfaces, such as seawalls and marsh vegetation, but does not involve dispersing or solubilizing the oil into the water column. They are sprayed on the oiled vegetation, allowed to soak for a short period, then the oil is removed by flushing, taking care to recover the released oil, most often using sorbents. Many products promoted as shoreline cleaning agents are essentially industrial cleaners that emulsify the oil, much in the same way that dishwashing soap cleans the grease off dishes. The treated oil is broken into small droplets that are kept in suspension by the surfactant. These products are called “lift and disperse” types, and they should not be used in any manner during an oil spill where they or the treated oil will be released to the environment. However, there are products that meet the “lift and float” description, where the product increases the effectiveness of flushing to remove and recover the oil. Refer to the "Selection Guide for Oil Spill Response Countermeasures" for more information about shoreline cleaning agents and their behavior and toxicity (online and interactive via http://nrt-sg.sraprod.com/build/#. As indicated in Table 3-1, they would be considered for use for medium and heavy oils that thickly adhere to the vegetation.

Pezeshki et al. (1995, 1997, 1998, 2001) conducted a series of laboratory and field experiments where they applied crude oils and Bunker C fuel oil to oiled salt and freshwater marsh plants in Louisiana, then applied the surface cleaning agent Corexit 9580 on some of the plants 1-2 days after oiling to compare impacts and recovery with oil alone. Note that they mostly applied the oil to the vegetation. They found that using Corexit 9580 on plants oiled with the crude oils had some short-term benefits of increasing gas exchange of the vegetation and decreasing leaf death, but the long-term outcome was similar regardless of treatment. They concluded that use of a shoreline cleaning agent with crude oil spills did not have any long-term positive or negative impacts on the recovery of oiled marshes. However, use of Corexit 9580 increased plant survival compared to oil alone for the Bunker C treatments.

Bizzel et al. (1999) conducted a similar field experiment in Texas using weathered Arabian Medium crude oil and a high and low dose of Corexit 9580 24 hours after oil application. They found that use of the cleaner did not affect microbial populations or the removal of oil from the top 5 cm of marsh soils.

One key point in these studies is that the shoreline cleaning agent was applied 1-2 days after oiling, which is not likely to occur during a real spill because of the time it would take to decide to use them, then get approval for their use. During the *Deepwater Horizon* spill, surface washing agents on oiled salt marsh test plots were not effective during testing in October 2010. Teas et al. (1993) found that use of shoreline cleaning agents helped with mangrove survival if applied within seven days, but not after longer periods.

Responders have considered using shoreline cleaning agents on oiled marshes to reduce the contact hazard to wildlife using the marsh. Michel et al. (1998) tested the use of Corexit 9580 on salt-marsh vegetation in Maine nine days after oiling by an IFO 380 (a moderate-heavy fuel oil) in late September. The agent removed about 50% of the oil on one side of the leaves; in comparison, ambient temperature water flushing removed no oil. Full-scale use was not recommended because very little oil was recovered; instead, a large amount of the released oil became suspended in the water and was not contained by boom or sorbents. Also, the logistics to apply the product to the wide band of oiled marsh in an area with a 3 m tidal range proved very difficult. One possible application might be to clean fringing vegetation along rivers and lakes, where the water level changes are relatively small. The marsh fringe is an important edge and transition zone that is heavily used by fish, invertebrates, and birds; thus, speeding the removal of oil as a contact hazard could have ecological benefits other than vegetation survival.

Enhancing Bioremediation (Nutrient Enrichment and Soil Oxidants)

Nutrient enrichment is a type of bioremediation that involves the addition of nutrients (generally nitrogen and phosphorus) to the marsh to accelerate the degradation of oil hydrocarbons by natural microbial processes. It is one of the least intrusive treatment options available for marshes. There are many types of fertilizers that can be utilized to supply the soil with the needed phosphate, nitrogen, and any other limiting nutrients; however, they can be categorized as one of three types: water-soluble inorganic nutrients, slow-release fertilizers, and oleophilic fertilizers. Nutrients can be applied by hand to specific areas or by aerial spraying of granules from a helicopter (as was done for the Chalk Point, Maryland spill in 2000).

In 2004 the U.S. Environmental Protection Agency published a comprehensive summary of bioremediation options for oil spills in salt marshes and relevant literature, and provided guidelines for design and planning of bioremediation treatments in salt marshes (Zhu et al. 2004). They published a similar work that included freshwater wetlands (Zhu et al. 2001). Both reports provide objective, scientific reviews of all the field and laboratory studies done at that time, and there has been little additional research of bioremediation since then that changes any of their conclusions. Recent reviews of bioremediation of coastal environments (Nikolopoulou and Kalogerakis 2009; Mercer and Trevors 2011) have come to the same conclusions. The key point about nutrient enrichment in marshes is this statement in Zhu et al. (2001):

> "*all the nutrients in the world would not stimulate biodegradation if oxygen were the primary limiting material*."

There are few feasible techniques to increase the availability of oxygen in fine-grained, organic-rich marsh soils; those techniques used on land, such as tilling, forced aeration, and the addition of chemical oxidants, are too damaging to marsh soils. The only "successful" treatments using nutrients to speed the microbial degradation of oil in marshes were where the oil was on the marsh surface, not penetrated into the soils. In these studies, the addition of nutrients did speed the rate of loss of the alkane fractions (the most readily degraded components in oil) but, at the end of the study (usually several months), the differences in degradation between treatments with and without addition of nutrients were small. Field studies and most laboratory studies were unable to demonstrate any increase in the rate of loss for the aromatic fractions, those that contribute most to the chemical toxicity of oil.

One exception was the series of greenhouse experiments by Mendelssohn and Lin (2002), where they were able to increase the rate of loss of the aromatic fractions with application of fertilizer, a soil oxidant (that converts slowly to hydrogen peroxide, providing a source of oxygen), and KH_2PO_4 to buffer the high pH that might be caused by the soil oxidant, but only in sods where the water table was kept at 10 cm below the marsh surface. Although the rate of loss of the aromatic fractions increased, the researchers ultimately concluded that the losses were likely due to the addition of the KH_2PO_4 rather than the soil oxidant. In another set of experiments, Mendelssohn and Lin (2002) compared application of fertilizer, microbial seeding, and soil oxidants on vegetation sods with mineral and sandy sediments. They found increased oil degradation, including the aromatics, four months after application of fertilizer, but not the microbial seeding or soil oxidant.

Zhu et al. (2004) conclude by saying that on some coastal wetlands, nutrients might still be a limiting factor and nutrient addition could speed oil degradation if the oil does not penetrate deeply into the anoxic zone of the marsh soils. They also point out that nutrient addition could stimulate plant growth, which could accelerate the overall recovery of the habitat. Several studies have been done to test this assumption, with conflicting results (Lin and Mendelssohn 1998; Lee et al. 2001; Mendelssohn and Lin 2002; Tate et al. 2012). Therefore, adding fertilizers may or may not have an effect on vegetation growth, depending on site conditions.

These conclusions mostly apply to freshwater marshes as well (Zhu et al. 2001). The main differences are that freshwater environments do not have the daily tidal flushing regime that can quickly wash out applied nutrients, so the amendments can last longer; and some freshwater wetlands can be nutrient limited, particularly highly organic peat and tundra environments.

Sometimes, an argument is made that adding nutrients, just in case they might be helpful, at least doesn't do any harm. However, any addition of nutrients to an oiled marsh needs to be based on site-specific considerations and good science.

Marsh Responses to *In Situ* Burning

A review of the literature and spill histories provided by responders identified 30 oil spills, three field experiments, and three laboratory studies where *in situ* burning (ISB) was conducted in marshes. Appendix D summarizes these 33 cases in chronological order. Of the 27 oil spills, 23 were light to medium crude oils and 4 were light refined products.

Vegetation Recovery after In Situ Burning: For those 21 spills (including two field experiments) listed in Appendix D where the vegetative recovery was documented from field studies or estimated based on the degree of recovery as of the last field survey, the vegetation is estimated to have recovered within:

- 1.5 years for nine spills (43%);
- 1.5 to 5 years for seven spills (33%);
- 5 to 10 years for two spills (10%); and
- Greater than 10 years for three spills (14%).

Of those three spills with greater than 10 years of recovery, two were in muskeg and peat soils, and one was the Chiltipin Creek site in Texas where other factors (drought, feral hog, and seismic survey damage) likely extended the recovery beyond 10 years. The two spills with vegetative recovery estimated to have occurred within 5-10 years were the Meire Grove site that had extensive physical damage resulting from other cleanup activities pre- and post-burning and the Lafitte Oil Field Site 3, where a site visit in year eight found the vegetation mostly recovered but lower in species richness and some elevated TPH in the soils.

> *Based on these results, when an ISB is used as an oil spill countermeasure in a wetland, if done following appropriate guidelines, the vegetation is likely to recover within five years, and more likely within 1-2 growing seasons.*

Figures 3-12 and 3-13 shows time-series photographs two marsh burns in Louisiana. Baustian et al. (2010) studied the recovery of the Chevron Empire marshes: plant biomass and species composition returned to control levels within nine months; although species richness remained somewhat lower. Aboveground and belowground plant productivity recovered within one growing season. They concluded that burning was very effective in allowing ecosystem recovery for oiled marshes.

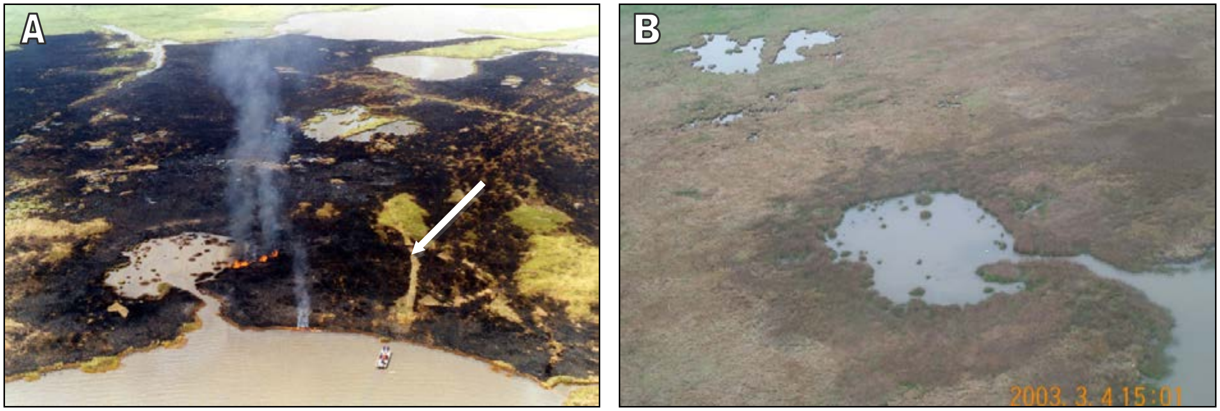

Figure 3-12. Mosquito Bay, Louisiana *in situ* burning of a condensate spill in a brackish water marsh. A: April 2001 right after the burn. The arrow points to the fire break created by laying down the vegetation with airboats. Note that the fire mostly burned to the downwind water edge. B: Same area in March 2003, showing good recovery of the vegetation. Photo credit: Louisiana Oil Spill Coordinators Office.

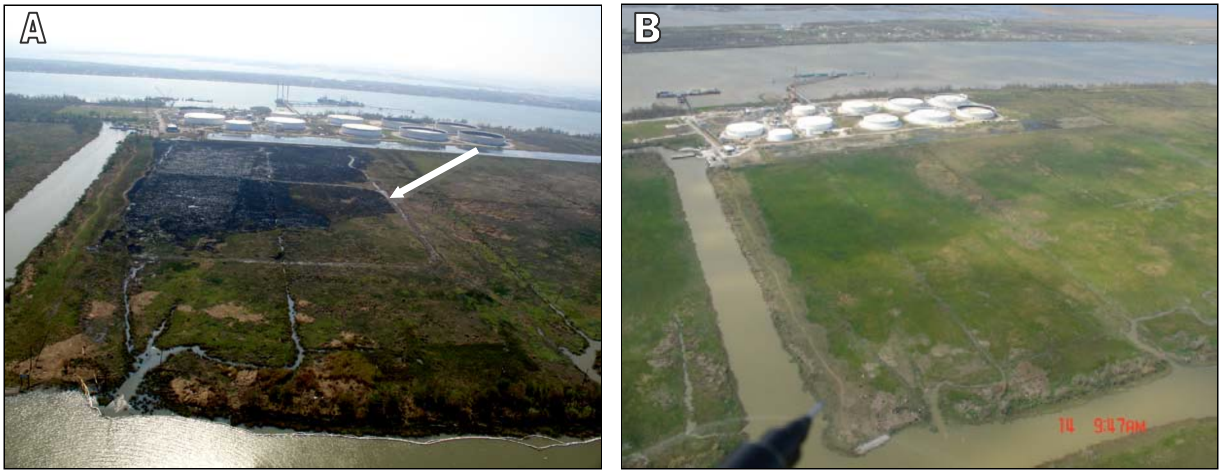

Figure 3-13. Chevron facility near Empire, Louisiana where in situ burning was conducted in a brackish water marsh. A: October 2005 right after the burn. The arrow points to the fire break created by laying down the vegetation with airboats. Photo credit: Amy Merten. B: March 2006, five months after the burn, showing good recovery of the vegetation. Photo credit: Gary Shigenaka.

<u>Oil Behavior and Weathering in Soils after In Situ Burning</u>: Most studies have documented that burning results in removal of most of the oil on the marsh surface, and residual concentrations generally decreased over time. Even at the Chiltipin Creek, Texas site, where TPH concentrations in the soil remained elevated in small areas for three years, by year five, the PAH concentrations in these small areas decreased to very low levels (Hyde et al. 1999).

Penetration of oil into marsh soils is of particular concern because of the slow rate of weathering in fine-grained, organic soils with low oxygen and flushing rates. Both field and laboratory burns have shown that burning does not remove any of the oil that has penetrated into the marsh soils. The Mosquito Bay, Louisiana spill of condensate was not burned until days 7 and 8 after the release, thus oil penetrated into the numerous fiddler crab burrows. After the burn, the condensate was readily visible in most burrows (Figure 3-14B); in fact, the oil would pool on the surface in footprints created by observers, then burst into flame because the soils were still hot enough to cause ignition of the vapors when exposed to air on the surface. Oil remaining in burrows was also noted at the Chevron Empire spill in Louisiana after Hurricanes Katrina and Rita (Figure 3-14A), when the oil stranded on the high marsh surface for weeks before it was burned (Merten et al. 2008).

Williams et al. (2003) also noted that the diesel penetrated into the sediments at a spill north of the Great Salt Lake, Utah and was not removed in the burn conducted six weeks after the release. The PAH concentrations actually increased after the burn, which they suggested was due to wicking of the oil in the soils by the heat of the burn. Eventually, the areas of persistently elevated PAHs in the soils were tilled and fertilized.

One common feature of these examples, where oil penetrated into the marsh soils that was not removed during the burn, is that the oil remained in the marsh for at least one week prior to the burn. Rapid removal of oil by burning would help reduce the potential for deep penetration and less efficient removal during a burn.

The very long recovery for the ISB in highly organic soils (peat, fen, muskeg) is directly related to the deep penetration of oil into these soils when the water table is below the surface. The heat of the fire reduces the viscosity of the oil, and it readily penetrates the loose organic soils. The Kolva River burn was conducted without any approvals and resulted in oil penetration of over 1 m (Hartley 1996).

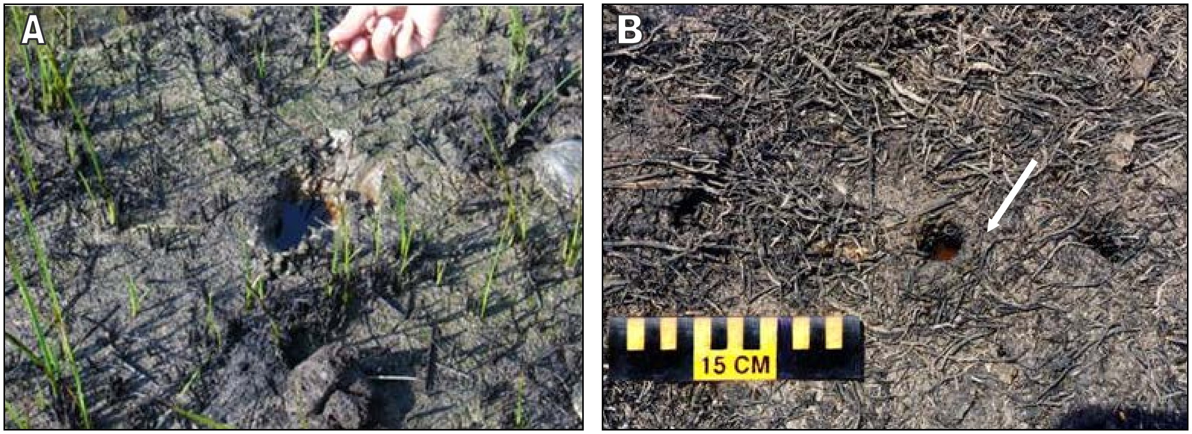

Figure 3-14. Left: *in situ* burning will not remove oil that has penetrated into the marsh soils. A: Chevron Empire burn; B: Mosquito Bay, Louisiana burn. Arrow points to the unburned, liquid oil in the burrow. Photo credit: Jacqueline Michel.

Blenkinsopp et al. (1996) found oil penetration to 40 cm in the bogs in northern Canada; the oil was only lightly weathered even after 24 years. They also noted that thick waxy crusts (burn residues), though highly weathered, formed physical barriers to plant regrowth. For other ISBs in marshes, the oil mostly stayed on the surface and was removed by natural weathering processes within a year or so (see Appendix D).

Mendelssohn et al. (2001) included in one of their laboratory experiments a study to determine if ISB affected the removal rate of oil penetrated into marsh soils. They added a small amount of either diesel or crude oil to the surface of the potted plants 24 hours prior to ignition in the burn tank—not enough to affect the vegetation, but enough to be able to track any reductions due to the burn. They found that burning with +10 cm, 0 cm, and -2 cm of water over the plants did not reduce the amount of the crude oil added to the soils, but reduced the amount of diesel added by a factor of 10. It is likely that elevated temperatures more readily mobilized the low-viscosity and less sticky diesel.

In summary, ISB in marshes and organic soils results in rapid removal of surface oil, but it will not remove oil that has penetrated into the soils. Under ideal conditions, there will be little subsurface oil; however, burns in peat soils can result in deeper penetration of oil into the subsurface. It is important to remove the burn residues shortly after the burn (flushing, manual removal, use of sorbents) because it has been shown that these residues weather slowly and can delay habitat recovery.

<u>Faunal Recovery after In Situ Burning:</u> There are very limited data on the impacts to marsh-associated fauna during an ISB and the relative rate of recovery after the burn. If studied at all, data are available for at most one year post burn. For the March 1993 burn of aviation fuel in a snow/ice covered pond at the Naval Air Station Brunswick, Maine, studies of fish, birds, mammals, and benthic communities showed normal species abundance and composition by summer (Metzger 1995). At the Meire Grove spill in Minnesota, again of light refined products that were burned in small pond in September 1992, initial impacts to benthic invertebrates were severe, but after one year, Zischke (1993) noted that there was considerable recovery, with higher numbers of invertebrates from the oiled/burned pond and higher midge species richness, compared to a control pond. Holt et al. (1978) documented impacts to invertebrates for the first month after a crude oil burn but recovery within six months for a small burn area in Texas in October, whereas McCauley and Harrel (1981) reported reduced invertebrate abundances in both oiled/burned and clean/burned study plots versus other treatments and controls in a brackish marsh along the Neches River in Texas six months after a January burn of crude oil. It should be noted that vegetative recovery for the Neches River burn was poor as well, due to high levels of fresh water due to floods. Michel et al. (2002) reported seeing large numbers of fiddler crabs six months after the Mosquito Bay, Louisiana ISB. Martin (2010, pers. comm.) reported seeing fresh crayfish burrows the day after the burn at Refugio, Texas. Tunnell et al. (1995) found differences in the fauna in ephemeral ponds for two oiled/burned ponds versus an unoiled/unburned control for two years after the burn at Chiltipin Creek, Texas, but not in year three (though there was very high variability in all years). Mendelssohn et al. (1995) reviewed the limited prescribed burning literature on impacts of burning (without oil) on fauna and found a few studies that showed no significant effects.

With such limited data, it is hard to make anything but general statements, such as, animals at the surface are likely to be killed if they are not able to escape into burrows or move out of the burn area. There is evidence that burrowers can survive the temperature effects of burning. Recovery is likely better if there are no burn residues or the residues are removed.

Guidelines for Considering *In Situ* Burning of Oil Spilled in Marshes

Oil spilled in marshes poses many difficult tradeoffs in terms of the potential impacts of the oil versus different response options. For ISB, the evaluation of the tradeoffs usually has to be conducted quickly, before the oil spreads, penetrates into the soils, weathers, or changes in some way that makes ISB less effective. In this section, guidelines for considering when to use ISB in a marsh are discussed, with as much scientific data to support them as possible.

Time of year: Though it is not possible to pick the time of year for a spill to occur, responders need to consider the time of year in determining how quickly vegetation may recover from a burn. Mendelssohn et al. (1995) assessed studies of prescribed burning (for habitat management) where burning resulted in an increase, decrease, or no change in plant growth compared to appropriate controls, by season. They reviewed 34 studies where recovery times were less than 1.5 years and 20 studies where recovery times were greater than 1.5 years. Burns in summer had the highest percentage of events that resulted in a decrease in vegetative growth. For burns with recovery times less than 1.5 years, 55% of the burns in summer resulted in a decrease in vegetative growth compared to 20% in fall, 33% in winter, and 11% in spring. For burns with recovery times greater than 1.5 years, the percentage of burns that resulted in a decrease in vegetative growth were 42% in summer, 25% in fall, 0% in winter, and 0% in spring. These studies showed that, regardless of season, for 68-80% of the time, prescribed burning resulted in vegetative growth that was equal to or greater than controls.

> *The rule of thumb, based on both understanding of the life history of plants and prescribed burning studies, is that vegetation recovery is likely to be slowest if burned during the summer and fastest if burned in the winter and early spring.*

Plant Species: Species vary in their tolerance to fire as seen in the prescribed burning and fire ecology literature (e.g., Nyman and Chabreck 1995), and thus in their likely response to ISB as a treatment option. Dahlin et al. (1999) provide a detailed, species-by-species summary of what is known from the fire ecology literature and an evaluation of the potential for using ISB for the following plant communities: trees, shrubs, grasses, desert habitats, and wetland grasses and sedges. All grasses and sedges were considered to have high or very high potential for a successful ISB, with the exception of *S. patens*, which was considered to be moderate-high because it can occur in high salt marshes where the soils may not be wet or flooded, potentially leading to longer recovery times and changes in the vegetative community.

Lin et al. (2005) noted that recovery after their ISB laboratory experiments was species-specific when there was not a water layer over the marsh soils during the burns. *Sagittaria lancifolia* and *S. alterniflora* are species that have large and/or shallow rhizomes that were affected more by burning, whereas *S. patens* and *D. spicata* are species that can have very dense stems (up to 5,000/m^2) and rhizomes occurring at deeper depths where thermal stress from burning is reduced. They also found that *S. patens* and *D. spicata* quickly generated new shoots from surviving rhizomes, thus were able to outcompete other species in the first several months. However, over time, the other species were able to catch up and the vegetation returned to its normal species composition. They concluded that surviving rhizomes of *S. patens* and *D. spicata* could rapidly recover after burning. This rapid regrowth of vegetation is important because the aboveground vegetation provides a pathway for oxygen transfer from air to the roots, which is essential for plant survival in waterlogged, low-oxygen soils.

However, species responses to oiling and burning can vary, depending on other factors. Lindau et al. (2003) found rapid recovery of stem height and density and carbon fixation after a field ISB experiment for both *S. alterniflora* and *S. lancifolia* after one year, with aboveground biomass higher than controls. They suggested that these species might be utilizing oil and dead vegetation from the burn as sources of nutrients.

<u>Marsh Soil Type</u>: The biggest concern with the use of ISB in marshes is for highly organic soils where the peat soil itself could ignite, causing lowering of the marsh elevation, damaging roots and the seed bank, etc. Oil degradation rates for subsurface oil in acidic, anaerobic soils are slow and can take many decades (more than 24 years as reported by Blenkinsopp et al. 1996). The amount of litter on the marsh surface at the time of the burn can also influence the recovery and composition of the vegetative community. Pahl et al. (1997) suggested that the ISB at the Rockefeller Refuge in Louisiana removed the litter, which favored the rapid growth of *S. robustus* over the pre-burn dominance of *D. spicata* and *S. alterniflora*. There are similar examples from the prescribed burning literature.

<u>Water Levels during a Burn</u>: Soil temperatures of 60-65°C are lethal to plants. Therefore, whether conducting a prescribed burn or responding to an oil spill, it is always recommended (but not required) that standing water should cover the marsh surface during the burn, to protect plant rhizomes from thermal stress and prevent ignition of organic soils. For oil spills, an additional benefit of a water layer is prevention of oil penetration into the marsh soils. The marsh sites, and the locations within some marshes, with some of the longest recovery periods include those that had little to no water present during the burn, such as Chiltipin Creek, Texas which was predicted to take 14-15 years to fully recover to its climax species distribution (Hyde et al. 1999).

Lin et al. (2002a, 2005) conducted a series of burn-tank experiments that replicated in-situ burn temperatures, with thermocouples inserted into the marsh soil of potted plants at different depths to help answer the question of how much water was enough to protect the plants during ISB. Their first study (Lin et al. 2002a) showed:

- A water layer of 10 cm was ample to protect the marsh soil from burning impacts, with soil temperature below 37°C and plant survival and regrowth high;

- A water table 10 cm below the marsh surface resulted in soil temperatures of 120°C at 2 cm soil depth and almost no post-burn recovery of *S. alterniflora*; and
- At water levels of 0 and 2 cm over the marsh surface, the soil temperatures were low enough for the plants to survive, but they died from exposure to the diesel oil used in the experiment.

With these results, Lin et al (2005) conducted another set of experiments to separate the oil stress from the thermal stress at water levels less than 10 cm over the soil surface. They also wanted to determine if the effect of ISB differs with the marsh type and oil type burned. This second study showed:

- Water layers of 2 and 10 cm overlying the soil surface were sufficient to protect marsh vegetation of all three types of marshes from burning impacts. Soil surface temperatures did not exceed 40°C with 10 cm and 50°C with 2 cm of water overlying the soil surface;
- A water table 2 cm below the soil surface resulted in soil temperatures of >100°C at 0 cm to <40°C at 5 cm below the soil surface and higher impacts to *S. alterniflora* (30% reduced survival) and *S. lancifolia* (50% reduction in survival) because these species have rhizomes close to the surface; and
- *S. patens* and *D. spicata* were not affected by ISB with the water table 2 cm below the soil surface (dense stems and deeper rhizomes).

Experience during ISBs at actual spills also indicates that, as long as the marsh soils are water saturated, the plants will mostly survive. More water is better, but not essential. However, burning of oil on dry marsh soils should be carefully considered in terms of the tradeoffs associated with different response options and resources at risk.

<u>Flooding Post-burn</u>: Studies of prescribed burns have shown that certain species are more likely to die if they are completely submerged under water for several weeks after the burn. *D. spicata, Panicum hemitomon,* and *Typha* spp. are particularly sensitive to post-burn submergence (Dahlin et al. 1999). Prescribed burns are often scheduled in the fall, when water levels are low, so the plants are better prepared for spring flooding. McCauley and Harrel (1981) attributed the very poor recovery of *S. patens* after test burning of a spill in the Neches River, Texas to persistent flooding for months. Pahl et al. (2003) also noted slower recovery of *D. spicata* when flooded after burning. Holt et al. (1978) reported the lowest recovery of a heavily oiled *S. alterniflora* occurred in an area of standing water.

<u>Oil Type</u>: Oil type and degree of weathering will influence the efficiency of the burn and the potential for, thickness, and type of burn residues remaining on the marsh surface. Heavier oil and more weathered or emulsified oil generate more burn residue. Table 3-3 summarizes the likely behavior of

burn residues from different oil types when burned on land. In addition, the burn residue from heavier oils can be heavier than water and sink, a behavior that is more likely for spills in freshwater habitats. Laboratory studies have shown strong correlation between the densities the original oil and the resulting burn residue: crude oils with densities greater than 0.864 g/cm^3 (or API gravity less than 32) are likely to produce burn residues that sink in seawater (S.L. Ross Environmental Research Ltd. 2002).

Table 3-3. Behavior of burn residues by oil type for on land burns (from Scholz et al. 2004).

Oil Type	Behavior of Burn Residue on Land
Gasoline products	• Will burn; will not leave a significant amount of residue.
Diesel-like products and light crude oils Diesel, No. 2 fuel oil, Light concentrate, West Texas crude oil	• Burn residue is mostly unburned oil that penetrated into the ground, root cavities, and burrows with small amount of soot particles that can be enriched in heavier PAHs. • Remains liquid; can be recovered with sorbents and flushing.
Medium crude oil and intermediate products South Louisiana crude oil, IFO 180, Lube oils	• Burn residue can be pockets of liquid oil, solid or semi-solid surface crusts or sheets, and heavy, sticky coating on sediments. • Liquid oil can be flushed. Semi-solid and solid residues can be manually removed. • Remaining residues can be tilled and fertilized in appropriate habitats.
Heavy crude oils and residual products Venezuela crude, San Joaquin crude, No. 6 fuel oil	• Difficult to burn, so often have to add a lighter oil to start the burn. • Leaves heavy, sticky residue that is a mix of unburned oil and semi-solid burn residue, requiring extensive cleanup. • Remaining residues can be tilled and fertilized in appropriate habitats.

Another factor concerning oil type (other than safety issues) is the toxic effects of the oil on the marsh community prior to the burn. Lin et al. (2005) did not detect any differences in response of ISB of diesel versus crude oil in their burn tests. However, several spills have shown that light fuel oils and condensates caused plant mortality during the period that the oil was in the marsh prior to the burn, such as the Mosquito Bay and Sabine Point spills of condensate in Louisiana and the diesel spill in Corrine, Utah (Michel et al. 2002). Burning under these conditions will not avoid vegetation and faunal mortality from oil exposure prior to the burn.

Fire Control: For most ISBs in marshes, the fire is extinguished when it reaches unoiled vegetation, particularly during the growing season when the vegetation is live. At this point, the smoke goes from black with soot to white with water vapor. However, real "control" of a fire in a marsh during a spill emergency is difficult, and responders have to be prepared for the fire spreading to unoiled areas. In two of the case histories in Appendix D, the burned area was much larger than the oiled area. In the Mosquito Bay spill, the burned area was eight times the oiled area (4.9 ha vs. 40 ha; Figure 3-12); for the Louisiana Point spill, the burn area ten times the oiled area (5.3 ha vs. 55 ha). The types of firebreaks possible in a marsh, such as laying down and wetting the vegetation using an airboat, are not sufficient to contain a hot fire. The burn can spread to unoiled areas at sites: 1) that have not been burned recently (thus have abundant natural fuel present); 2) where fire breaks cannot be completely cleared; 3) without a lot of free-standing water; and 4) with dry or dead vegetation.

Selecting Appropriate Cleanup Endpoints for Marshes

The NOAA Shoreline Assessment Manual (NOAA 2013) includes a discussion of the process for establishing cleanup endpoints for different habitats. Cleanup endpoints appropriate for marshes are generally as follows:

- No free-floating oil in the marsh
- No oil on vegetation that can rub off on contact
- No oil greater than 0.5 cm thick on the marsh platform
- As low as reasonably practicable, considering the allowed treatment methods and net environmental benefit

It is the last cleanup endpoint that requires the most discussion in terms of the tradeoff between the degree and duration of impacts from the oil versus the degree and duration of impacts associated with removal actions. From the discussion of cleanup methods in this chapter and the rates of recovery of oiled marshes in Chapter 2, clearly marshes most often recover on their own within 1 year for light to moderate oiling. In most cases, natural recovery is the best option. However, when marshes are heavily oiled, and particularly with thick oil on the marsh surface, removal actions are often needed to remove as much of the oil as needed to speed the overall rate of recovery, without causing more harm than good.

Restoration as Part of the Response

Marshes that are severely affected by either the oiling or response operations may be more susceptible to habitat loss by enhanced erosion during the time it takes for the vegetation to naturally recover. In these cases, it may be necessary to include restoration actions as part of the response.

Figure 3-15 shows the benefits of this kind of restoration effort to quickly re-establish healthy vegetation at a site in Louisiana following the *Deepwater Horizon* oil spill. The site was a research effort and not part of the response. But, it obviously was effective.

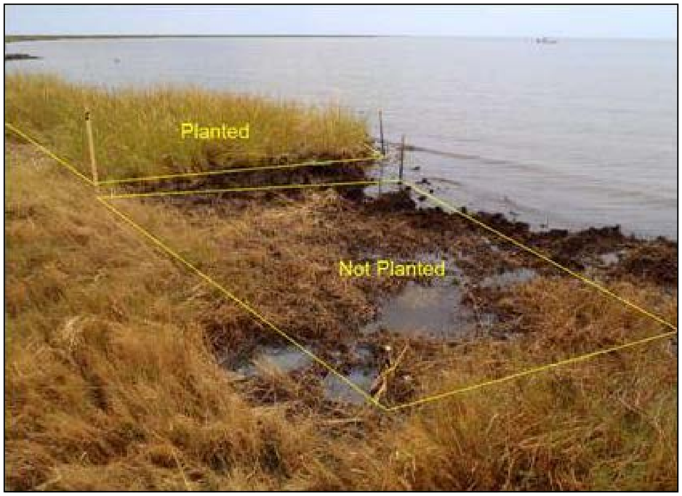

Figure 3-15. Tulane University research project where *S. alterniflora* was planted (bare root) along the heavily oiled and highly erosional shoreline in N. Barataria Bay, Louisiana, immediately following oil cleanup treatments. The treated and planted plot had good vegetative cover as of September 2012, whereas the treated but unvegeted plot had higher shoreline erosion. Photo credit: Scott Zengel.

When oil removal requires intrusive methods that damage the marsh vegetation, it may be necessary to conduct marsh restoration. Removal of oil from the marsh interior during the Chalk Point spill in Maryland required extensive trenching (Figure 3-4). Once the response operations were terminated, the Responsible Party conducted a marsh restoration project that involved filling back in of the trenches and re-planting of the vegetation. Figure 3-16 shows the photographs of one of the heavily disturbed but restored areas only one year after vegetation re-planting. According to Gundlach et al. (2003), vegetative recovery was 70-80% after one year, and nearly 100% after two years.

After the Arthur Kill/Exxon Bayway spill, some of the areas where the vegetation died and did not re-establish were replanted three years later. Bergen et al. (2000) monitored vegetative recovery at marshes along denuded/planted marshes, denuded/not planted, and unoiled marshes over the period 1994-1997 and found that the planted areas recovered well.

Based on these results, replanting of marsh areas with high vegetation mortality should be of high priority.

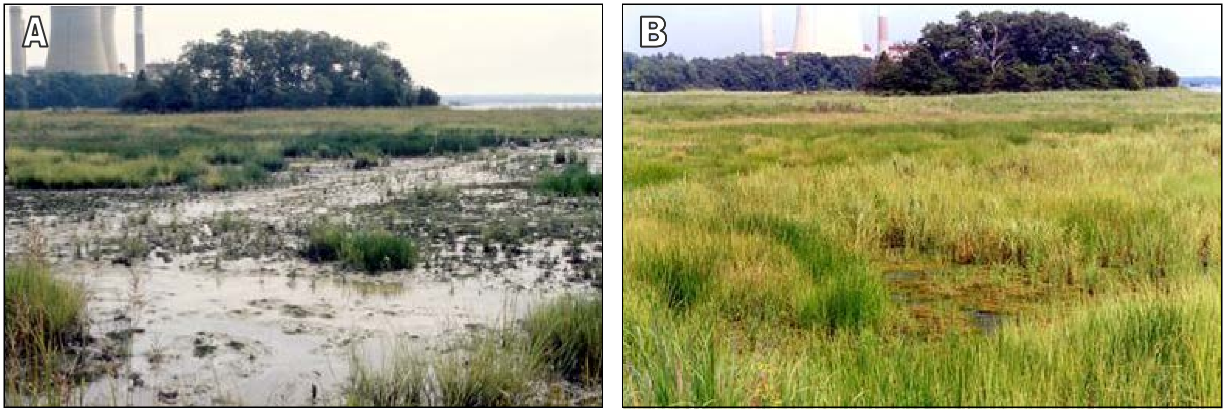

Figure 3-16. Restoration of the area of trenching and flushing (Figure 3-4) at the Chalk Point spill site. A: July 2000; B: July 2001, one year later. Photo credit: Jacqueline Michel.

Selecting Appropriate Response Options for Speeding Recovery of Oiled Marshes

Table 3-4 provides a matrix of likely marsh oiling conditions and potential response options, along with guidance on key issues and constraints based on the information summarized in Chapters 2 and 3. Again, it is important to note that often multiple response options will be used during a spill, for different oiling conditions or different phases of the response.

Table 3-4. Guidance on selecting appropriate response options for oiled marshes.

Oiling Condition	Response Options	Key Issues/Constraints
Free-floating oil on water in the marsh	*In situ* Burning	- Safety, fire control, sufficient water layer or saturated soils, oil type (mousse not likely to burn), amount of oil residue that will still need removal, time of year, species sensitivity, marsh soil type (peat soils are highly sensitive), flooding post-burn could cause plant mortality
	Vacuum	- Can remove large amounts of oil quickly before it becomes stranded, work from boats at water's edge will limit access to interior oil, ability to concentrate the oil to increase effectiveness, need to decant water to improve efficiency, avoid foot traffic on marsh surface
	Low-pressure Flushing	- Access, particularly ability to generate enough flow to push oil towards recovery devices, high oil viscosity will reduce effectiveness, potential to disturb soils
	Sorbents	- Loose sorbents (pads, snare) must be removed immediately, use walking boards or deploy from boats, can be slow and labor intensive

Oiling Condition	Response Options	Key Issues/Constraints
Thicker oil (>0.5 cm) on marsh surface	Natural Recovery	- Degree of exposure to physical removal processes, potential for exposure hazards for animals and long-term impacts to vegetation
	Manual Removal (rake, scrape)	- Access, use walking boards, risk of damage to live vegetation and disturbing soils, can speed of weathering of residues, use loose sorbents as temporary contact barrier after treatment
	Vacuum	- Access, avoid foot traffic or use walking boards, potential to gouge the marsh soils and remove vegetation, likely to leave thick patches, use low-pressure flushing to increase oil removal
	Low-pressure Flushing	- Access, particularly ability to generate enough flow to push oil towards recovery devices, high oil viscosity will reduce effectiveness, potential to erode soils
	In Situ Burning	- Safety, fire control, saturated soils to prevent oil penetration into the soils, time of year may affect plant recovery, oil type (mousse not likely to burn), amount of oil residue that will still need removal, species sensitivity, marsh soil type (peat soils are highly sensitive), potential to change soil elevation if organic soils burn, flooding post-burn could cause plant mortality
Thinner oil (<0.5 cm) on marsh surface	Natural Recovery	- More likely to weather to a thin, dry crust and be removed by natural processes
	Same Options as for Thicker Oil	- Consider risks of causing more damage during removal actions compared to rate of natural weathering
Heavy oil on vegetation	Natural Recovery	- Preferred tactic, unless there are key species of concern at risk
	Passive Sorbents	- Use only as long as oil is being released, closely monitor to make sure that the sorbents are properly deployed, remove prior to high water or waves to prevent stranding in the marsh
	Loose Organic Sorbents	- Consider how long before the oil weathers to a dry coat, application should be only a thin coating on the vegetation, will be difficult to apply to marsh interiors
	Vegetation Cutting	- Consider only if there are key species of concern at risk, consider how long before the oil weathers to a dry coat, may need to cut accessways to reach interior oil, use walking boards, test different tools to determine best tactic
	Surface Washing Agents/Flushing	- Use when necessary to reduce contact hazard quickly, must wash to water (so only use when water levels cover the soils), use only products that lift and float, potential short-term increased aquatic toxicity
Light to moderate oil on vegetation	Natural Recovery	- Preferred tactic particularly for light oils, small areas, dormant vegetation, some exposure to waves and/or currents
	Passive Sorbents	- Use only as long as oil is being released, closely monitor to make sure that the sorbents are properly deployed, remove prior to high water or waves to prevent stranding in the marsh
	Loose Organic Sorbents	- Consider how long before the oil weathers to a dry coat, application should be only a thin coat on the vegetation, will be difficult to apply to interior of the marsh

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CHAPTER 4. MARSH CASE STUDIES

Much of what we know about the impacts of oil and response options on marsh habitats has been learned through observations at spills. Case studies provide the basis for evaluating the tradeoffs of different response options, both during an emergency response and in planning for spills. Many of the studies of past spills have been cited in Chapters 2 and 3. In this chapter, four case studies are summarized, focusing on different types of oil and treatment methods used, and highlighting the lessons that were learned and have influenced future spill responses. The case studies are presented in chronological order.

Barge Florida, Buzzards Bay, Massachusetts, September 1969

Acute Toxicity and Long-term Impacts of No. 2 Fuel Oil

Up to 185,000 gallons of No. 2 fuel oil were spilled from the barge *Florida* into Buzzards Bay, Massachusetts in September 1969 resulting in heavy oiling of the Wild Harbor estuary. This spill has been well studied for nearly forty years because of its close proximity to the Woods Hole Oceanographic Institute. Salt marshes died within a few weeks, and in heavily oiled sediments, all benthic life was killed (Sanders et al. 1980). Two years later, soils with greater than 1-2 mg/g oil contained no living plants; vegetation regrowth occurred by rhizome spreading from the edge of live vegetation (Burns and Teal 1979). The heavily oiled marsh areas had fewer benthic species, dominated by opportunistic species such as the polychaete *Capetilla captitata* that would bloom then crash, indicating poor recruitment for five years (Sanders et al. 1980). Krebs and Burns (1977) followed the impacts of the spill on fiddler crabs for seven years. Starting in 1971, they documented decreases in fiddler crab density, reduced juvenile settlement, heavy overwinter mortality, uptake of oil into tissues, and behavioral disorders including locomotor impairment and abnormal burrowing. They found correlations of these effects with the persistence of the alkyl naphthalenes (2-ringed PAHs) in the oil. Only when these compounds decreased in 1976-77 was there successful recruitment of juvenile crabs, which started the recovery of adult populations seven years after the spill.

Nearly 40 years later, Culbertson et al. (2007) documented that, in a small area that still contained relatively unweathered oil in the subsurface, fiddler crabs avoided burrowing into oiled layers, suffered delayed escape responses, had lowered feeding rates, and achieved 50% lower densities than in control areas. Studies 38 years after the spill showed that mussels transplanted into the oiled areas had slower growth rates, shorter mean shell lengths, lower condition indices, and decreased filtration rates, and salt marsh vegetation showed reduced stem density and above- and belowground biomass

(Culbertson et al. 2008a,b). Peacock et al. (2005) showed that the oil persisted in a narrow band several meters wide and about 50 m long in the mid- to lower intertidal zone adjacent to one tidal channel, in the area where the oil initially was reported as being the heaviest. Thus, the areal extent of the persistent oil is small relative to the initial oiled area. They found that the highest oil concentrations (1-14.1 mg/g TPH) were between 4-20 cm below the surface, and they estimated that 100 kg of oil remained, representing 0.02% of the original spill volume.

Many factors combined to cause the acute toxic impacts and persistence of the subsurface oil from the *Florida* spill: Initial heavy loading (the oil was pushed by winds and tides into the impacted bay and persisted there for many days), a tidal range of nearly 2 m (so that the oil that stranded on the marsh at high tide was able to penetrate the sediments as the tides and groundwater levels in the marsh dropped), organic soils with slow weathering rates, a net depositional area (with sediment accumulation rates of 0.35 cm/year; White et al. 2005); and a sheltered setting.

Amoco Cadiz, Brittany, France, March 1976

Intrusive Treatment Delays Marsh Recovery

The T/V *Amoco Cadiz* spilled 70 million gallons of Arabian and Iranian light crude oil off the coast of Brittany, France in March 1976. The extensive marsh at Ile Grande was heavily oiled, and the French military used vacuuming, high-pressure flushing, and excavation in attempts to clean the marsh (Figure 4-1). By 1978, there were extensive areas with no vegetation cover. In many areas, only the aboveground marsh vegetation and oil had been removed; in other areas the entire marsh surface including the root mat had been removed to a depth of over 30 cm, and the creek banks were almost completely lacking vegetation, leading to extensive erosion.

Seneca and Broome (1982) conducted experimental then larger-scale replanting activities to speed the rate of recovery. They eventually planted 9,700 transplants, half of them along the creek banks. Baca et al. (1987), in studies eight years later of the marshes that were intensively cleaned compared to oiled but not cleaned marshes and an unoiled marsh, found that the oiled but not cleaned marsh had recovered within five years by natural processes. In contrast, the oiled/cleaned/replanted marsh at Ile Grande took 7-8 years to recover based on field transect data. The slower recovery was attributed to the destruction and compaction of roots, removal of the marsh substrate, and erosion of channels due to the lack of vegetation along the channel banks. They found that plantings improved the rate of recovery because the vegetation stabilized open areas and provided attachment substrates for seeds and propagules, which sped the overall rate of revegetation (which was key to recovery of the marsh).

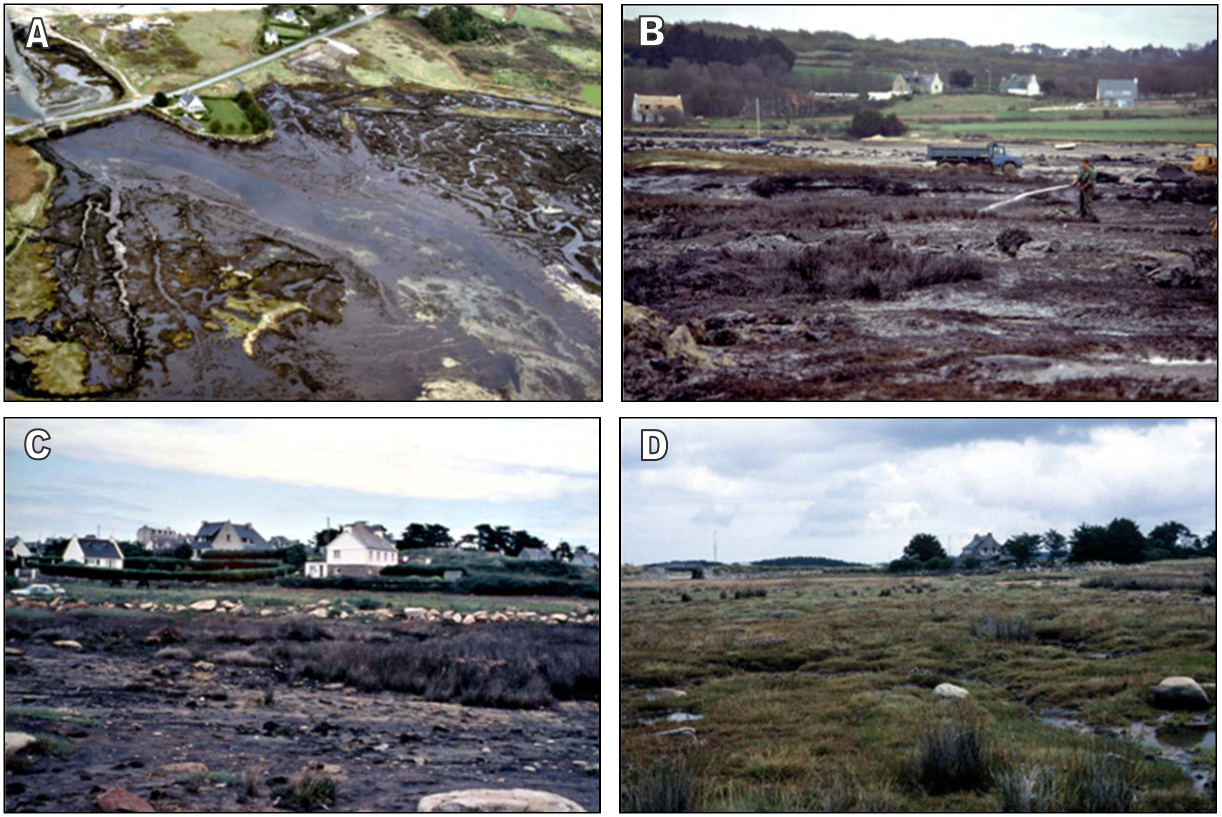

Figure 4-1. Heavily oiled marsh at Ile Grande, France from the *Amoco Cadiz* oil spill. A: Aerial view of the heavily oiled marsh in March 1978. B: High-pressure flushing during cleanup by the French army in April 1978. C: Condition of the marsh in Fall 1978 showing extensive removal of the vegetation and the substrate. D: Condition of the marsh in 1986, eight years later showing late vegetation recovery. Photo credit: A. Miles Hayes; all others: Erich Gundlach.

The rate of oil degradation in the marsh soils was a function of the initial degree of oil contamination, as studied by Mille et al. (1998) who collected soil samples seven times between 1978 and 1991. At the site with the lowest oiling (initially at 1,900 ppm TPH), the n-alkanes degraded within four years and all the oil was degraded after thirteen years. At sites with the highest oiling (33,000 and 230,000 ppm TPH), it took between 6-13 years for the n-alkanes to be degraded, and oil was still present thirteen years later.

Gilfillan et al. (1995) used historical aerial photograph from 1971 and 1990 to assess the long-term recovery of marshes that were cleaned and not cleaned. They found that the oiled and cleaned marshes at Ile Grande had between 23 and 39% less vegetated area, compared to an adjacent oiled and not cleaned marsh that had increased in area by 21%. They were able to map the distribution of marsh vegetation using aerial photographs and ground-control data into high marsh and low marsh. In 1971 prior to the spill, the cleaned marsh was composed primarily of high marsh; in 1990, the proportion of low marsh to high marsh increased significantly. In contrast, the composition of the marsh vegetation in the oiled and not cleaned marsh had not changed between 1971 and 1987. They attributed these changes in marsh coverage and type in the cleaned marsh to the removal of up to 50 cm of marsh soils during cleaning, which lowered the intertidal elevation of the marsh surface. Marsh vegetation is very sensitive to elevation and the frequency and duration of flooding. Because of the excessive sediment removal during cleaning, there was a shift in the vegetation to low- and mid-marsh species. Gilfillan et al. (1995) concluded that full recovery to pre-spill conditions will require sediment accretion.

This spill provided good scientific data that intrusive cleanup in a marsh will slow the overall rate of recovery, thus such treatment should be carefully evaluated, and greatly influenced future response strategies in spills around the world.

Chalk Point, Patuxent River, Maryland, April 2000

Long-term Monitoring of Heavily Oiled Interior Marsh

On 7 April 2000, an estimated 140,000 gallons of a mixture of No. 6 and No. 2 fuel oils were released into Swanson Creek, the Patuxent River, and downstream tributaries from a pipeline rupture going into nearby Chalk Point Power Generating Station. The spill affected an estimated 76 acres of brackish marsh (dominated by *S. cynosuroides* and *S. alterniflora)*, with extensive areas of heavily oiled interior marsh habitat. There was intensive treatment including trenching, flushing, and use of sorbents in accessible marsh areas (see Figures 3-4 and 3-15); however, there was no treatment in other heavily oiled interior marsh areas that had limited access. Because of the predicted long-term persistence of oil-related impacts, NOAA funded a study of the oiled wetlands in 2007, seven years after the initial spill (Michel et al. 2009).

Overall, the oil in the highly organic marsh soils had undergone little to no additional weathering since Fall 2000, based on comparisons of PAH depletion ratios from samples collected in Fall 2000, Summer 2001, and Summer 2007. There were likely two factors limiting natural weathering processes in the marsh soils: slow physical removal processes and low oxygen availability. The interior marsh habitat is flooded by daily tides through many small channels. During spring high tides, there can be 20-30 cm of water in the marsh. The marsh surface has a lot of micro-topography with low areas between dense clumps of stems that hold pools of water during low tide. The soils in these low areas are very soft and water saturated. During spring low tides, the marsh soils do drain as low as 30 cm, as evidenced by the fact that the oil penetrated to these depths in some areas. Tidal flushing may have been a mechanism for removal of bulk oil stranded on the surface initially; however, it would not be effective at mobilizing oil from below the marsh surface. There are few bioturbating benthic biota in these marshes. Photo-oxidation does not occur below ground. Therefore, the only other removal mechanism would be microbial degradation, which obviously is very slow in these soils. With the slow weathering of the oil, nearly half of the 24 soil samples collected in 2007 showed evidence of toxicity in amphipod toxicity tests.

Visually, the marsh vegetation looked like it had recovered; however, the stem density and stem height of *S. alterniflora* (but not *S. cynosuroides*) were significantly lower in the oiled versus unoiled sites. In contrast, belowground biomass was significantly lower in the *S. cynosuroides* habitats but not the *S. alterniflora* habitats. The reasons for these differences may be related to the relative distribution of above- versus belowground biomass and the types of biomass for each species. *S. cynosuroides* has more and larger rhizomes and the rhizome biomass has a peak at 10-20 cm; thus, this species was more likely exposed to the highly concentrated oil that persisted in the root cavities along the rhizomes. Some of the black oil observed in the cores occurred along rhizomes, which were partially hollow and dead. Roots and rhizomes in the soil would grow until they encountered zones of oil that would slow growth and could eventually lead to death. *S. alterniflora* has about an equal proportion of roots to rhizomes and the rhizomes are smaller, so any reductions in the biomass of the rhizomes may have had a lesser effect on the overall belowground biomass. Alternatively, the lower belowground biomass of *S. alterniflora* may be in less contact with the oil.

This study showed that marshes can grow in oiled soils, but there can be long-term sublethal effects than can reduce overall health and productivity of the marsh ecosystem.

Deepwater Horizon, Northern Gulf of Mexico, 2010

Intensive Treatment of Thick and Persistent Oil

The *Deepwater Horizon* spill released an estimated 4.9 million barrels of South Louisiana sweet crude into the Gulf of Mexico over an 87-day period, from 20 April to 15 July 2010. The heaviest marsh oiling occurred in salt marshes (*S. alterniflora*, *J. roemerianus*) in northern Barataria Bay, Louisiana. Persistent oiling conditions in these areas included heavily oiled vegetation mats (aboveground vegetation laid over by oiling, which died but remained rooted in place) and wrack lines that in many cases overlaid a thick layer of emulsified oil on the marsh substrate. As of fall of 2010, much of the oil layer averaged 2-3 cm in thickness and did not appear to have significantly weathered. Because of concerns that aggressive treatment might cause more harm than leaving the oil in place, a series of treatment tests were conducted in October and December 2010, using a random assignment of treatment methods to 28 plots that averaged 6 m in length and up to 15 m deep. There were two zones within each plot (Figure 4-2) as described by Zengel and Michel (2011):

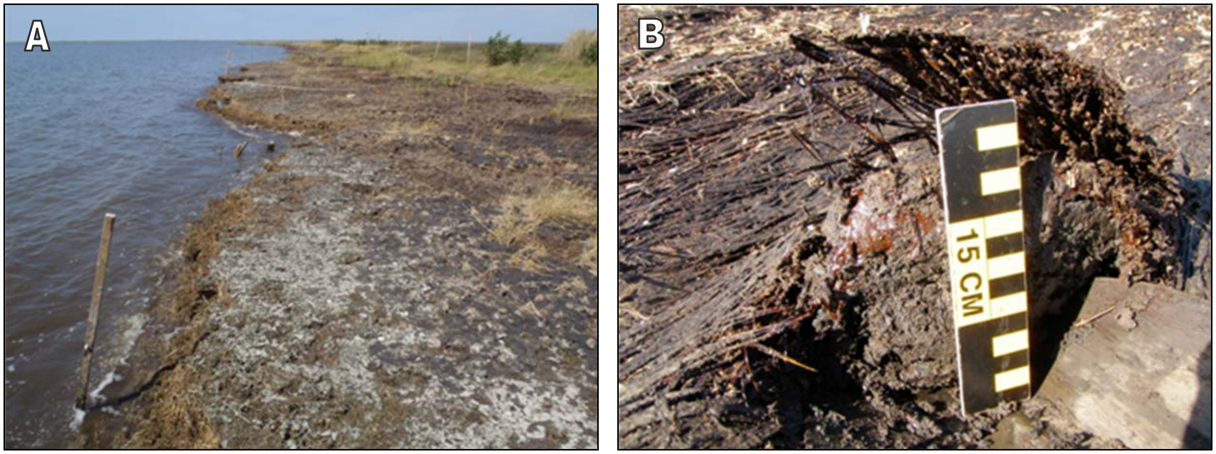

Figure 4-2. The two zones of heavy oiling along the marshes in N. Barataria Bay, Louisiana after the *Deepwater Horizon* oil spill. A: Zone A along the outer marsh edge, where the surface residue was hardened and crusty. B: Zone B was inland of Zone A and consisted of an oiled vegetation mat overlying a 2-3 cm thick mousse layer. Photo credit: Scott Zengel.

"Oiling Zone A" was a 1-3 m wide band on the lower marsh edge consisting of exposed surface oil residue with typically broken (51-90%) to continuous (91-100%) distribution and cover (≤1 cm) thickness. The oil residue had a hard, crusty to tarry surface layer and included the presence of thin algal mats and surface cracking. The aboveground vegetation in this zone had sloughed off leaving only short vegetation stubble. During the treatment tests, this oiling zone was not treated, because the oil appeared to be relatively weathered and due to concern that treatment could destabilize the seaward marsh edge and potentially lead to increased erosion.

"Oiling Zone B" was a 5-10 m wide band on the marsh platform extending from Oiling Zone A to the inland extent of oiling. Zone B included oil on both the vegetation and sediments. The vegetation oiling consisted of dead, laid over, rooted vegetation forming heavily oiled vegetation mats with a continuous oil coat (<0.1 cm thickness) of tarry consistency along the entire length of the plant stems, as well as heavily oiled wrack deposited at the high-water line. The sediment oiling consisted of continuous thick mousse (>1 cm) trapped under the oiled vegetation mats and wrack (Figure 4-3). Much of this mousse was 2-3 cm thick across the marsh platform, and was typically heaviest near the oiled wrack, to 5-8 cm thick. Subsurface oiling conditions were also observed, including burial of oiled vegetation mats or the underlying mousse layer by fine sediments or organic detritus. Instances of oiled crab burrows or oiled shoot/root channels were also observed. Oiling conditions in Zone B were the focus of the treatment testing and monitoring, and are emphasized below and in subsequent sections.

Monitoring of the plots post-treatment indicated that intensive raking and cutting were most effective at oil removal and did not cause excessive damage to the marsh soils. Based on the results, a shoreline treatment recommendation was written, directing the operational treatment of specific areas from mid-February to the end of September 2011. In all, over 11 km of the most heavily oiled marshes in northern Barataria Bay were treated by removing oiled wrack (including cutting the tarry wrack into sections for removal, where needed), raking to lift the oiled vegetation mat, cutting the oiled vegetation mat with a hedge trimmer for removal, additional raking and cutting where needed, scooping or scraping thick mousse layers from the marsh surface, and light raking and loose natural sorbent (bagasse) application as the workers backed their way out of the plots.

Both manual and mechanical methods were used. Manual treatments consisted of workers on walking boards using hand tools and power hedge trimmers (Figure 4-3) and were used throughout the cleanup. Power hedge trimmers were more effective than string trimmers or "weed whackers," and also may have been less damaging to the vegetation (allowing a straighter, cleaner cut) and safer for workers (no projectiles, no spraying of oil). Mechanical treatments were conducted from April to June 2011 and included barge-based and large airboat-based platforms positioned adjacent to marsh treatment areas that were equipped with long-reach hydraulic arms coupled with attachments including grapples, rakes, cutting devices, and "squeegees" to conduct marsh treatments (Figure 4-4). The "squeegee" devices were used to scrape thick mousse from the marsh surface after the heavily oiled wrack and vegetation mats were removed. Mechanical work was always followed by manual treatment.

Figure 4-3. Manual cutting and raking heavily oiled wrack removal in 2011 treatment of heavily oiled marshes in N. Barataria Bay, Louisiana, during the *Deepwater Horizon* oil spill. Photo credit: Scott Zengel.

Such heavy and persistent oiling may require intensive treatments, which can be effective as long as the allowed methods are well defined and there is close monitoring and guidance during operations, including periodic review and adaptation of methods that are causing too much damage. For the *Deepwater Horizon* spill, these methods were applied to only the most heavily oiled marshes (1% of the total length of oiled marshes). NOAA continues to monitor the treatment test areas, with initial results showing that manual cleanup treatments had a positive effect on oil conditions and vegetation regrowth. Tests of replanting immediately after cleanup treatment also seemed to be especially beneficial for vegetation recovery.

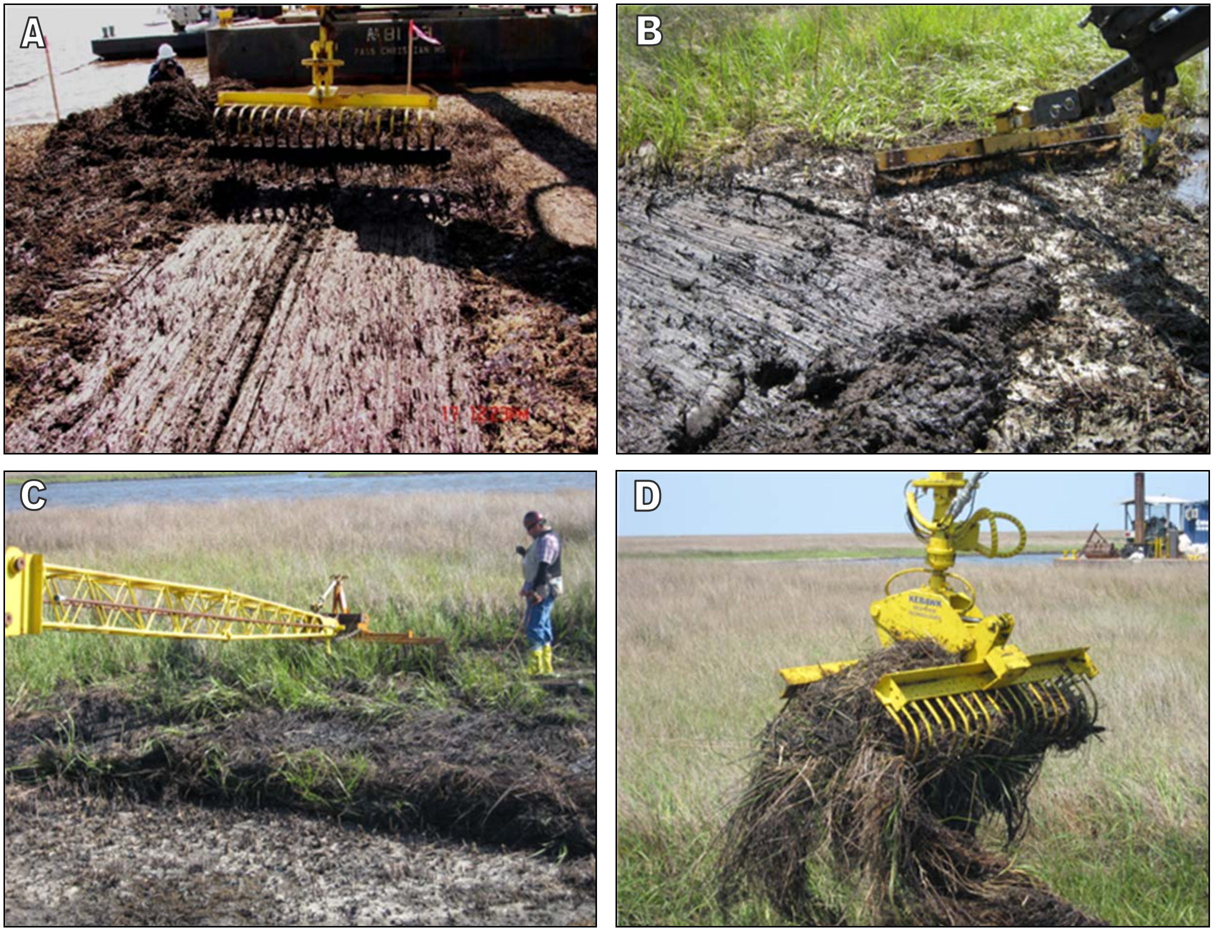

Figure 4-4. Mechanical treatment methods used to remove the thick oil and oiled vegetation mats on the marsh surface in N. Barataria Bay, Louisiana, during 2011. A: Raking of oil/mat on the outer platform would gouge the marsh soils if done too deeply. Photo credit: Jeffrey Leonick. B: Flat “squeegee” used to scrap the thick oil into piles for removal. C: Raking of the oiled wrack line had to be carefully guided to minimize removal of live vegetation. D: Grappling of the piles of oiled wrack was efficient and minimized foot traffic. Photo credit B-D: Jacqueline Michel.

For Further Reading

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Appendix A. Summary of the literature on impacts of light refined oils on marshes.

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Spills					
Florida barge, Buzzards Bay, MA Sanders et al. 1980; Teal et al. 1992; Peacock et al. 2005; Culbertson et al. 2007, 2008a,b	Sept 1969	No. 2 fuel oil/ 185,000 gal	Salt marsh/ *S. alterniflora, Salicornia virginica, S. patens* No cleanup in marshes	2 yr: Vegetation dead in heavily oiled areas; Alive in lightly oiled areas 7 yr: Fiddler crabs recovering in some areas; Not in areas with elevated naphthalenes 30 yr: Moderately weathered oil present at 8,000 ppm at depths of 12-16 cm 40 yr: Oil residues impacting fiddler crabs, ribbed mussels, and marsh vegetation	>40 yr
Bouchard 65 barge Buzzards Bay, MA Hampson and Moul 1978; Hampson 2000; Peacock et al. 2007	Oct 1974	No. 2 fuel oil/ 3,170,000 gal	Salt marsh/ *S. alterniflora, Salicornia virginica* No cleanup in marshes	3 yr: Complete mortality of vegetation and erosion rates 24x unoiled areas in heavily oiled marsh; Lightly oiled marsh showed lower biomass; Macroalgae disappeared, microalgal mat increased 17 yr: Vegetation slowly recovered; Eroded areas not recovered 30 yr: Weathered oil residues in surface sediments	>20 yr, more if consider marsh erosion
Exxon Bayway, Arthur Kill, NY Burger 1994; Bergen et al. 2000	Jan 1990	No. 2 fuel oil/ 567,000 gal	Salt marsh/ *S. alterniflora* No cleanup in marshes	0.5 yr: 7.6 ha of salt marsh killed; 2.8 ha recovering; Extensive fiddler crab and ribbed mussel mortality 3 yr: No recovery of most of the denuded areas, so replanted; Oil in sediments to 90 cm, at up to 55,000 ppm TPH 6-7 yr: Very little regrowth in unplanted area, no seedling survival; Planted areas mostly successful	>7 yr in unplanted areas

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Kinder Morgan Pipeline Spill, Suisun Marsh, CA	Apr 2004	Diesel/ 123,774 gal	Diked marsh *Salicornia virginica, Scirpus spp., Typha* Extensive removal of oiled soils/fertilized/ tilled	0.3 yr: Heavily oiled area near pipeline break was tilled/fertilized; Vegetation along the channels showed good recovery; Initial high mortality of biota in channels	1-4 yr except the tilled area
Field/Greenhouse Experiments					
North Greenland field oiling experiment Holt 1987	Aug 1982	Arctic diesel oil/ 10 L/m^2	Upland grassland, and three types of dwarf-shrub heath	3 yr: Dwarf-shrub heath showed no recovery; Graminoids showed almost no recovery except for *Carex bigelowii* which recovered moderately; Forbs showed only a few seedlings; Mosses showed good recovery in wet plots/no recovery in dry plots	>3 yr
Galveston Bay, TX Alexander and Webb 1985	Nov 1981; May 1983	No. 2 fuel oil/ 1 L/m^2 on soil, 1.5 L/m^2 on soil and lower plants, 2 L/m^2 on soil and entire plant	Salt marsh/ *S. alterniflora*	1 mo: 100% mortality at 2 L/m^2 and about 50% at 1.5 L/m^2 5 mo: vegetation at 1.5 and 2 L/m^2 had ~50-99% mortality 12 mo: 1.5 and 2 L/m^2 lower vegetation biomass	1 yr for soil and lower stem oiling; 2 yr for higher and entire plant oiling
Galveston Bay, TX Webb and Alexander 1991	Sept 1983	No. 2 fuel oil/ 1 L/m^2 on soil, 1.5 L/m^2 on sediment and lower plants, 2 L/m^2 on soil and entire plant	Salt marsh/ *S. alterniflora*	3 d: Chlorosis when applied to vegetation, not soil 9 mo: vegetation at 2 L/m^2 was mostly dead, regrowth from the edges of the plot; 12 mo: 2 L/m^2 treatment ~50% recovered, from rhizome growth from plants outside the plots; Other treatments slightly lower stem density than controls; No oil accumulation in soils	>1 yr; likely <2 yr

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Greenhouse experiment, LA Lin et al. 2002		No. 2 fuel oil/ pre-mixed with soil at 10 doses from 0 to 640 mg oil/g soil	*S. alterniflora* culms	3 mo: doses of No. 2 fuel oil as low as 29 mg/g significantly decreased belowground biomass; There was a strong dose-response relationship for biomass, stem height, stem density, evapotranspiration rate, and Microtox toxicity	N/A
Greenhouse experiment, LA Lin and Mendelssohn 2009		Weathered diesel mixed at 7 doses from 0 to 456 mg oil/g soil	*Juncus roemerianus* culms	1 yr: doses ≥160 mg/g reduced live stem density, ≥80 mg/g reduced stem height and above- and belowground biomass; Pots with plants had higher degradation of alkanes than those without plants	N/A
Jervis Bay, Australia Clarke and Ward 1994	N/A	Diesel/ 1 L/m^2	Salt marsh/ *Sarcocornia quinqueflora, Sporobolus virginicus*	1-12 mo: Near complete mortality and no growth of plants; New growth eliminated for up to one year; High mortality of littorina snails, with limited recovery after one year; Pulmonate snails recovered within one year	N/A

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Appendix B. Summary of selected light to medium crude oil spills and experiments in marshes.

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Spills					
T/V *Metula* Strait of Magellan, Chile Baker 1993; Owens et al. 1999	Aug 1974	Light Arabian crude and Bunker C fuel oil/ 16.2 million gal	Salt marsh/ *Salicornia ambigua, Suaeda argentinensis* Not cleaned	1.5 yr: Thick mousse up to 30 cm on the marsh surface; no cleanup conducted 18 yr: Thick oil remains (mean of 4.1 cm, range up to 8 cm); little sediment on top; oil still soft and fresh looking; Little plant recovery, mostly small *Salicornia* rooted below the oil 23 yr: Most marsh still bare in areas with 10-15 cm oil; areas with thin oil layer (<2.4 cm) starting to revegetate; very small plots tilled in 1993 showed higher recolonization	>30 yr
T/V *Amoco Cadiz* Brittany, France Vandermeulen et al. 1981; Baca et al. 1987; Gilfillan et al. 1995	March 1976	Arabian and Iranian light crude/ 70 million gal	Salt marsh/ Heavily cleaned (Also see case study in Chapter 4)	4 yr: Heavily oiled but untreated marsh recovered 7-8 yr: Heavily oiled untreated marsh recovered based on field data 14 yr: Heavily oiled treated marsh had less vegetated area and change in marsh community to low marsh because of excessive soil removal based on remote sensing data	4-8 yr untreated; >14 yr treated
T/V *Esso Bayway* Neches River, TX McCauley and Harrel 1981	Jan 1979	Light Arabian crude / 275,000 gal	Salt marsh/ *S. patens*/ Flushing/ Burning/Cutting plots	7 mo: Flushed plots showed best recovery; burned and clipped plots showed little/no recovery Note: All plots were flooded continuously by high water during the study.	<1 yr for flushing; >1 yr for burn/cut

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Pipeline spill, Galveston Bay, TX Alexander and Webb 1987	Jan 1984	Light crude/ 6,720 gal	Salt marsh/ *S. alterniflora* Mostly not cleaned, affected 6.4 km	<u>4-5 mo</u>: Heavily oiled sites had plant mortality, little regrowth; up to 100 mg/g TPH; light-moderately oiled sites showed little effects <u>7-8 mo</u>: Heavily oiled sites (10.5-18.3 mg/g TPH) had reduced densities of stems?; no oil visible in other sites <u>16 mo</u>: Bare areas had 1-51 mg/g TPH <u>32 mo</u>; Vegetation recovering but there were 2-3 m of erosion	>3 yr
Pipeline spill, Mississippi River, LA Hester and Mendelssohn 2000	Apr 1985	Louisiana crude/ 12,600 gal	Brackish marsh/ *S. patens* *S. alterniflora* *Distichlis spicata* 20 ha heavy oiling, treated	<u>1 yr</u>: High mortality in 20 ha impacted area <u>4 yr</u>: Nearly complete vegetative recovery, though some soil contamination still present	>4 yr
Fidalgo Bay, WA Hoff et al. 1993, Hoff 1995	Feb 1991	Prudhoe Bay crude/ 30,000 gal	Fringing salt marsh/ *Salicornia virginica, D. spicata*/ Flushing, vacuum	<u>16 mo</u>: Foot trampling was most detrimental to vegetation, washing with vacuum most effective and minimized impacts to vegetation	3-4 yr
Gulf War oil spill Arabian Gulf Barth 2002 Research Planning Inc 2003 Höpner and Al-Shaikh 2008	Jan-Mar 1991	Kuwait crude oil/ 520 million gal	Salt marsh/ *Halocnemon, Arthocnemon, Suaeda, Salicornia* No cleanup was conducted; extensive remediation conducted 2012-2014	<u>10 yr</u>: 25% of study sites showed no recovery at all; 20% fully recovered; 55% showing some recovery <u>16 yr</u>: Continued evidence of recovery, mostly by crabs re-occupation of tidal channels <u>22 yr</u>: Continued evidence of recovery, mostly in the lower marsh by annuals; very slow recovery of perennial vegetation in the upper marsh; remediation by re-activation or construction of new tidal channels speeding the rate of recovery	From 10 to >30 yr

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Three pipeline spills, Pass a Loutre, Mississippi Delta, LA Lin et al. 1999	Jan 1993; Oct 1993; Jan 1994	S. Louisiana crude/ 42-500 gal depending on site	Fresh water marsh/ *Phragmites australis* 500 gal: Intense cutting/flushing 210 gal: Light cleanup with sorbents 42 gal: No cleanup	<u>1-2 yr</u>: Intense cleanup site had very low soil TPH levels and full vegetation recovery; Light cleanup sites had elevated soil TPH and higher plant growth, indicating a stimulatory effect; No cleanup site (oil was contained within the boom for nearly 2 yr) had very elevated soil TPH and high plant mortality	<2 yr for cleaned area; >2 yr for no cleanup site
Deepwater Horizon LA Lin and Mendelssohn 2012	April-July 2010	Macondo-252 crude oil/ 4.9 million barrels	Salt marshes/ *S. alterniflora*; *J. roemerianus*	<u>7 mo</u>: Heavy oiling of vegetation and soils killed both *S. alterniflora* and *J. roemerianus*; Moderate oiling reduced above-ground biomass and stem density for *J. roemerianus* only	N/A
Field/Greenhouse Experiments					
Field/St. Louis Bay/MS De La Cruz et al. 1981	Late winter 1974	Empire Mix and Saudi Arabian crude 0.25-1.5 L/m^2 on marsh surface; and 1- 10 repeated oiling of 0.6 L/m^2	Irregularly flooded tidal marsh/ *J. roemerianus*	<u>3 mo</u>: High (up to 14 mg/g) oil uptake in aboveground tissues <u>6 mo</u>: oil in tissues decreased to 2.5-4 mg/g <u>9 mo</u>: oil in tissues to background <u>12 mo</u>: no oil in belowground tissues <u>1-7 mo</u>: reduced growth for all single oiling, with dose-response relationship; plant death for 1.5 L/m^2 and repeated oiling <u>3 yr</u>: all plants regardless of oiling fully recovered	1 yr for 0.5-2 L/m^2; 2 yr for 2.4 L/m^2; 3 yr for 3.6-6 L/m^2
Greenhouse/LA DeLaune et al. 1979	May 1976	S. Louisiana crude/ 4-32 L/m^2 maintain 5 cm water layer	Salt marsh/ *S. alterniflora*	<u>4 mo</u>: 4-8 L/m^2 reduced generation of new shoots because of persistent film on the water surface; at 16-32 L/m^2 no new shoots formed	N/A

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Field/ Louisiana DeLaune et al. 1979	May 1976	S. Louisiana crude/ 1-8 L/m^2 added to 0.25 m^2 circular plots	Salt marsh/ *S. alterniflora*	4 mo: No significant difference in above-ground biomass harvested at end of the first growing season 16 mo: No significant difference in above-ground biomass harvested at end of the second growing season Note: oil did not come in contact with leaves	>1 yr
Field/Galveston Bay, TX Alexander and Webb 1985	Nov 1981; May 1983	Arabian and Libyan crude: 1 L/m^2 on soil, 1.5 L/m^2 on sediment and lower plant, 2 L/m^2 on soil and entire plant	Salt marsh/ *S. alterniflora*	1 mo: Live biomass reduced for oiling of entire plant and soil for both seasons 5 mo: Live biomass reduced for oiling of entire plant and soil for May application, not November 12 mo: Live biomass reduced for oiling of entire plant and soil for May application, not November	>1 yr for growing season / highest oiling
Greenhouse/North Carolina Ferrell et al. 1984	N/A	Venezuela crude (API =24)/ 100% on plants, 32 L/m^2 on water, both on plant/on water	Salt marsh/ *S. alterniflora* transplants in sand and 2 parts and 1 part marsh soil	3 mo: 100% oil on plants increased mortality and decreased stem density, aerial dry weight, and regrowth; Regrowth completely inhibited for treatments with oil on the water; Better regrowth in sods with marsh soils vs. sand	N/A
		20 cm on plants, 32 L/m^2 on water, both on plant/on water	Brackish marsh/ *S. cynosuroides* transplants in sand	3 mo: 20 cm oil on plants had no effect mortality, stem density, aerial dry weight, and regrowth; Oil penetration into the soil caused ~50% mortality and reductions in stem density, aerial dry weight, regrowth, and root mass	N/A
Greenhouse/LA Lin and Mendelssohn 1996	Aug 1991	S. Louisiana crude/ Up to 24 L/m^2	Fresh marsh/ *Sagittaria lancifolia*	1 yr: Significant increase in biomass and stem density Note: oil did not come in contact with leaves, oil was mostly in the soil	0 yr

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Greenhouse/LA Lin and Mendelssohn 1996	Aug 1991	S. Louisiana crude/ >8 L/m^2	Salt/brackish marsh/ *S. alterniflora* *S. patens*	1 yr: No regrowth of biomass at levels of 8-24 L/m^2 Note: oil did not come in contact with leaves, oil was mostly in the soil; *S. patens* showed more short-term impacts compared to *S. alterniflora*	N/A
Greenhouse/LA Lin and Mendelssohn 2012	Nov 2010	Macondo-252 crude oil (weathered)/ 0-100% of shoot height oiled; 70% with repeated oiling every 4 d for 2 mo; 8 L/m^2 to soil surface	Salt marshes/ *S. alterniflora*; *J. roemerianus*	7 mo: For *S. alterniflora* effects persisted for the 70% repeated oiling and soil oiling only, even 100% oiling recovered to the level of the controls; For all metrics, *J. roemerianus* showed higher mortality at lower oiling exposures, starting at higher than 30% oiling	<1 yr for single dose to *Spartina* longer for *Juncus*
Greenhouse/ AL Anderson and Hess 2012	Jul 2011	S. Louisiana crude (fresh, weathered 3 d; 3 weeks)/ 6 L/m^2, 12 L/m^2, 24 L/m^2 to soils with simulated tidal flushing	*J. roemerianus*	2.5 mo: TPH in soils for the 3 loadings were 13.3 ± 1.6, 25.0 ± 3.1, and 48.0 ± 16.1 mg/g; live stem counts reduced to 5-25% of controls; photosynthesis rate = 50% of controls–no differences with degree of weathering; Roots died and did not regrow	N/A

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Appendix C. Summary of selected heavy fuel oil spills and experiments in marshes.

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Spills					
T/V *Arrow* Chedabucto Bay, Nova Scotia Thomas 1973, 1978 Gilfillan and Vandermeulen 1978	Feb 1970	No. 6 fuel oil/ 3 million gal	Salt marsh/ *S. alterniflora* The heavily oiled cove was not cleaned	<u>2 yr</u>: Extensive vegetation mortality, due in part to chronic re-oiling; heavy mortalities of soft-shell clams <u>6 yr</u>: Continued differences in biomass between oiled and control stations; soft-shell clams and periwinkles also affected	>6 yr
Mill River, CT Burk 1977	Jan 1972	Heavy fuel oil/ unknown volume	Freshwater ponds/ 23 species No information on cleanup methods	<u>0.5 yr</u>: Annual vegetation severely affected, with disappearance of 7 species and declines in 3 species post-spill <u>3 yr</u>: Annual species recovering, particularly in high marsh <u>4 yr</u>: High and mid marsh communities recovered; low marsh still showed low species richness and diversity	>4 yr
T/V *Golden Robin* Dalhousie, New Brunswick, Canada Vandermeulen and Jotcham 1986	Sept 1974	Bunker C/ 42,000 gal	High salt marsh/ *S. alterniflora* *S. patens* Various cleanup methods attempted	<u>0.75 yr</u>: First attempts to clean heavily oiled marsh, using manual removal of oiled vegetation, digging and spading of soils and vegetation, mechanical plowing, sod cutting, and burning. None were successful, and mechanical methods greatly disturbed the soils <u>2-3 yr</u>: Poor recovery of vegetation in all test plots; oil contamination to at least 10 cm and up to 20 cm; asphaltic layer 1-3 cm thick <u>3-10 yr</u>: Gradual vegetation recovery, most rapid for plots with manual treatment or burning <u>11 yr</u>: Most plots fully recovered vegetation; soils still contaminated; burial by clean sediment up to 15 cm	~ 10 yr

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
Barge *Nepco-140* St. Lawrence River, NY Alexander et al. 1981	June 1976	Bunker C/ 308,000 gal	Freshwater marsh/ *Typha* Intensive cleaning and cutting	1 yr: *Typha* growth where oiled and cut was 75 cm taller than where not cut, but had no flowers 2 yr: *Typha* growth and flowering were normal (note the water levels were low after cutting, so the cut stalks were always above water)	<2 yr
Bolivar Peninsula, TX Webb et al. 1981	Oct 1977	No. 6 fuel oil/ 42,000 gal	Salt marsh/ *S. alterniflora/* Several hectares/ Cleanup by raking and vegetation cutting	7 mo: Full recovery by the first growing season; total plant coverage caused death of the aboveground vegetation; when the upper 1/3 was not oiled, plants survived	<1 yr
T/V *Lang Fonn* Potomac River, MD Krebs and Tanner 1981	Dec 1978	No. 6 fuel oil up to 10 cm in a small cove/25,000 gal	Salt marsh/ *S. alterniflora/* Cleanup by raking and vegetation cutting	~2 yr: Vegetation mortality and no regrowth in soils with >16,000 ppm TPH, reduced growth at 5,000 ppm, and stimulation at <2,000 ppm; periwinkles and ribbed mussels much reduced	>2 yr
Barge *STC-101*, Chesapeake Bay, VA Hershener and Moore 1977	Feb 1976	No. 6 fuel oil 250,000 gal	Salt marsh/ *S. alterniflora/* Manual oil removal and vegetation cutting	3 mo: High mortality of periwinkles, slight mortality of ribbed mussels, no impact to oysters, new shoots shorter 7 mo: Periwinkles similar to controls, higher mortality of oyster spat in oiled marsh, and vegetation had higher stem density, shorter stems, and more flowering that showed an increase in net productivity	1 yr
Cape Fear River, NC Baca et al. 1983, 1985	April 1982	Heavy fuel oil 400,000 gal	Riverine brackish marsh/ *S. alterniflora, S. cynosuroides, Scirpus olneyi, Juncus effuses/* limited cutting	2 mo: 48 km of marsh shoreline was oiled; initial mortality of heavily oiled fringing vegetation; less mortality when only the lower parts of the plants were oiled 2 yr: All vegetation that was not cut was fully recovered and even increased in width; cut vegetation died with no re-growth	<2 yr

Spill Name/ Location/Citation	Oiling Date	Oil Type/ Volume Spilled	Habitat/Species/ Cleanup Method	Results by Years Post-spill	Years to Recovery
T/V *Julie N* Fore River, ME Michel et al. 1998	Sept 1996	IFO 380 and No. 2 fuel oil 170,000 gal	Salt marsh/ *S. alterniflora* *S. patens*/ 10.2 ha/ No active cleanup	1 yr: All plots had stem heights and density similar to unoiled controls, but there were 96 patches of dead vegetation, likely from exposure to the No. 2 fuel oil	1 yr except for the 96 patches
Lake Wabamun, Alberta, Canada Wernick et al. 2009	Aug 2005	Bunker C 39,340 gal	Freshwater lake, *Schoenoplectus tabernaemontani (= Scirpus validus)*/reed cutting, vacuum	2 yr: Post-spill transect length, total cover, and biomass were not significantly different between exposed and reference lake basins, except for a few areas with reduced biomass, likely due to treatment effects	<2 yr
M/V *Westwood*, Howe Sound, British Columbia, Canada Challenger et al. 2008	Aug 2006	IFO 380/ 7,630 gal	Salt marsh / *Eleocharis palustris,* *Carex lyngbyei*/ 4.2 ha/ Sediment removal, vegetation cutting	1 yr: Heavily oiled/untreated *Carex* had similar stem density/height and aboveground biomass to lightly oiled and unoiled controls; large reductions in these for sediment removal and trampling but not cutting only; For *Eleocharis*, in heavily oiled areas, areas that were flushed or cut showed positive effects; Very elevated TPH and PAH in trampled areas	N/A
Field/Greenhouse Experiments					
Georgia salt marsh Lee et al. 1981	Nov-Dec 1978	No. 5 fuel oil at 150 L over 4,000 m^2 (0.0375 L/m^2)	Salt marsh/ *S. alterniflora*	1.6 yr: High mortality of periwinkle snails; no change in populations of fiddler crab, oysters, or mussels; mud snails increased in density to scavenge on dead animals	>2 yr
Galveston Bay, TX Alexander and Webb 1985	Nov 1981; May 1983	No. 6 fuel oil: 1 L/m^2 on soil, 1.5 L/m^2 on sediment and lower plants, 2 L/m^2 on soil and entire plant	Salt marsh/ *S. alterniflora*	1 mo: Live biomass reduced by ~50% for oiling of entire plant only and May application, not November 5 mo: dead biomass higher for both treatment with oil on vegetation and May application, not November 12 mo: No differences for oiled plots for all seasons and treatments	1 yr

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Appendix D. Spills and experiments where *in situ* burning was conducted in marshes.

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Intracoastal City Well Blowout or McCormick Well Blowout/ Intracoastal City, LA Castle 2012	Nov 1975	S. Louisiana waxy crude (pour point of 80°F)/110,000 bbl spilled/estimated 30,000 bbl burned Minor waxy residue was observed locally	Brackish marsh/ *Spartina* spp.	~70 ha, including area oiled by rainout of the blowout plume, heavily coating the plant canopy	Wetlands had been burned annually by trappers, and were due for burning at the time of the blowout. Observations of a test burn conducted by the USCG showed new growth after 1 week. Survey in April 1976 showed significant re-growth in burn areas except where berms and other earthworks were constructed	1-2 yr
Harbor Island, TX Holt et al. 1978	Oct 1976	Crude oil/377 bbl though only a small amount was burned	Salt marsh/ *S. alterniflora,* black mangrove	0.1 ha heavily oiled, burned by err	<u>0.5 yr</u>: *S. alterniflora* biomass = 60% of unoiled/unburned controls; Lowest recovery was in area of standing water; 100% mortality of mangroves in burn area	N/A but likely <2 yr
ESSO Bayway, Port Neches, TX McCauley and Harrel 1981	Jan 1979	Light Arabian crude/6,545 bbl small marsh islands burned in cleanup experiment	Brackish marsh/ *S. patens*	Small marsh island, with 3 plots of 3 m^2; flooded	<u>0.6 yr</u>: Biomass in oiled/burned was 3% of unoiled/unburned controls; Burned/unoiled biomass was 1.5% of unoiled/unburned controls; Poor recovery due to persistent high water levels (3-55 cm) and low salinity (~ 0 ppt) post-treatments	N/A but likely <5 yr
Trans-Alaska Pipeline, Fairbanks, AK Buhite 1979	Feb 1978	Prudhoe Bay crude/16,000 bbl spilled, 500 bbl burned	Ponded tundra with water depth from a few cm to 1 m	0.8 ha burned on Day 63	<u>0.5 yr</u>: entire area was fertilized, with 50% plant regrowth during the first growing season	N/A but likely <5 yr

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Black Lake, West Hackberry, LA Overton et al. 1981	Sept 1978	Light Arabian crude/72,000 bbl spilled/most burned	Lacustrine and fringing marsh	N/A	Sediment samples collected at 1, 16, 29, and 53 weeks post-spill showed only background contamination. Foliage samples collected 1 and 16 weeks post-spill showed elevated PAHs from soot deposition several km from the site; At 29 weeks, foliage samples showed no contamination	N/A
Texaco Lafitte oil field Site 2, LA Mendelssohn et al. 1995	May 1983	S. Louisiana crude/ 282 bbl/some cleaned before burn	Brackish marsh/ *S. patens, D. spicata, S. alterniflora*	N/A	11 yr: No significant differences in soil TPH, live biomass, total biomass; Burned area higher species richness than unoiled control (7.6 vs 4.8), but not significant	N/A
Texaco Lafitte oil field Site 3, LA Mendelssohn et al. 1995	Sept 1986	S. Louisiana crude/ 4 bbl	Coastal brackish marsh/ *S. alterniflora, D. spicata*	N/A	8 yr: Soil TPH was 162 mg/g at the burn site vs 2 mg/g at the control site (may have been a more recent spill); No significant differences in live and total plant biomass and live-to-dead biomass; species richness in oiled/burned plots was 2.8 vs 6.6 in control plots; Overall recovery was ranked good	<8 yrs
Friendship II Pipeline, Kekcse, Hungary Nagy 1991	Jan 1988	Crude/ 2,657 bbl spilled/ 30 bbl burned	Peat and bog wetland (mostly sedges and reeds)	5.4 ha	1.5 yr: Sedge and reed vegetation recovered to near the original plant density	1.5 yr

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Imperial Oil, British Columbia, Canada Moir and Erskin 1994	June 1990	Canadian crude oil/ 840 bbl spilled/ majority burned	Freshwater wetland bog	2 ha burned on Day 2; bog was flooded	Day 5: new vegetation appeared; site was seeded and fertilized 0.75 yr: Vegetation was recovering and no oil was apparent on the site or stream	N/A
Pass a Loutre, Mississippi Delta, LA Mendelssohn et al. 1995	Aug 1990	S. Louisiana crude/several hundred bbl spilled/most burned	Freshwater marsh/ *Phragmites australis*	5.25 ha burned shortly after the spill	4 yr: Soil TPH was not different for oiled/burned vs 2 control sites; Live and total plant biomass and live:dead ratio were higher at the oiled/burned sites; overall recovery was ranked excellent	<4 yr
Chiltipin Creek, TX Gonzalez and Lugo 1995; Tunnell et al. 1995; Hyde et al. 1999	Jan 1992	S. Texas light crude/ 2,950 bbl spilled; 1,150 bbl burned; 80-85% burned Asphaltic, taffy-like residue covered the marsh surface and was manually removed	High marsh/ *D. spicata, Batis maritima. Borrichia frutescens*	6.5 ha burned on Day 4, 10 ha oiled; variable water levels	1.6 yr: high % cover but mostly by *D. spicata* 2.6 yr: Increase in species diversity, bare area declining; little change in TPH, but more weathered 3.6 yr: no change; apparent "steady state" 7 yr: increase in bare area, species diversity but affected by drought and damage from feral hogs and seismic survey	Predicted 14-15 yr based on trajectory for climax species
Texaco Lafitte oil field Site 1, LA Mendelssohn et al. 1995	June 1992	S. Louisiana crude /1 bbl	Brackish marsh/ *S. patens, D. spicata, J. roemerianus*	N/A	2.4 yr: No significant differences in soil TPH, live and total plant biomass, or species richness for oiled/burned and control plots, but there was a trend towards lower biomass in the oiled/burned plots; Burned plots had higher live-to-dead plant biomass; Overall recovery was ranked as moderate to good	~2.5 yr

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Meire Grove, MN Amoco Pipeline Zischke 1993; Mendelssohn et al. 1995	Sept 1992	Fuel oil and gasoline. 2,500 bbl spilled/unknown amount burned	Freshwater wetland pond/ *Typha* spp.	0.8 ha burned on Day 2 of discovery, but leaked for 10 days	Shortly after the burn: # invertebrate taxa/m^2 was 18 times higher at control vs oiled/burned pond 1 yr: considerable recovery in invertebrates 2 yr: Residual signs of trampling; Live plant biomass was 35 x higher and total plant biomass was 50 x higher in control pond vs oiled/burned pond; No differences in soil TPH; overall recovery was ranked poor	>2 yrs but likely <10 yr
Naval Air Station, Brunswick, ME Eufemia 1993; Metzger 1995	Mar 1993	JP-5 aviation fuel/ 1,512 bbl spilled/ 500 bbl burned No burn residue	Freshwater pond *T. latifolia, Sparganium americanum*	~1 ha burned on Day 8	0.4 yr: Studies of vegetation, fish, birds, mammals, benthic community, water quality, sediment quality oiled/burned vs control sites the following summer; No differences in plant cover or soil TPH; normal species abundance and distribution. Increase of *S. americanum* (burreed) over cattails, which was beneficial	<0.5 yr
Kolva River Basin Pipeline Spill Site 5, Komi, Russia Hartley 1996	1995	Crude oil/unknown volume because of multiple leaks from 1986-1994	Muskeg swamp with no outlet	6 ha burned	Burned violently for 20 hours, creating so much heat that the oil was driven deep into the peat mat; Burn residue on the surface was extremely viscous and oily, making further cleanup almost impossible	N/A but likely decades

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Rockefeller State Refuge, LA Hess et al. 1997; Pahl et al. 1997, 2003	Mar 1995	Condensate/40 bbl No burn residue	Brackish marsh/ *S. patens, D. spicata, S. alterniflora. Scripus robustus*	40 ha burned on Day 5; some water on marsh surface; Studied oiled, oiled/burned, and control transects	0.6 yr: burned transects: total cover 50% of other treatments; *S. patens* 14% of other treatments; *S. robustus* much higher (*D. spicata* slowed by post-burn flooding), thus stem density 30% of other treatments; Soil TPH decreased to background 2.6 yr: stem density, live biomass, total percent cover, and species composition of oiled/burned and oiled similar to control	3 yr
Refugio, TX Clark and Martin 1999	May 1997	Refugio Light and Giddings Stream crudes 90% burned Minor burn residue	Freshwater wetland/ *Borrichia frutescens, S. spartinae*	2.4 ha burned on Day 3	Observed new crayfish burrows shortly after the burn. Wetland was used for cattle grazing	N/A
Vermillion 16 Freshwater City, LA Henry 1997	July 1997	Condensate, API 50/ unknown amount spilled/most burned	Brackish marsh/ *Scirpus* spp, *S. patens, D. spicata*	3-4 ha burned on Day 13 after report; had been leaking 4 mo	During the burn, there was 5-10 cm of standing water in the thick vegetation 0.5 yr: very little vegetation re-growth–the site looked like an open pond. Plant death attributed to the 4 mo of exposure to the light crude.	N/A
Chevron Pipeline MP 68, Corrine, UT Williams et al. 2003 Michel et al. 2002	Jan 2000	Diesel/100 bbl 75-80% burned No burn residue	Freshwater wetlands, alkali flats, snow and ice cover	5.2 ha burned on 10 March, 1.3 ha on 27 April	Burn area = 1.3x intended area. Vegetation died in heavily oiled areas, burning not effective in removing oil penetrated into sediments or reduce toxic effects prior to burn; 4.1 ha fertilized and tilled in 2000/2001 to get PAH levels below criteria of 20 mg/kg	N/A. but likely <5 yr

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Louisiana Point, LA Michel et al. 2002	Feb 2000	Condensate/ unknown amount spilled or burned; No residue	High salt marsh/ *D. spicata, Borrichia frutescens, Batis maritime, S. patens*	5.3 ha oiled, 55 ha burned on Day 3 0.5-1 cm water over marsh during the burn	0.6 yr: In burned areas, total cover 64% and stem density 22% of control, *B. frutescens* and *D. spicata* much reduced. Stem density lower for all species 1.6 yr: total cover 76% and stem density 80% of control, with stem density of *B. frutescens* at 10%, *D. spicata* at 32%, and *Batis* at 120%	>1.6 yr, but likely <5 yr
Ruffy Brook, MN Michel et al. 2002	July 2000	Medium crude oil/>50 bbl 80% burned; tar-like residue ~1 cm thick, manually removed	Ponded freshwater wetland	1.2 ha burned on Day 1; 0.3-1 m of water in pond	1 yr: All herbaceous vegetation recovered; willows died (they are known to be sensitive to fire); No evidence residues sank	<1 yr
Mosquito Bay, LA Michel et al. 2002	April 2001	Condensate/ >1,000 bbl; No residue	Brackish marsh/ *S. patens, D. spicata, S. cynosuroides*	4.9 ha oiled, 40 ha burned on Days 7 and 8; 1-10 cm water layer on marsh	After the burn, oil in burrows still present 0.5 yr: burned/lightly and unoiled vegetation recovered with abundant fiddler crabs present, burned/heavily oiled areas along creek banks died, so did not reduce toxicity from contact with condensate prior to burn	<0.5 yr for lightly oiled areas; 1 yr for heavily oiled areas
Enbridge Pipeline, Cohasset, MN Leppälä 2004	July 2002	Canadian crude/ 6,000 bbl spilled, 3,000 bbl burned; significant residue that was thicker	Freshwater forested/ scrub-shrub with peat base	4.5 ha affected, 2.4 ha burned on Day 1, lasted 24 hours	Vegetation recovery was estimated to take many years because the deep excavation post-burn, as well as the burning of trees	Many years, likely >10 yr

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Chevron Texaco #2 Tank Battery, Sabine NWR, LA Entrix 2003	Aug 2002	S. Louisiana crude/ 150-300 bbl; pockets of oil and oil residues with nets and sorbent materials	Brackish marsh/ S. patens, *Typha latifolia*	1.4 ha burned on Day 4	0.7 yr: 80-90% cover in burn area, slight hydrocarbon odor in sediments; Mean 2,150 ppm TPH 1.2 yr: cattail 6 ft tall and seeds abundant, *S. patens* 3 ft tall; Mean 8 ppm TPH	1 yr
Chevron Pipeline MP 68, Corrine, UT Earthfax Engineering Inc 2003	Nov 2002	Gasoline No residue	Freshwater wetlands, alkali flats,	8.4 ha affected, 5.5 ha burned on Day 5	50% evaporated, 25-30% burned, rest in soils	N/A
Chevron Empire, LA Myers 2006; Merten et al. 2008; Baustian et al. 2010	Oct 2005	S. Louisiana crude/ 100-200 bbl; Some burn residue that was sticky and liquid (unburned) oil in burrows; removed with sorbents and natural flushing	Brackish marsh/ *S. patens* *Schoenplectus americana* (chairmaker's bulrush), *D. spicata*	11 ha burned on Days 44-45 after the initial release during Katrina; 0-10 cm water over the marsh	30 d: new vegetation 30-60 cm high 0.75 y: Plant biomass and species composition in oiled/burned returned to control levels; However, species richness remained somewhat lower in the oiled and burned areas compared to the reference areas; 1-1.5 yr: No differences between oiled/burned and control sites for sediment accretion, cellulose decomposition, and the rate of recovery from experimental disturbances (lethal and non-lethal removal of vegetation)	1 yr
Field/Greenhouse Experiments						
Field/Texas Kiesling et al. 1988	?	No. 2 fuel oil/crude; Field experiment of flushing, cutting, burning	Salt marsh/ *S. alterniflora*	1m^2 field plots	1 yr: Biomass did not differ among treatments for both oil types; Burning increased oil in sediment by 27-72%	1 yr

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Field/ Terrebonne Bay, LA Lindau et al. 1999	Aug 1995	S. Louisiana crude/ 2 L/m^2 Field experiment of oiled, oiled/burned, control	Salt marsh/ *S. alterniflora,*	2.4 m x 2.4 m plots, oiled stems and leaves	1 yr: no difference between oiled/burned, oiled, and control for plant density and biomass, carbon fixation; Stem height for burned plot was higher than others	1 yr
Field/ Terrebonne Bay, LA Lindau et al. 2003	Aug 1995	S. Louisiana crude/ 2 L/m^2 Field experiment of oiled, oiled/burned, control	Salt marsh/ *S. alterniflora,* Fresh marsh/ *Sag. lancifolia*	2.4 m x 2.4 m plots, oiled stems and leaves	0.25 yr: 83% reductions in carbon fixation, live stem density and plant height for oiled and oiled/burned vs. control; 1 yr: all oiled/burned plots had 100+% recovery compared to controls; Oiled plots were at 62% of controls	1 yr
Greenhouse experiment/LA Smith and Proffitt 1999	April 1997	Venezuela crude, 0, 4, 8, 16, and 24 L/m^2 to the sediment surface	Three clones of *S. alterniflora*	Laboratory pots, oiled/ burned, oiled for 5 oil loadings; n=3 Water level at the sediment surface	0.5 yr: oiled/burned had increased survival relative to oiled-only groups in all except the highest two oil dosages; At 16 L/m^2 oiled/burned, survival was slightly reduced; at 24 L/m^2, survival was 10-50%; New shoots died with >1 cm oil on the surface; For biomass, oiled/burned was higher than oiled for loadings of 4-16 L/m^2 oil, but all significantly decreased	N/A

Spill Name/ Location/ Citation	Burn Date	Oil Type/Volume Spilled/Burned	Habitat/ Species	Burn Area	Results by Years Post-burn	Years to Recovery
Burn-tank experiments/ LA Mendelssohn et al. 2001; Lin et al. 2002	Aug 1999	Diesel 1.5 L/m^2	*S. alterniflora*	Laboratory pots, water depths 10, 2, 0, -10 cm (n= 5), burn duration 400 and 1400 s	<u>0.6 yr</u>: 10 cm water over the soil surface kept temperatures <37°C with high plant survival and regrowth; with 0 and 2 cm water, the soil temperatures were low, but diesel still killed the plants; water at 10 cm below the soil surface resulted in high soil temperature (120°C at 2 cm depth) and almost complete mortality; No plants survived at temperature >60°C at 2 cm soil depth; Burning did not remove oil that had penetrated into the soil	N/A
Burn-tank experiments/ LA Lin et al. 2005; Bryner et al. 2003	Aug 2000	S. Louisiana crude and diesel 0.5 L/m^2 added to the soil before the burn (this dosage will not severely affect the plants but is high enough to analyze effectiveness of burning in removing oil from the soil)	*S. alterniflora S. patens/D. spicata, Sag. lancifolia*	Laboratory pots, water depths 10, 2, -2 cm (n= 5), burn duration 700 s	<u>1 yr</u>: 10 and 2 cm water over the soil surface kept temperatures at <40 and <50°C, respectively, with high plant survival and regrowth; Water at 2 cm below the soil surface resulted in temperature 80-100°C at 0.5 cm depth; *S. patens* and *D. spicata* survived 2 cm of soil exposure (dense stems, deeper rhizomes, and rapid regrowth), whereas *S alterniflora* (30% reduced survival) and *Sag. lancifolia* (50% reduction in survival) because its rhizomes are shallow); Burning did not remove the crude oil added to the soil before the burn; Burning did remove more of the diesel	N/A

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September 2013

U.S. DEPARTMENT OF COMMERCE
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Acting Under Secretary of Commerce for Oceans and Atmosphere, and NOAA Administrator

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OIL SPILLS AND TOURISM: THEY DON'T MIX

The Deepwater Horizon Oil Spill Disaster in 2010 marked the worst environmental tragedy in our nation's history, with effects that will last years, if not decades. Despite claims made by oil companies that offshore drilling is safer than ever before, the fact remains that the technologies surrounding offshore drilling are relatively unchanged, and the risk of a major spill is still a very real possibility.

The effects of the BP oil spill on tourism, particularly on the Gulf Coast of Florida, serve as a case study to demonstrate how an oil spill along the Atlantic coast might affect tourism in nearby states, especially in areas that were not directly tainted by spilled oil. Much can be learned from academic journal articles, interview data related to trends in the hotel industry, and Google Trends, which tracks how often a search term is entered relative to the total search volume in a region.

Tourism significantly declines after an oil spill:

In Louisiana, leisure visitors spent much less money following the oil spill.

- Leisure visitor spending in 2010 dropped **by $247 million**, though this loss was somewhat offset by an increase in spending from the oil spill cleanup.[1]
- Leisure visitor spending was **projected to lose $422 million from 2010 through 2013**.[2]

As the spill spread along the coast, so did **hotel cancellations throughout the Gulf Coast**.[3]

- 2 weeks after the spill, 35% of hotels surveyed had cancellations.
- Cancellation percentages rose 4 weeks after the spill (44%) and 6 weeks after the spill (60%).

Hotels in Gulf Coast states had **difficulty booking future events**.[4]

- 2 weeks after the spill, 42% had difficulty booking future events.
- The difficulty in booking future events doubled over the next two surveys (4 and 6 weeks).

There was a **marked decline in Gulf Coast tourism interest** following the oil spill.[5]

- Tripadvisor.com reports that in the 20 day period after the spill, consumers searched 52% less for Pensacola, FL., 65% less for Gulf Shores, AL., and 48% less for Destin, FL.
- 26% of those who had plans to visit Louisiana postponed or canceled.
- Texas, Mississippi, and Florida found that 17% had postponed or canceled their planned vacation.

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[1] The Impact of The BP Oil Spill on Visitor Spending in Louisiana: Revised estimates based on data through 2010 Q4 , *Tourism Economics, prepared for the Louisiana Office of Tourism (June 2011)*
[2] Id.
[3] The Gulf Coast Oil Spill and the Hospitality Industry , *The Knowland Group (August 16, 2010)*
[4] Id.
[5] Potential Impact of the Gulf Spill on Tourism , *Oxford Economics, prepared for the U.S. Travel Association (July 22, 2010)*

Events and room bookings were canceled throughout the summer. However, those travelers did not stay home - **they took their vacations elsewhere**.

- Substitute destinations, such as North Carolina, Massachusetts, and Maine, were chosen in place of Gulf destinations, indicating a high aversion to the Gulf region.[6]
- Tourism visits declined by 4% in cities along the Gulf Coast region of Florida, while tourism visits increased by 5% in cities along the Atlantic coast.[7]
- The length of time that tourists avoid an area after an environmental disaster like an oil spill depends on a variety of factors, but can be months and even years.[8]

Regional tourism declined, even in areas that did not experience oil pollution:

As soon as the oil spilled, hoteliers began to experience booking cancelations, **even in areas without oiled beaches.**[9]

- 25% of surveyed travelers thought that the Florida Keys and Panama City Beach were areas affected by the spill, even though oil had not washed ashore at these locations.
- Even if only one-fifth of those people change their vacation plans, **a 5 percent decrease in annual visitors** would cost Florida **$3 billion in lost visitor spending, $182.5 million in lost sales tax collections and 48,000 lost jobs.**[10]

When asked where oil was present on the shores of Florida's beaches, readers **incorrectly believed that there was oil on West Coast Florida beaches** from St. Petersburg to the Florida Keys (16%), South Florida from Miami to Palm Beach (8%), and all the way up the East Coast from Daytona to Amelia Island (5-6%).[11]

The result of the misperception of oil on Florida's beaches was that many leisure **travelers were less likely to visit Florida in the months following the spill.**

The risks to the hospitality and recreation industries on the East Coast are too high. This may help to explain why more than 110 **communities oppose offshore drilling**, and the opposition continues to grow and form a united front to protect our oceans. For more information, please contact act@oceana.org.

TripAdvisor Page Views
% Change in Share of U.S.

Twenty days until...	20-May	20-Jun	18-Jul
West Palm Beach	14%	17%	9%
Daytona Beach	1%	-4%	3%
Hilton Head	-4%	0%	1%
Miami	16%	1%	-2%
Myrtle Beach	1%	3%	-2%
Outer Banks	-11%	-8%	-4%
Fort Lauderdale	5%	-1%	-5%
Biloxi	-24%	-16%	-14%
Key Largo	-24%	-28%	-14%
Clearwater	-20%	-26%	-17%
Fort Myers Beach	-20%	-31%	-29%
Panama City Beach	-18%	-31%	-30%
Destin	-9%	-25%	-48%
Pensacola	-41%	-52%	-52%
Gulf Shores	-19%	-47%	-65%

Source: TripAdvisor

The table above shows the percentage drop in the share of TripAdvisor U.S. page views for various destinations in the Atlantic/Gulf regions after the BP oil spill. Data shows that searches for travel take place on Google typically a few weeks to about a month before the actual travel.

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[6] Id.
[7] Id.
[8] Id.
[9] The Gulf Coast Oil Spill and the Hospitality Industry , *The Knowland Group (August 16, 2010)*
[10] Id.
[11] Response: Deepwater Horizon Update, VISIT FLORDIA (*August 2010*)

11

Ecotoxicological Studies Focusing on Marine and Freshwater Fish

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11.1 INTRODUCTION

PAHs originate from both point and diffuse sources (see Chapters 1, 2 and 3). Studies from the 1960s onwards began to report that many coastal harbors, as well as inland rivers and lakes, were contaminated with varying levels of PAHs and more attention began to be placed on assessing their ecotoxicological potential. This assessment included laboratory studies with individual and complex mixtures of PAHs, as well as field studies on feral organisms. Important observations were made on tumors and other pathologies in fish from contaminated environments, with effects being strongly linked in some instances to PAHs. However, in field studies there is invariably the problem of other confounding contaminants, such as organochlorine pesticides and metals, and laboratory studies remain critical for providing insight into the relative importance of various risk factors that may be associated with field observations. Also, unlike many organochlorine compounds and metals that have potential to bioconcentrate in fish tissues, PAHs are rapidly metabolized (e.g. Lemaire *et al.* 1990; see also Chapter 5) and do not bioconcentrate to any substantive extent. Accordingly, there is always the possibility that they are causing adverse effects yet providing little or no chemical signature — the so-called 'hit and run' potential for PAHs.

This overview draws upon information from field and laboratory studies over the past 15 years or so linking various biological effects in fin-fish to PAHs. It

PAHs: An Ecotoxicological Perspective. Edited by Peter E.T. Douben.

includes information on biochemical, histopathological, genetic, immunological, reproductive, developmental and behavioral effects, which are grouped in relation to field studies (Table 11.1) as well as laboratory studies with complex mixtures of PAHs (Table 11.2) and individual PAHs (Table 11.3). The studies are selective but considered to be fairly representative for each category. Establishing either a strict scientific or 'legal' standard of causal evidence for environmental effects seems improbable. However, considering the combination of field and laboratory studies presently available and using a weight of evidence approach, we suggest that levels of PAHs commonly found in many marine and freshwater environments are important risk factors for various aspects of fish health.

With respect to introductory caution, it should be noted that both weathered petroleum and pyrolytic sources of PAHs are found in association with a variety of N, S and O substituted analogs, phenols, etc., and these compounds, as well as PAHs may be contributing to overall toxicity, even though the term 'PAHs' is commonly used. Parental PAH alone represents a large class of compounds and various studies often measure different numbers of compounds in reporting 'total' PAH or estimate total PAH by fluorescence, making it difficult to make interstudy comparisons between exposures and biological effects. Furthermore, many studies do not measure alkylated forms of PAHs, and this could underestimate toxicity potential at sites primarily contaminated with petroleum hydrocarbons, which are enriched in some alkylated PAHs (e.g. Neff and Anderson 1981).

Due to space limitations, common fish names are used in the tables. Common names can also be more meaningful for a general reader, with the understanding that a specialist can refer to the references.

11.2 FIELD AND LABORATORY STUDIES

11.2.1 BIOCHEMICAL EFFECTS

Biochemical effects have been observed in fish in coastal waters, lakes and rivers in a number of countries. Most observations have been on alteration of phase I and to a much lesser extent phase II enzymes, which play a key role in detoxification and other biochemical processes (e.g. Stegeman and Hahn 1993). Other biochemical effects have occasionally been reported (e.g. changes in hormones, energy reserves and serum enzymes) but most observations in the environment and the major proportion of those noted in Table 11.1 have been on the induction of mixed-function oxygenase (MFO) enzymes, which belong to the phase I group of enzymes. The MFO family of enzymes have iron-containing hemoproteins, cytochrome(s)-P450, as terminal oxidases and terms like induction of MFO enzymes or cytochrome(s)-P450 are often used synonymously. MFO enzymes have been the subject of considerable attention,

TABLE 11.1 Examples of effects observed in feral fish which may be variously linked to PAHs

Species	Effects						Locations	References
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development		
American plaice	X						Canada, Baie des Anglais	Lee *et al*. (1999)
Atlantic croaker	X			X			USA, Galveston Bay	Willet *et al*. (1997)
Atlantic tomcod				X			USA, urban and industrialized areas	Stein *et al*. (1994)
Atlantic tomcod	X						Canada, Miramichi Estuary	Courtenay *et al*. (1995)
Barbel	X						Italy, River Po	Vigano *et al*. (1998)
Brown bullhead		X					USA, Black River	Baumann and Harshbarger (1985)
Brown bullhead				X			USA/Canada, different lakes	Pandrangi *et al*. (1995)
Brown bullhead					X	X	USA, Black and Cuyahoga Rivers	Lesko *et al*. (1996)
Brown bullhead	X						USA, Black River	Steevens *et al*. (1996)
Brown bullhead	X	X					USA, Niagara River	Eufemia *et al*. (1997)
Brown bullhead	X						France, Wetland area of Camargue	Buet *et al*. (1998)
Brown bullhead		X				X	USA, Schuylkill River	Steyermark *et al*. (1999)
Brown bullhead	X	X				X	USA, Lower Great Lakes	Arcand-Hoy and Metcalfe (1999)
Brown bullhead	X			X		X	USA, Black River	McFarland *et al*. (1999)
Carp				X			USA/Canada, lakes	Pandrangi *et al*. (1995)
Carp		X					USA, West Point Lake	Pritchard *et al*. (1996)
Carp	X						Czech Republic, ponds	Machala *et al*. (1997)
Carp	X						France, Wetland area of Camargue	Buet *et al*. (1998)
Carp	X						USA, Ottawa River	Cormier *et al*. (2000)
Carp bream	X					X	Russia, Rybinsk reservoir	German and Kozlovskaya (2001)
Cat fish	X			X			USA, Devil's Swamp	Winston *et al*. (1988)
Cat fish		X					USA, Black River	Baumann and Harshbarger (1995)
Cat fish	X			X			USA, Galveston Bay	Willet *et al*. (1997)
Chinook salmon	X			X			USA, Puget Sound	Stein *et al*. (1995)
Chinook salmon			X				USA, Puget Sound	Arkoosh *et al*. (1998)
Chinook salmon	X						USA, Commencement Bay	Stehr *et al*. (2000)
Chub	X						Italy, River Po	Vigano *et al*. (1998)
Cod	X						Norway, Soerfjorden	Goksoyr *et al*. (1994)
Cod	X						Norway, Soerfjorden	Beyer *et al*. (1996)
Cod		X	X				Norway, Soerfjorden	Husøy *et al*. (1996)
Cod	X						Norway, Sunndalsfjord	Naes *et al*. (1999)

(*continued overleaf*)

TABLE 11.1 (*continued*)

Species	Effects						Locations	References
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development		
Crayfish				X			The Netherlands, River Meuse	Schilderman *et al*. (1999)
Dab	X						Germany, German Bight	Westernhagen *et al*. (1999)
Dab	X			X			UK, coastal waters	Lyons *et al*. (2000)
Deep sea fish	X						Mediterranean Sea	Escartin and Porte (1999)
Eel				X			The Netherlands, freshwater sites	Van Shooten *et al*. (1995)
Eel	X			X			The Netherlands, freshwater sites	Van der Oost *et al*. (1996)
Eel	X						France, Wetland area of Camargue	Buet *et al*. (1998)
Eel	X						UK, Thames estuary	Livingstone *et al*. (2000)
Eel		X					Canada, St. Lawrence River	Couillard *et al*. (1997)
Emerald rockcod	X						Antarctica, bays	Miller *et al*. (1999)
English sole		X					USA, Puget Sound	Malins *et al*. (1984)
English sole		X					USA, Eagle Harbor	Malins *et al*. (1985)
English sole		X					Canada, Vancouver Harbor	Goyette *et al*. (1988)
English sole					X	X	USA, Puget Sound	Casillas *et al*. (1991)
English sole	X						USA, different estuaries	Collier *et al*. (1992a)
English sole					X	X	USA, Puget Sound	Collier *et al*. (1992b)
English sole					X	X	USA, Puget Sound	Johnson *et al*. (1997)
English sole		X					USA, Pacific and Atlantic coasts	Myers *et al*. (1998)
European flounder		X					Elbe River	Kohler (1990)
European flounder	X						Norway, Soerfjorden	Beyer *et al*. (1996)
European flounder		X	X				Norway, Soerfjorden	Husøy *et al*. (1996)
European flounder		X					The Netherlands, coastal	Vethaak and Wester (1996)
European flounder	X	X					Estonia	Bogovski *et al*. (1997)
European flounder	X						Germany, German Bight	Westernhagen *et al*. (1999)
European perch	X						Canada, St. Lawrence River	Hontela *et al*. (1992)
Fish (various)				X			USA, Buffalo and Detroit Rivers	Maccubbin *et al*. (1990)
Flounder	X						Chile, polluted bays	Rudolph and Rudolph (1999)
Flounder		X					Chile, polluted bays	George-Nascimento *et al*. (2000)
Grenadier	X	X					Norway, coastal sites	Forlin *et al*. (1996)

TABLE 11.1 (*continued*)

Species	Effects						Locations	References
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development		
Herring				X			USA, *Exxon Valdez* oil spill	Hose and Brown (1998)
Med. sea perch	X						France, Mediterranean Sea	Lafaurie *et al*. (1993)
Med. sea perch	X			X			North-west Mediterranean Sea	Burgeot *et al*. (1996)
Milkfish	X						Taiwan, river	Chen *et al*. (1998)
Mullet				X			South-eastern Black Sea	Karakoc *et al*. (1998)
Mummichog		X					USA, Elizabeth River	Volgelbein *et al*. (1990)
Mummichog				X			USA, Elizabeth River	Rose *et al*. (2000)
Nase	X						Italy, River Po	Vigano *et al*. (1998)
Oyster toadfish			X				USA, Elizabeth River	Collier *et al*. (1993)
Perch	X			X			Sweden, Baltic Sea coastline	Ericson *et al*. (1998)
Pike	X						Canada, St. Lawrence River	Hontela *et al*. (1992)
Pike	X						The Netherlands, small Amsterdam lakes	Van der Oost and Heida (1993)
Pikey bream	X						Australia, creeks	Cavanagh *et al*. (2000)
Plaice		X					France, *Amoco Cadiz* spill	Haensly *et al*. (1982)
Rainbow trout	X						Italy, River Po	Vigano *et al*. (1994)
Rainbow trout	X	X					Estonia, River Narva	Tuvikene *et al*. (1999)
Red mullet	X			X			North-west Mediterranean Sea	Burgeot *et al*. (1996)
Red mullet	X						France, Mediterranean Sea	Mathieu *et al*. (1991)
Roach	X						The Netherlands, Amsterdam lakes	Van der Oost and Heida (1993)
Roach	X	X					Estonia, River Narva	Tuvikene *et al*. (1999)
Rock sole						X	USA, Puget Sound	Johnson *et al*. (1998)
Salmon chum	X						USA, Commencement Bay	Stehr *et al*. (2000)
Sardine	X						Spain, north coast	Peters *et al*. (1994)
Sculpin	X			X			Iceland, harbors	Stephensen *et al*. (2000)
Smooth flounder	X						Canada, Miramichi Estuary	Courtenay *et al*. (1995)
Spot			X				USA, Elizabeth River	Weeks and Warinner (1984)
Spot	X						USA, Elizabeth River	Van Veld *et al*. (1990)
Spot			X				USA, Elizabeth River	Faisal *et al*. (1993)

(*continued overleaf*)

TABLE 11.1 (*continued*)

Species	Effects						Locations	References
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development		
Starry flounder	X						USA, different estuaries	Collier *et al*. (1992a)
Starry flounder		X					USA, San Francisco Bay	Stehr *et al*. (1997)
Starry flounder		X					USA, Pacific and Atlantic coasts	Myers *et al*. (1998)
Surf perch	X	X					USA, California petroleum seep	Spies *et al*. (1996)
Tilapia				X			Taiwan, Damsui River	Liu *et al*. (1991)
White croaker		X					USA, Pacific and Atlantic coasts	Myers *et al*. (1998)
White sucker		X					Canada, Lake Ontario	Cairns and Fitzsimons (1988)
White sucker	X	X	X				USA, Sheboygan River	Schrank *et al*. (1997)
White sucker						X	USA/Canada, St. Lawrence River	Ridgway *et al*. (1999)
White sucker	X						USA, Ottawa River	Cormier *et al*. (2000)
Winter flounder		X					USA, coastal sites in New England	Gardner *et al*. (1989)
Winter flounder	X	X				X	USA, north-east coast	Johnson *et al*. (1992)
Winter flounder		X					USA, Jamaica Bay	Augspurger *et al*. (1994)
Winter flounder		X					USA, Boston Harbor	Moore and Stegeman (1994)
Winter flounder				X			USA, urbanized areas	Stein *et al*. (1994)
Winter flounder	X						Canada, Sydney estuary	Vandermeulen *et al*. (1996)
Winter flounder	X	X	X				Canada, St. John's Harbor	French *et al*. (2000)
Yellow perch				X			USA/Canada, different lakes	Pandrangi *et al*. (1995)
Yellow perch						X	Canada, St. Lawrence River	Hontela *et al*. (1995)
Yellow perch	X	X					Estonia, River Narva	Tuvikene *et al*. (1999)

since they are commonly induced in fish and other animals upon exposure to a variety of molecules of environmental importance, including PAHs. The sensitivity of fish to induction by petroleum sources of PAHs, including in small boat harbors, was noted over 25 years ago (Payne 1976) and induction has been widely observed since then in fish from marine and freshwater environments (e.g. Payne *et al*. 1987; Stegeman and Hahn 1993).

TABLE 11.2 Examples of effects observed in fish upon exposure to complex mixtures containing PAHs

Species	Effects							Contamination	References
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development	Behavioral		
American plaice					X	X		Sediments, Baie des Anglais (CAN)	Nagler and Cyr (1997)
Blenny	X							North sea crude oil	Celander *et al*. (1994)
Bullhead	X							Sediments, Hamilton Harbor (CAN)	Leadly *et al*. (1999)
Catfish	X			X				Sediments, Black Harbor (USA)	Di Giulio *et al*. (1993)
Cod	X			X				Mechanically dispersed crude oil	Aas *et al*. (2000)
Dab	X							Sediments, Liverpool Harbor (UK)	Livingstone *et al*. (1993)
Eel	X							Po River water (Italy)	Agradi *et al*. (2000)
Eel	X			X				Petroleum distillate extract	Pacheco and Santos (2001)
English sole		X						Sediment extract, Eagle Harbor (USA)	Schiewe *et al*. (1991)
English sole				X				Sediments, Eagle Harbor (USA)	French *et al*. (1996)
Flounder	X							Sediments, Rotterdam Harbor (The Netherlands)	Eggens *et al*. (1996)
Goby							X	Diesel contaminated sediment	Gregg *et al*. (1997)
Herring					X			*Exxon Valdez* oil	Pearson *et al*. (1995)
Herring	X							Water fractions of crude oil	Thomas *et al*. (1997)
Herring		X		X		X	X	Weathered Alaska crude oil	Carls *et al*. (1999)
Herring					X	X		Weathered creosote-treated pilings	Vines *et al*. (2000)
Medaka		X						Sediments, Great Lakes	Fabacher *et al*. (1991)
Medaka						X		Outboard motor emission waters	Koehler and Hardy (1999)
Pacific halibut						X		Oil-laden sediments	Moles and Norcross (1998)
Pink salmon						X		Weathered *Exxon Valdez* crude oil	Heintz *et al*. (1999)
Pink salmon				X				*Exxon Valdez* oil	Roy *et al*. (1999)

(*continued overleaf*)

TABLE 11.2 (*continued*)

Species	Effects							Contamination	References
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development	Behavioral		
Pink salmon							X	Weathered *Exxon Valdez* crude oil	Wertheimer *et al*. (2000)
Rainbow trout						X		Effluent from a petroleum refinery	Rowe *et al*. (1983)
Rainbow trout				X				Sediment, Lake Ontario	Metcalfe *et al*. (1990)
Rainbow trout								Sediments, Hamilton Harbor (CAN)	Balch *et al*. (1995)
Rainbow trout	X							Sediments, Skagerrak, Kattegat (Norway)	Magnusson *et al*. (1996)
Rainbow trout	X					X		Crankcase oil	Hellou *et al*. (1997)
Rainbow trout	X							Creosote contaminated sediment extracts	Hyoetylaeinen and Oikari (1999)
Rainbow trout			X					Creosote microcosms	Karrow *et al*. (1999)
Rainbow trout	X							Santa Cruz River water	Petty *et al*. (2000)
Silverside minnow					X			Combusted crude oil fraction	Al-Yakoob *et al*. (1996)
Spot							X	Sediments, creek, South Carolina (USA)	Marshall and Coull (1996)
Spot	X	X						Creosote contaminated sediment	Sved *et al*. (1997)
Spot							X	Sediment contaminated by produced water	Hinkle-Conn *et al*. (1998)
Sturgeon					X			Weathered coal tar sediments	Kocan *et al*. (1996)
Surf smelt						X		Sediments, Puget Sound (USA)	Misitano *et al*. (1994)
Tilapia	X							Sediments, Damsui River (Taiwan)	Ueng *et al*. (1995)
Tilapia	X							Sediments, Hong Kong	Wong *et al*. (2001)
Winter flounder	X					X		Petroleum-contaminated sediments	Payne *et al*. (1988)
Winter flounder			X					Petroleum-contaminated sediments	Payne and Fancey (1989)
Winter flounder		X						Petroleum-contaminated sediments	Khan (1995)
Yellow perch	X	X						Oil sands mining-associated waters	Van den Heuvel *et al*. (2000)

TABLE 11.3 Examples of effects observed in fish upon exposure to individual PAHs

Species	Effects							Exposure	Reference
	Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development	Behavioral		
Antarctic rockcod	X							B[a]P*	McDonald *et al*. (1995)
Arctic charr	X							B[a]P	Wolkers *et al*. (1996)
Bluegill	X		X					Anthracene	McCloskey and Oris (1993)
Bluegill	X							Anthracene	Choi and Oris (2000)
Brook trout	X			X				B[a]P	Padros *et al*. (2000)
Brown bullhead	X			X				B[a]P	Ploch *et al*. (1998)
Brown trout				X				B[a]P	Mitchelmore *et al*. (1998)
Carp	X							B[a]P, chrysene	Van der Weiden *et al*. (1994)
Carp	X							MC*	Marionnet *et al*. (1998)
Catfish								B[a]P, DMBA*	Martin-Alguacil *et al*. (1991)
Catfish	X			X				B[a]P, BNF*	Ploch *et al*. (1998)
Catfish	X							B[a]P	Willett *et al*. (2000)
Catfish					X			Acenaphthene	Dwivedi (2000)
Catfish	X							BNF, DMBA	Weber and Janz (2001)
Chinook salmon			X					B[a]P	Arkoosh *et al*. (1998)
Chinook salmon	X							BNF	Campbell and Devlin (1996)
Eel	X			X				B[a]P, BNF	Pacheco and Santos (1997)
Eel	X							B[a]P, BNF	Schlezinger and Stegeman (2000)
European flounder	X							B[a]P, phenanthrene	Rocha Monteiro (2000)
European flounder				X				B[a]P	Malmstrom (2000)
Fathead minnow				X	X			Anthracene	Tilghman-Hall and Oris (1991)
Fathead minnow					X			Fluoranthene	Diamond *et al*. (1995)
Fathead minnow							X	Fluoranthene	Farr *et al*. (1995)
Fathead minnow		X						Fluoranthene	Weinstein *et al*. (1997)
Fathead minnow				X				B[a]P	White *et al*. (1999)
Goby	X							B[a]P	Zheng *et al*. (2000)
Grouper	X							B[a]P, BNF	Peters (1995)
Guppy		X						DMBA	Hawkins *et al*. (1989)
Killfish	X			X				B[a]P	Willett *et al*. (1995)
Mummichog	X							B[a]P	Van Veld *et al*. (1997)
Mummichog				X				B[a]P	Rose *et al*. (2001)
Ovale sole	X	X						B[a]P	Au *et al*. (1999)
Oyster toadfish			X					DMBA	Seeley and Weeks-Perkins (1997)

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TABLE 11.3 (*continued*)

Species	Effects: Biochemical	Histopathologic	Immunological	Genetic	Reproductive	Development	Behavioral	Exposure	Reference
Rainbow trout		X						B[a]P	Hose *et al*. (1984)
Rainbow trout	X			X				B[a]P	Masfaraud *et al*. (1992)
Rainbow trout				X				B[a]P	Potter *et al*. (1994)
Rainbow trout	X	X						MC	Khan and Semalulu (1995)
Rainbow trout					X			Phenanthrene	Passino-Reader *et al*. (1995)
Rainbow trout	X							B[a]P	Cravedi *et al*. (1998)
Rainbow trout	X							Retene	Fragoso *et al*. (1999)
Sea bass		X						B[a]P	Lemaire *et al*. (1992)
Sea bass	X							BNF	Novi *et al*. (1998)
Sheepshead minnow		X						B[a]P, DMBA	Hawkins *et al*. (1991)
Tilapia		X	X					B[a]P	Holladay *et al*. (1998)
Tilapia		X	X					DMBA	Hart *et al*. (1998)
Tomcod	X							B[a]P, BNF	Courtenay *et al*. (1999)
Various species	X			X				B[a]P	Chen *et al*. (1999)
Zebrafish				X				B[a]P	Hsu *et al*. (1996)
Zebrafish	X	X				X		Retene	Billiard *et al*. (1999)

*B[a]P, benzopyrene; MC, methylcholanthrene; DMBA, dimethylbenzanthracene; BNF, β-naphthoflavone.

Presumptive evidence strongly indicates that the MFO enzyme induction observed in fish in many harbors, lakes and rivers worldwide, and represented in Table 11.1, is variably linked to contamination by PAHs. This is further supported by a large number of observations on induction in a variety of species upon exposure to complex mixtures of PAH, as well as individual PAH compounds (see Tables 11.2 and 11.3). Although studies on complex mixtures of PAH provide more information of environmental relevance than studies on single compounds alone, systematic studies on single compounds are critical for shedding light on which compounds or compound types may be causing different biological responses. Most of the observations on biochemical effects in Tables 11.2 and 11.3 refer to induction of MFO enzymes. Also, observations of induction in fish around petroleum development sites in the marine environment (e.g. Stagg *et al.* 1995) and in the vicinity of major oil spills (e.g. Kurelec *et al.* 1977; George *et al.* 1995) provide unambiguous evidence for PAH-linked induction in the environment. Further, both MFO induction and pathological effects have been observed in fish taken in the vicinity of natural petroleum seeps in the Santa Barbara Channel (Spies *et al.* 1996).

Is MFO enzyme induction itself of ecotoxicological importance, beyond being an index of chemical exposure sufficiently high to elicit a biological response? There is a body of literature associating MFO induction with the production of damaging free radicals and adducts, which are important in mutagenic and carcinogenic processes (e.g. Stegeman and Hahn 1993). Induction in fish has also been specifically linked with effects on reproduction, as well as correlated with various organ and cellular disturbances (e.g. Johnson *et al.* 1988; Payne *et al.* 1988; Au *et al.* 1999). Because induction of enzyme systems is also an energy-utilizing process, any energy loss of a non-essential nature could theoretically alter other biochemical and physiological processes in a maladaptive manner. Therefore induction of MFO enzymes, and especially prolonged induction of relatively high levels, can be considered a risk factor for fish health.

11.2.2 HISTOPATHOLOGICAL EFFECTS

Observations on visible signs of disease in marine fish, such as lymphocystis, skin ulceration and fin rot, date back several decades. During the early 1970s considerable interest arose as to whether chemical contamination could play a role in the etiology of these or other fish diseases and extensive studies began to appear from surveys in waters of various countries, including the USA (e.g. Couch 1985; Ziskowski *et al.* 1987), the Netherlands and Belgium (e.g. Moller 1981; Banning 1987), Germany (e.g. Dethlefsen *et al.* 1987) and the UK (e.g. Bucke *et al.* 1983). In addition to external diseases, more emphasis began to be placed on histopathological studies for neoplasms and other tissue and organ abnormalities, which can be powerful indicators of serious chemical injury. Many important observations have been made over the past 25 years linking histopathological effects including carcinogenesis in fish to polluted waters containing elevated levels of PAH. Examples of studies carried out in marine and estuarine waters of the USA, Canada, and Europe, where PAHs likely contributed to a variable degree to some of the effects observed in fish, are noted in Table 11.1. Most of the studies drew attention to the possible importance of PAHs while other studies, although not drawing attention to PAHs, were carried out in areas known to contain elevated levels of the compounds. Although not included as case studies in Table 11.1, some of the effects noted in fish around pulp mills (e.g. Khan and Payne 1997) could also be due in part to PAHs, since such large industrial plants can be expected to release a level of oil and grease into the aquatic environment. Examples of studies carried out in the Great Lakes and in rivers in the USA, where PAHs have been linked to various pathological effects, are also noted in Table 11.1.

Although the evidence linking adverse histopathological effects in fish to pollution is quite strong, it is difficult to resolve cause–effect relationships for various classes of chemicals. However, a few studies of freshwater and marine systems in the USA and Canada have provided very strong evidence for PAHs being a causal factor in fish pathology, including skin and skeletal disorders, liver abnormalities, and neoplasms. The linkage with PAHs is quite robust in these particular studies, since they were carried out at sites heavily contaminated with PAHs from creosote or coke.

As noted previously, creosote is a mixture distilled from coal tar, while coke is the residue left after coal distillation. Both are highly enriched in PAHs (and associated analogs and phenols) and commonly known to be quite mutagenic, carcinogenic, and cytotoxic. The Black River in Ohio was historically contaminated with rather extreme levels of PAHs from a coking plant, and up to 30% of the brown bullheads collected in the vicinity of the plant had neoplasms, including cholangiocarcinomas and hepatocarcinomas (Baumann and Harshbarger 1985). Another site where skin and liver tumors in fish have been linked to discharges from a coking facility is in Hamilton Harbor in Lake Ontario, near a large steel plant (Cairns and Fitzsimons 1988). In the case of creosote, fish collected from a portion of the Elizabeth River in Virginia, where creosote was a major contributor to the total PAH load in the river, displayed cancer and other pathological effects (Vogelbein *et al.* 1990). High levels of PAHs from creosote have also been associated with neoplasms and other liver disorders in marine fish, e.g., detailed studies carried out on English sole in Puget Sound found a relatively high prevalence of disease in Eagle Harbor which was historically linked with creosote contamination (Malins *et al.* 1985).

These case studies provide strong evidence for environmental PAHs contributing to carcinogenesis and other disorders in fish. Any case studies where pathological effects decline upon reduction or cessation of inputs of PAHs would provide the strongest evidence. Notable in this regard are observations on the marked decline in liver neoplasms in bullhead from the Black River upon closure of the coking facility (Baumann and Harshbarger 1995), and reductions of several types of liver diseases in English sole following a sediment capping project in Eagle Harbor (Collier and Myers, 1999).

Results obtained in experimental studies with fish chronically exposed to sediments containing petroleum or pyrolytic sources of PAHs, as well as various studies with extracts of PAHs from contaminated sediments, or industrial formulations such as creosote or for instances with specific PAHs, provide additional support for PAHs being a likely cause for some of the pathological effects found in fish in highly contaminated environments. Examples of several such studies are noted in Tables 11.2 and 11.3 and a few are discussed in more detail. Weathered petroleum contains complex mixtures of PAHs and aliphatic hydrocarbons, and whereas chronic exposure of flounders to sediments contaminated with weathered petroleum produced a variety of effects, no such effects were found in flounders similarly exposed to sediments containing a petroleum source of complex aliphatic hydrocarbons, indicating that PAHs were primarily responsible for the effects observed in fish with weathered petroleum (Payne *et al.* 1995). Epidermal hyperplasia and papillomas developed in brown bullheads upon skin painting with PAH extracts of contaminated sediments from the Buffalo River (Black *et al.* 1985). Hepatocellular neoplasms and a spectrum of other abnormalities were found in the livers of rainbow trout after microinjection of eggs with extracts of sediments from the Black River (Maccubbin *et al.* 1987). Similar results were obtained by Balch *et al.* (1995) with extracts of sediments from Hamilton Harbor. Other studies have been carried out with fish exposed *in vivo* to extracts of PAHs added to water or sediment. A variety of liver abnormalities and neoplasms were exhibited by medaka upon exposure to extracts of sediments from the Black and Fox Rivers (Fabacher *et al.* 1991). Fin erosion was also a sensitive response in medaka, as well as in spot exposed for short periods to relatively low levels of creosote in sediments (Sved *et al.* 1997).

Concerning individual PAHs, the carcinogenic potency of B[a]P is well known and has produced neoplasms in fish (e.g. Hawkins *et al.* 1991). Ultrastructural studies have also revealed a variety of pathological features in the liver and intestinal tissues of fish exposed to B[a]P (Lemaire *et al.* 1992; Au *et al.* 1999). PAHs that have little or no carcinogenic potential may have considerable potential to cause other pathological effects. Naphthalene is probably best known in this regard, but other PAHs are also important, e.g. severe fin erosion and liver necrosis were some of the effects found in catfish exposed to relatively low concentrations of acenaphthene for short periods (Dwivedi 2000). Also, anthracene, which is known to produce highly cytotoxic by-products in some organisms when they are simultaneously exposed to the chemical and ultraviolet light, damaged the gills of fish upon exposure to quite low levels of the compound (McCloskey and Oris 1993).

Considering the variety of information available from field and laboratory studies, it is reasonable to state that PAHs have the potential to play a role, and possibly sometimes a major role, in the production of pathological effects in feral fish, including skin disorders, organ abnormalities, and neoplasms.

11.2.3 IMMUNOLOGICAL EFFECTS

Immune responses can involve two components, a specific humoral component, such as antibody production, or a non-specific cellular component, such as phagocytosis of bacteria by white blood cells (e.g. Ellis 1989). Alteration of immune systems in fish by contaminants may affect their susceptibility to bacterial, viral, and parasitic infections, decrease their resistance to carcinogenesis, or impair vital functions such as tissue repair. There are few environmental studies which have linked chemically mediated diseases to impairment of immunological functions. Examples of a few such studies are noted in Table 11.1.

Impaired phagocytic functions, including reduced phagocytosis, chemotactic response and chemiluminescence, were found in oyster toadfish, spot and hogchoker taken from a site in the Elizabeth River in Virginia which, as previously noted, is heavily contaminated with PAHs (Weeks and Warinner 1984; Seeley and Weeks-Perkins 1991). Mitogen-stimulated lymphocyte proliferation was also suppressed in spot from the same River (Faisal and Huggett 1993). Hematological changes were noted in white suckers in the lower Sheboyan

River in Wisconsin, which contains high levels of PAHs (Schrank *et al*. 1997). Altered leucocyte numbers (and other effects) were also observed in winter flounders in St. John's Harbor, Newfoundland, in comparison with reference fish taken outside the harbor (French *et al*. 2000). Sediments in this harbor are contaminated throughout with high levels of PAHs but relatively low levels of PCBs and other organochlorines.

A series of recent studies (summarized in Arkoosh and Collier 2002) in the USA suggest that polluted estuaries may be one of the factors contributing to the decline of wild Pacific salmon, an issue of major socioeconomic interest in the USA and Canada. Juvenile salmon taken from contaminated estuaries in Puget Sound and challenged with pathogenic bacteria, exhibited higher mortality than reference fish similarly challenged. It is not known to what degree PAHs are linked with this important field observation of immunotoxicity in salmon, but these studies did show that laboratory exposure of salmon to either a single PAH or a PAHs model mixture could also cause increased mortality following pathogen challenge.

There are few studies dealing with the immunotoxicity to fish of mixtures of PAHs or individual PAHs. Some examples are given in Tables 11.2 and 11.3. Several immunological parameters were evaluated in rainbow trout exposed to creosote and one of the most notable changes was alteration of blood leucocytes. This specific study provided important dose–response information and it is of interest that the LOEC was quite low, around 1 ppb of PAHs in water (Karrow *et al*. 1999).

Weathered petroleum is highly enriched in PAHs, making it a suitable candidate for studies of mixtures of PAHs (see also Section 11.2.2). Macrophage centers represent the primitive analogs of lymph nodes in mammals and are believed to be an integral component of the cellular immune system in fish. Payne and Fancey (1989) recorded a change in the number of macrophage centers in the livers of flounders chronically exposed to a petroleum source of sediment PAHs at levels commonly found in nearshore and inland waters.

Alteration of immunological status can theoretically result in a variety of adverse outcomes, but increased susceptibility to infectious diseases is usually of most concern. Susceptibility to disease is important for any species, especially those identified by agencies as being 'species at risk'. The observation of compromised immune systems in juvenile salmon in Puget Sound, which may be linked to contamination by PAHs, is of note in this regard. Also of interest are laboratory and field observations on alteration of blood cells, which are an important immunological component and can be readily measured.

11.2.4 GENETIC EFFECTS

Damage to genetic material can theoretically lead not only to carcinogenesis and other pathological effects in animals living now, but also produce genetic diseases in future generations. Research on the genotoxic potential of various environmental chemicals, drugs, etc. have been ongoing for decades, with earlier work emphasizing cytogenetic and mutational studies and later investigations incorporating biochemical endpoints, such as specific mutations in DNA, DNA strand breaks, oxidative damage to DNA, or adduct formation (e.g. Pfeifer 1996).

An appreciation of the different types of screening carried out in the environment for genotoxic chemicals is provided by De Flora *et al.* (1991). Notably, a wide variety of individual and complex mixtures of PAHs, as well as environmental extracts enriched in PAHs, have exhibited genetic toxicity in different assays, including bacterial systems. The metabolism of PAHs is extensively covered in Chapter 5, including the formation of DNA adducts. This section briefly reviews other aspects of genetic toxicity, focusing on studies with fish proper.

Genetic toxicity of a more overt and 'visibly' harmful nature can be assessed microscopically with classical cytogenetic techniques. Also, within the past few years an electrophoretic technique based on light microscopy (the Comet assay), which measures DNA fragmentation of individual nuclei, has been used more extensively (e.g. Pfeifer 1996). Classical cytogenetic techniques revealed significant genetic toxicity in herring larvae, in association with the *Exxon Valdez* spill in Alaska, and effects were correlated with levels of PAHs found in mussels in the area (Hose and Brown 1998). Cytogenetic toxicity was also observed in fish larvae exposed as eggs to low levels of petroleum-derived PAHs (Carls *et al.* 1999). The studies with herring eggs are of special interest in this regard, since effects were noted with very low levels of PAHs from weathered oil, which should be essentially free of toxic monoaromatics such as benzene and xylene. DNA fragmentation, as assessed by the Comet technique, can also be interpreted as rather overt DNA damage, and it is of interest that blood cells of bullheads from the Detroit River in the USA and Hamilton Harbor in Canada displayed distinctly higher levels of DNA fragmentation than the cells of fish from reference sites (Pandrangi *et al.* 1995).

Damage to gametic DNA is of special importance, since effects could potentially be transmitted between generations. Studies on the induction of mutations or heritability of pollutant-induced mutations are difficult. However, some studies are beginning to appear pointing to a potential for PAHs to induce mutations in fish and possibly heritability; e.g. exposure of fish larvae to weathered Prudhoe Bay crude oil produced mutations in high frequency at hot spots in oncogenes (Roy *et al.* 1999). Although it is not known whether mutations were actually involved, it is of special interest that White *et al.* (1999), in their studies with fathead minnows exposed to B[a]P, produced heritable reproductive effects in fish removed from the original exposure by at least one generation.

There is a substantial body of evidence from field and laboratory studies indicating that PAHs are a significant risk factor for genetic toxicity in fish. Most studies have dealt with the formation of DNA adducts (Chapter 5), but classical cytogenetic techniques have demonstrated genetic damage in fish larvae, with some of the lowest levels of PAHs known to produce effects, while field studies indicate that environmental levels of PAHs may sometimes be sufficiently high to cause overt damage to DNA in the blood cells of adult fish. Interestingly, there is also recent evidence pointing to a potential for PAHs to produce mutations in reproductive cells, resulting in effects which could be transmitted intergenerationally.

11.2.5 REPRODUCTIVE EFFECTS

Reproductive toxicity is one of the most important types of toxicity because of its potential for producing adverse effects at a population level. Hypotheses about a role for pollutants in the low fertility and high prevalence of embryo mortality in some stocks of salmon in the Great Lakes go back to the 1970s (e.g. Leatherland *et al.* 1998). Organochlorines and especially PCBs have often been implicated in reproductive toxicity, and studies in the coastal waters of Germany and Denmark have provided evidence for effects (Dethlefsen 1988). Establishing cause–effect relationships for hypotheses about reproductive toxicity is obviously quite difficult, especially in relation to the potential for broad-scale geographical effects. Also, since PAHs are rapidly metabolized and do not necessarily bioaccumulate to any extent, there is always a possibility that they are causing toxic effects yet providing no evidence for the effects. Point sources of contaminants are more suitable for study, and it is of interest that within the past decade or so there have been a number of observations on altered reproductive functions in fish taken close to pulp mills (e.g. McMaster *et al.* 1996). Natural and selected industrial chemicals have also recently been linked to feminizing effects in male fish taken near sewage outfalls (e.g. Purdom *et al.* 1994). Such observations have stimulated considerable interest about the potential for contaminants, including PAHs, to affect fish reproduction.

Are field studies at sites contaminated with relatively high levels of PAHs indicating PAHs to be a risk factor for reproductive toxicity? English sole from sites in Puget Sound exhibited changes in gonadal development, altered levels of plasma steroids and reduced spawning success (Johnson *et al.* 1988; Casillas 1991) but effects were less pronounced in rock sole in the area (Johnson *et al.* 1998). Also, winter flounders from contaminated estuaries in the northeastern USA displayed little evidence of reproductive dysfunction (Johnson *et al.* 1994). But it was noted that English sole, unlike winter flounders, reside in contaminated estuaries for longer periods throughout their period of vitellogenesis and the greater exposure during this period may account in part for the apparent differential sensitivity between the two flounder species.

A study of freshwater fish in highly degraded habitats found no evidence of serious reproductive dysfunction as displayed by fecundity indices (Lesko *et al.* 1996). Brown bullheads from the Black and Cuyahoga Rivers in Ohio containing sediments with elevated levels of PAHs actually had fecundity indices that were equal to or greater than those of reference populations. However, the potential for producing adverse effects on the fecundity in fish in polluted systems could be compensated for by other factors, such as enhanced food supply due to the absence of competing predators (whose absence could also be linked directly or indirectly to contamination). The authors suggest such a possibility for the lack of effects in bullheads in these highly degraded systems. Also, it should be noted that although fecundity is commonly used as a sign of reproductive health, the quality of eggs or sperm could theoretically be impaired without effects on fecundity or other morphological indices.

The chronic toxicity studies needed to elucidate the effects of chemicals on reproduction in aquatic organisms having a long reproductive cycle require facilities and resources not commonly found in most laboratories. Accordingly, there is limited experimental information to assess the effects of complex mixtures of PAHs on reproduction in fish. Since fecundity indices may often be insensitive indicators of reproductive toxicity, it is also important to place greater emphasis on studies related to the quantity and quality of offspring produced. Nagler and Cyr (1997) recently reported quite novel findings in this regard, with respect to the impairment of sperm quality and effects on egg hatchability in American plaice chronically exposed to sediments highly contaminated with PAHs from the Baie des Anglais in Quebec.

Are there laboratory studies with complex mixtures of PAHs relatively free of other contaminants or individual PAHs to support hypotheses about a potential for PAHs to adversely affect fish reproduction? Certain PAHs closely resemble steroid hormones and exhibit weakened estrogenic or anti-estrogenic responses in some model systems (Nicolas 1999; Navas and Segner 2000). Benzo[a]pyrene decreased circulating levels of 17β-estradiol in fish (Thomas 1988; Singh 1989) and effects on both steroidogenesis and steroid secretion have been noted in *in vitro* studies with ovarian tissue (Afonso *et al.* 1997; Monteiro *et al.* 2000). We measured steroids in the plasma of fish chronically exposed to sediments highly enriched in either petrogenic or pyrolytic sources of PAHs (Idler *et al.* 1995). Steroid concentrations were reduced with both sources of PAHs but it is of interest to note that similar results were obtained with sediments highly contaminated with aliphatic hydrocarbons.

There is some evidence from field and laboratory studies indicating PAHs as a potential risk factor for reproduction in fish. However, since long-term chronic toxicity studies are rather difficult, with reproductive studies being some of the most difficult, experimental evidence is generally lacking about the importance of PAHs to cause reproductive toxicity in fish, especially in relation to dose–response relationships. Also, although measures such as gonad morphometrics and fecundity are commonly used to assess reproductive health in fish, these endpoints may be underestimating potential reproductive effects and more emphasis should also be placed on assessing the quality of sperm and eggs.

11.2.6 DEVELOPMENTAL EFFECTS

Potential for effects on growth and development is also an important issue, with developing organisms usually displaying a greater sensitivity to chemicals than adults. Since adverse developmental effects can be expected with any chemical should dosages be sufficiently high, the environmental relevance of chemical dosage is always a key factor to consider. Notable in this regard are observations that weathered oil, which is essentially free of low molecular weight monoaromatic compounds such as benzene and xylene, produced sublethal effects in herring larvae exposed to extracts of oil in water containing as little as 0.4 ppb PAHs (Carls *et al.* 1999) (Table 11.2). Relatively low concentrations of PAHs in sediment could also affect growth of larvae. Misitano *et al.* (1994) indicated a potential for effects on growth as assessed by DNA content, in larval surf smelt exposed for 96 h to sediment containing 1.5 ppm PAHs. Effects on juvenile fish, which are important in recruitment, would be of special relevance from a fisheries perspective. Growth was reported to be reduced in juvenile flounder chronically exposed to as little as 1.6 ppm of a petroleum source of PAHs in sediment (Moles and Norcross 1998). Provided for general interest in the tables are other examples of 'developmental' effects, but these mainly relate to effects such as slight changes in organ or somatic condition indices and not to developmental effects *per se.*

Although experimental data is limited, there is indication that development may be affected in larval and juvenile fish upon exposure to very low levels of PAH.

11.2.7 BEHAVIORAL EFFECTS

Studies on animal behavior have generally received little attention in aquatic ecotoxicology but alteration of behavior may be an important environmental consideration for fish in some circumstances. Negative effects might include susceptibility to predation, alteration of homing behavior or avoidance of feeding areas. On the other hand, avoidance of contaminated areas could reduce toxicological risk. It is of interest that under Canadian law, avoidance of feeding areas or loss of habitat could have paramountcy over toxicological risks to fish, short of observations on lethality.

There is limited information to assess behavioral effects of PAHs on fish. A few examples are given in Tables 11.2 and 11.3. Presumably of most importance would be sediments impacted with relatively high levels of PAHs such as around oil spill sites. Feeding behavior was affected in goby by suspended sediments contaminated with a diesel oil source of PAHs in the 200 ppm range, with cessation of feeding around 200 ppm (Gregg *et al.* 1997). Alteration of feeding behavior was also noted in spot exposed to a diesel source of PAHs in the 120 ppm range (Hinkle-Conn *et al.* 1998).

Weathered oil remaining in sediments after oil spills has been the subject of considerable attention. It is of interest that pathological effects, including altered swimming behavior, were exhibited by herring larvae hatched from eggs exposed to weathered *Exxon Valdez* oil containing 1 ppb PAHs (Carls *et al.*

1999). As previously noted, this is an extremely low level of PAHs to produce a biological effect.

Considering contamination of water or sediments, contamination of sediments with relatively high levels of PAHs is probably of most importance with respect to any potential for effects on fish behavior. It is not known whether the large quantities of PAHs in water that are expected after large oil spills might affect important behavioral responses in fish, such as alteration of homing behavior to natal rivers by salmon, but compounds with greater solubility and aromaticity, such as xylenes and phenols for instance (or low molecular weight sulfur compounds), are likely of greater importance in this regard.

11.3 CONCLUSION

This overview attempts to summarize information from field and laboratory studies over the past 15 years or so, linking various biological effects in fish to PAHs. It includes information on biochemical, histopathological, genetic, immunological, reproductive, developmental and behavioral effects. The suite of effects are grouped in relation to (a) field studies, (b) laboratory studies with complex mixtures of PAHs and (c) individual PAHs.

All relevant studies could not be included but those listed are a fair representation of biological effects that have been reported. Particular attention has also been given in the case of field studies to observations on different species in various countries. Although select numbers of studies noted in the tables are discussed, the studies altogether provide a body of information for use in a weight-of-evidence approach for assessing the ecotoxicological potential of PAHs for fish. This approach is useful because of the improbability of establishing either a strict scientific or 'legal' standard of causal evidence for regulatory bodies to use in assessing environmental effects associated with mixed contamination. Combining field and laboratory studies and using a weight-of-evidence approach, we suggest that levels of PAHs commonly found in many marine and freshwater environments are causing or contributing to health effects in fish.

Although use of different analytical methods makes it difficult to commonly equate levels of PAHs reported to be present in the environment, it is noted that concentrations of PAHs in sediments in the 10 ppm range have been found in coastal, estuarine and riverine waters of a number of countries, while guidelines indicate potential for effects in fish in the 1 ppm range or lower. Also, concentrations of PAHs in water as low as 1 ppb have been reported to cause serious effects in fish larvae and sublethal effects in fish. This leads to the question of whether contamination by PAHs along major shipping routes, and especially oil tanker routes, or for instance in association with release of large volumes of production waters at petroleum development sites, is adversely affecting populations of fish larvae, including species of commercial importance. It is noted in this regard that concentrations of PAHs as high as in the 100–200 ppb range have been reported to be present along major tanker routes in the Arabian Sea.

Most studies to date have centered around elucidating effects on fish health in association with elevated levels of PAHs in sediments in urban coastal zones or similarly contaminated rivers. Given the apparent extent of contamination in these areas, which may increase, it is important to continue to address potential fish health problems in such regions of concern. This assessment should also place more emphasis on laboratory-based chronic toxicity studies in order to provide better insight into dose–response relationships and thus better information for development of sediment quality guidelines. However, effects have also recently been reported with very low concentrations of PAHs in water. This points to a potential for effects on fish and especially larvae over broad-scale areas away from urban coastal zones and this aspect needs to be addressed much more fully.

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TECHNICAL REPORT

PETROLEUM SAFETY AUTHORITY NORWAY (PSA)

MATERIAL RISK - AGEING OFFSHORE INSTALLATIONS

REPORT NO. 2006-3496
REVISION NO. 0

DET NORSKE VERITAS

DET NORSKE VERITAS

TECHNICAL REPORT

Date of first issue:	Project No.:
08.11.2006	71610482
Approved by: Bjørn-Andreas Hugaas, Head of Section	Organisational unit: Materials Technology and Failure Investigation
Client: Petroleum Safety Authority Norway (PSA)	Client ref.: Rolf H. Hinderaker

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Summary:

On request from Petroleum Safety Authority Norway (PSA), DNV has prepared technological summaries regarding material risk on ageing equipment and installations in the oil and gas industry offshore. The report contains an introductory chapter on degradation mechanisms in general, followed by five sections containing technological reviews of metallic materials with respect to degradation mechanisms and failure modes, and its effect on ageing installations and equipment. Five areas, which are exposed to degradation of materials due to ageing, are covered in this report:

- Load bearing structures (concrete and steel)
- Pipelines
- Subsea equipment
- Drilling and wells
- Mooring system

Specific structures, systems or equipment have been selected from each of the above mentioned areas, and a technological review has been given based on in-house experience. This contains an evaluation of:

- Failure modes introduced by the degradation mechanism
- Occurrence of the degradation mechanism
- Limitations of the material introduced by the degradation mechanism
- Uncertainty of the material when degraded
- Future challenges of the material related to the degradation mechanism
- Effect of the degradation mechanism on the robustness of the installation

Report No.:	Subject Group:
2006-3496	E3
Report title: Material risk - Ageing Offshore Installations	
Work carried out by: K.Lønvik, B.H.Leinum, E.B.Heier, A.Serednicki, O.E.Gjørv (NTNU), T.Myhre, B.Søgård, L.H.Moen, B.E.Sogstad, M.O.Saugerud	
Work verified by: See Section 1 "Introduction"	

Date of this revision:	Rev. No.:	Number of pages:
08.11.2006	0	58

Indexing terms

Key words	Service Area
Corrosion, Erosion, Wear, HISC, Fatigue, Corrosion fatigue, Degradation, Concrete, Steel Structures, Pipelines, Subsea Equipment, Drilling, Wells, Mooring System	Asset Operation
	Market Sector
	Upstream, Oil and Gas

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1 INTRODUCTION

On request from Petroleum Safety Authority Norway (PSA), DNV has prepared technological summaries regarding material risk on ageing equipment and installations in the oil and gas industry offshore. The report contains an introductory section on degradation mechanisms in general, followed by five sections containing technological reviews of metallic materials with respect to degradation mechanisms and failure modes, and its effect on ageing installations and equipment. Five areas, which are exposed to degradation of materials due to ageing, are covered in the report:

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A summary of the degradation mechanisms and failure modes relevant within the five areas listed above, is presented in Appendix A.

The authors and the persons who have performed the verification for each section are listed below:

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3. Load bearing structures, steel
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7. Mooring system
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2 DEGRADATION MECHANISMS

Relevant degradation mechanisms are presented in Section 2.1.

All mechanisms, which are relevant for the five areas mentioned in Section 1, are included in this chapter.

2.1 Degradation mechanisms

A degradation mechanism is here defined as a disintegration of a metallic material due to the impact of the operating environment and forces. Degradation can be due to erosion, corrosion or stresses induced by cyclic/dynamic loads and other specific environmental impacts. The degradation mechanism can result in metal loss (as uniform or localised attacks) or cracking (e.g. fatigue, stress corrosion cracking, embrittlement). Degradation mechanisms related to metal loss and fatigue are typically time dependent (e.g. wall thinning), whilst cracking mechanisms are of more abrupt nature.

2.1.1 Erosion

Erosion can be defined as physical removal of surface material due to numerous individual impacts of solid particles, liquid droplet or implosion of gas bobbles (cavitation). Erosion is a time dependent degradation mechanism, but can sometimes lead to very rapid failures.

In its mildest form, erosive wear appears as a light polishing of the upstream surfaces, bends or other stream-deflecting structures. In its worst form, considerable material loss can be obtained.

2.1.2 Corrosion

2.1.2.1 General

Corrosion is caused by a chemical (or electrochemical) reaction between a metal and its environment that produces a deterioration of the material and sometime its properties. For corrosion to occur, the following basic conditions must be fulfilled:

- metal surface exposed to environment (bare steel in physical contact with the environment)
- electrolyte (e.g. water containing ions, the electrolyte must be able to conduct current)
- an oxidant (a chemical component causing corrosion (e.g. oxygen, carbon dioxide)

If one of these conditions is not present, no corrosion will occur.

Table 2-1 summarises prospective corrosion mechanisms for subsea oil and gas production equipment.

The presence of organic acids and sulphur containing compounds (e.g. elemental sulphur) may aggravate the corrosion in the system.

Table 2-1 Internal and external corrosion mechanisms in a subsea oil and gas production environment.

Corrosion Mechanism	***External***	***Internal***	***Chemical reaction***	***Time dependency***
O_2-corrosion	X	X	$2Fe + H_2O + 3/2O_2 \rightarrow 2FeO(OH)(s)$ ("rust")	Time dependent
CO_2-corrosion [1)] (sweet corrosion)	NA	X	$Fe + H_2O + CO_2 \rightarrow FeCO_3(s) + H_2$	Time dependent
Microbiologically induced corrosion (MIC)	X	X	Fe + "bacteria related oxidant" $\rightarrow Fe^{2+}$	Time dependent/ abrupt nature
Sulphide stress cracking (SSC) (corrosion due to H_2S)	NA[2)]	X	$2H+ \rightarrow H\cdot(ads)$ $H\cdot(ads) \rightarrow H_2(ads)$ (inhibited by H_2S) $H\cdot(ads) \rightarrow H\cdot(abs)$	Abrupt nature

[1)] Not anticipated on corrosion resistant alloys

[2)] Under certain conditions high levels of H_2S might occur in the seabed, however, such condition is not anticipated to occur on the Norwegian shelf.

2.1.2.2 External corrosion

External corrosion is for most submerged equipment controlled by the use of an external corrosion coating and a cathodic protection (CP) system. The design of the CP system is dependent on the design life of the equipment and the type and quality of the external coating system in question.

Some subsea components may not be provided with a CP-system (e.g. for chains) or CP will not be efficient due to shielding (e.g. water filled hollow profiles). For carbon steel components a corrosion allowance (CA) must then be added. The CA that must be added will depend on the availability of oxygen (oxidant). For areas with limited access of oxygen, such as within hollow profiles of structural steel, the corrosion rate will be low and a moderate CA is tolerable, whereas for instance chains that are freely exposed to seawater, a higher corrosion rate must be accounted for. The CA is normally determined as a part of the design and is based on the specified design life of the component.

Certain corrosion resistant alloys (CRA's) and titanium are resistant to seawater corrosion under North Sea ambient seawater conditions and can be used without cathodic protection.

2.1.2.3 Hydrogen Induced Stress Cracking

Cathodic protection may be detrimental for some materials due to Hydrogen Induced Stress Cracking (HISC). Hydrogen Induced Stress Cracking (HISC) is caused by a combination of load/stress and hydrogen embrittlement (HE) caused by the ingress of atomic hydrogen into the metal matrix formed at the steel surface due to the cathodic protection (CP).

High strength steel (SMYS > 500 MPa) and some corrosion resistance materials (13Cr-steel and duplex stainless steels) are susceptible to HISC, see /1, 2, 3, 4/. Solution annealed austenitic stainless steels and nickel based alloys are generally considered immune to HISC.
DNV RP B401 Sec. 5.5 /1/ gives recommendations with regards to materials maximum hardness level and the specified minimum yield strength for safe combinations with CP. Bolts in martensitic steels heat treated to SMYS up to 720 MPa and maximum hardness level of 350HV (ASTM A182 grade B7 and ASTM A320 grade L7) have documented compatibility with CP (see also Norsok M-001 Sec. 5.6 /5/).

Factors influencing HISC of duplex stainless steel have been recapitulated in a draft version of DNV RP F-112 /2/ with recommendation for design criteria based on best practice and on today's knowledge (strain/stress criteria).

HISC is abrupt of nature and it is expected to occur during the first years of the installations design life if the conditions are ideal.

2.1.2.4 Internal corrosion

CO_2-corrosion; CO_2-corrosion or sweet corrosion is not anticipated for corrosion resistant materials (e.g. 13Cr, 316L, 22Cr, 25Cr, Alloy 625). Carbon steel, however, will be subjected to CO_2-corrosion. The corrosion rate is dependent on the partial pressure of CO_2, the temperature, the flow regime and the water in-situ pH. The corrosion takes the form of localised- ('pitting'), uniform- and grooving- (e.g. longitudinal, transverse) attacks and is a time dependent degradation mechanism. CO_2-corrosion can be mitigated by the use of corrosion inhibitors and/or by pH- stabilisation of the process fluid (primarily applicable for pipelines).

O_2-corrosion; Internal corrosion due to the presence of O_2 is in principle not expected in oil and gas production systems since no oxygen shall be present in the process medium. Ingress of oxygen may increase the corrosion in the system.

Water used for water injection can be either deaerated or aerated, which will have an impact on the corrosivity. Due to the removal of oxygen in deaerated water, the corrosion rate of carbon steel will be low, whereas in systems carrying aerated water a higher corrosion rate must be anticipated. Oxygen corrosion is a time dependent corrosion mechanism and takes principally the form of uniform corrosion, but localised attacks may also occur ('pitting').

Corrosion resistance alloys (CRA's) and titanium can be used for seawater service but there are certain design limitations regarding the use of such materials (e.g. temperature, presence of crevices, chlorination etc.). Corrosion of CRA takes the form of localised attack. Unfortunate combination of material and operating environment will for most cases result in a corrosion failure during the initial phase of an installation's life.

Environmental cracking due to H_2S; Corrosion due to the presence of H_2S is primarily related to environmental cracking (i.e. sulphide stress cracking (SSC)). Both carbon steel and CRA's are susceptible to SSC. The risk for SSC is dependent on the partial pressure of the H_2S, the in-situ pH-value, total tensile stress, chloride ion concentration, presence of other oxidant etc. (for details reference is made to ISO-15156). Below a critical partial pressure of H_2S no SSC is expected to occur. However, for partial pressures above this limit there is an increasing risk for SSC and the environmental condition is termed as sour. The resulting failure mode is cracking and it is of abrupt nature. SSC is controlled by specification of the material properties (e.g. hardness) and the manufacturing process. For susceptible materials, environmental cracking is expected to occur during the initial phase of production and is not expected to have a time dependent development similar to 'sweet' corrosion.

Older petroleum installations may experience a souring of the wells (the produced amount of H_2S increases) and the production environment turns from sweet to sour. The risk for environmental cracking should for such cases be subjected to evaluations with respect to the material properties and the new service condition.

Microbiologically Induced Corrosion (MIC); The two best known bacteria of concern for the oil and gas industry are the sulphate reducing bacteria (SRB) and the acid producing bacteria (APB). They may live synergistically in colonies attached to the steel surface, where the SRB bacteria live beneath the APB colony. SRB bacteria live in oxygen-free environments and use sulphate ions in the water as a source of oxygen. H_2S is produced as a waste product from the SRB, producing a corrosive environment locally in connection with the colony of bacteria. The risk for obtaining MIC will depend on the availability of nutrients, temperature, water and flow condition. MIC takes the form of localised attack causing a pinhole leakage of a pipe. High corrosion rate can be anticipated (>1 mm/year) if the conditions are ideal. MIC has been obtained in oil production systems as well as on steel exposed (e.g. anchor chains) to seabed sediments. The location of MIC is difficult to predict. For pipeline systems, treatments with biocide may be effective as a preventive measure. A common source for bacteria in a closed system is seawater. Use of untreated seawater for hydro testing should therefore be avoided.

Galvanic corrosion; Galvanic corrosion may occur when there is an electrical coupling between dissimilar metals. The least noble material (anode) will be sacrificed on behalf of the noblest material (cathode). The extent of accelerated corrosion resulting from galvanic coupling is affected by the electrochemical potential difference between metallic couple, the nature of the environment (corrosivity) and the area ratio of anodic- and cathodic areas (small anode to cathode area ration is unfavourable).

Galvanic corrosion is a time dependent form of corrosion and result in a uniform corrosion attack. The possibility for obtaining galvanic corrosion should be evaluated during the design phase. For cases where a galvanic couple is inevitable, a distance spools of a non-conducting material can be installed or installation of a galvanic spool with sufficient wall thickness where the material is intended to corrode (i.e. sacrificial spool).

2.1.3 Fatigue

Fatigue failures occur in parts which are subjected to alternating, or fluctuating (other used terms are dynamic or cyclic), stresses. Fatigue cracking is usually initiated at stress raisers such as sharp geometric transitions, welds, notches or internal material flaws such as slags, cracks (e.g. quench cracks or weld lack of fusion defects). A minute crack starts at a localized spot and gradually spreads over the cross section until the component/member fails due to overloading of the remaining cross section area.

Fatigue results in an almost brittle-appearing fracture, with no gross deformation at the fracture. Crack propagation may be divided into stages, the most important being “initiation” and “propagation”. Depending on the material, the stress level and eventual environmental impact - initiation or propagation may be the stage constituting the main part of the lifetime of the component/member.

Fatigue failure is caused by a critical localized tensile stress which is very difficult to evaluate and therefore design for fatigue failure is based primarily on empirical relationships using nominal stresses. A fatigue failure can usually be recognized from the appearance of the fracture surface, which shows a smooth region, due to the fatigue crack propagation through the section (being more or less insensitive to the microstructure and hence orientated perpendicular to the principle direction of the applied stress), and a rough region, where the member has failed when the remaining cross section was no longer able to carry the load. Frequently the progress of the fracture is indicated by a pattern of parallel lines, or “beach marks”, progressing inward from the point of initiation of the failure.

Three basic factors are necessary to cause fatigue failure. These are (1) a maximum tensile stress of sufficiently high value, (2) a large enough variation or fluctuation in the applied stress, and (3) a sufficiently large number of cycles of the applied stress. In addition, there are a host of other variables, such as stress concentration (e.g. geometric transitions), corrosion (localised corrosion e.g. preferential weld corrosion), temperature (stresses induced by linear thermal expansion or differences in thermal expansion coefficients between different materials), overload (over-torque of bolts), metallurgical structure (e.g. direction of texture vs. direction of applied stress), residual stresses (welding or forming residual stresses), and combined stresses, which tend to alter the conditions for fatigue.

Recommendations for design when considering fatigue resistance of offshore structures are given in DNV-RP-C203 /6/. This recommended practice contains among other things a collection of fatigue resistance (S-N) curves developed for different configurations (e.g. types of welded joints), surface finish and environmental conditions. For some of the curves assumptions regarding defect sizes, and hence corresponding NDT requirements are given.

Crack propagation may be both adversely or favourably affected by variable amplitude loading. Intermittent high stresses may create large plastic zones with “residual” compressive stresses ahead of the crack toe and hence retard crack growth.

2.1.4 Corrosion fatigue

On a general level fatigue is affected by environmental conditions and in particular by corrosion. HISC and sour service conditions, as described in Section 2.1.2.2 and 2.1.2.3, respectively, may facilitate fatigue crack initiation. Metal loss by corrosion will generally enhance crack growth, but under reversed loading conditions (tension – compression) corrosion products may reduce the "impact" of the total stress range due to "crack closure".

Corrosion fatigue occurs in metals as a result of the combined action of cyclic stress and a corrosive environment. For a given material, the fatigue strength (or fatigue life at a given maximum stress value) generally decreases in the presence of an aggressive environment.

When corrosion and fatigue occur simultaneously, the chemical attack accelerates the rate at which fatigue cracks propagate. Materials which show a definite fatigue limit when tested in air at room temperature show no indication of a fatigue limit when the test is carried out in a corrosive environment.

Corrosion fatigue crack growth might be influenced by many variables, such as those listed in Section 2.1.3, but also by environmental variables (gaseous or liquid environment, partial pressure of damaging species in gaseous environments, temperature, pH).

A number of methods are available for minimizing corrosion fatigue damage:

- The choice of material for this type of service should be based on its corrosion resistant properties rather than the conventional fatigue properties (ex. stainless steel over heat-treated steel)
- Protection of the metal from contact with the corrosive environment by metallic or non-metallic coatings (provided that the coating does not become ruptured from the cyclic strain)
- Addition of a corrosion inhibitor in closed systems to reduce the corrosive attack
- Elimination of stress concentrators by careful design

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/5/ Norsok M-001 rev. 4, Materials selection

/6/ DNV-RP-C203 (2005) Fatigue design of offshore steel structures

3 LOAD BEARING STRUCTURES

3.1 Concrete

3.1.1 Introduction

When the first concept of fixed concrete structures for offshore oil and gas exploration and production in the North Sea was introduced in the late 1960´s, the offshore technical community showed much scepticism. At the same time, however, the results of a comprehensive field investigation of more than 200 existing conventional concrete sea structures such as bridges and harbour structures along the Norwegian coastline were published /1/. The overall good performance of these structures recorded even after a service life of 50-60 years contributed to convincing the technical community that also concrete could be a reliable construction material for oil and gas installations in the North Sea. However, the appearance of corrosion on embedded reinforcement steel that typically took place on the conventional concrete sea structures already after a service period of 5-10 years was not acceptable. Therefore, in order to gain acceptance for the first offshore concrete platform in the North Sea, both increased concrete qualities and concrete covers beyond the requirements of current concrete codes were required. Secondly, much stricter programs for QA/QC compared to the existing design and construction practice had to be introduced.

Already during the construction of Ekofisk Tank, the first edition of “Recommendations for design and construction of concrete sea structures” was published by the international organization for prestressed concrete /2/. Thereafter, both Norwegian Petroleum Directorate in their Regulations /3/ and Det Norske Veritas /4/ in their Rules had adopted new and stricter durability requirements for fixed offshore concrete structures.

After the first breakthrough for use of concrete in developing the Ekofisk oil field, a rapid development took place. During the period from 1973 to 1995 altogether 28 major concrete platforms containing more than 2.5 million cubic meters of concrete were installed in the North Sea. Several of these installations are now successively approaching the intended service life of 25-30 years.

Considering the harsh and hostile marine environment in the North Sea, the question has been raised on how these concrete structures have performed so far. As there is need for a continued service of the installations beyond the service life originally designed, the question also has been raised for how long these fixed concrete structures can be safely operated.

3.1.2 Main degradation mechanisms

Although deteriorating mechanisms such as chemical seawater attack, freezing and thawing, and expansive alkali reactions all present some potential durability problems for offshore concrete structures, it is relatively easy to avoid such problems by taking the necessary precautions at an early stage of planning, designing and construction. For oil containment vessels, aggressive bacteriological environment may also represent a potential problem of concrete degradation. For a dense, high-quality concrete, however, such a degradation should not represent any durability problem.

The concrete platforms located in the Norwegian Sector of the North Sea were designed and constructed in accordance with the requirements specified in /2, 3, 4/, and reflected the current “state-of-the-art” which in the main are still relevant and acceptable.

For concrete structures in the marine environment, extensive experience demonstrates, however, that it is not the disintegration of the concrete itself but rather the electrochemical corrosion of embedded steel reinforcement and pre-stressed tendons which poses the most critical and greatest threat to the durability and long-term performance of the structures /5/. As long as it is possible to prevent or retard the chlorides penetration into concrete, all embedded steel is very efficiently protected from corrosion by electrochemical passivation of the highly alkaline concrete.

The high alkalinity of concrete may be neutralized by a reaction between the atmospheric carbon dioxide and the calcium hydroxide solution in the concrete. Such a carbonation process will also impair the passivity of embedded steel causing steel corrosion. For a dense, high-quality concrete in a moist environment, however, carbonation is limited to a very thin surface layer and does not represent real problem to marine concrete structures.

As soon as the chlorides from seawater have reached embedded reinforcement and prestressed steel and the passivity of the steel is broken, a complex system of galvanic cells will develop causing electrochemical corrosion of the steel /6/. The rate of embedded steel corrosion will then primarily be controlled by the availability of dissolved oxygen in the cathodic areas and the electrical resistivity of the concrete. The area ratio of the depassivated parts (anodic areas) and the passive parts (cathodic areas) in the galvanic cells is also an important factor for the corrosion rates. The electrical resistivity of concrete in moist marine environment is normally so low that it does not become the governing factor for the electrochemical corrosion process /7/.

Both in the tidal and in the splash zones of concrete sea structures, oxygen is available in plenty, so that a high corrosion rate can take place in the zones /8/. Only for the constantly submerged parts of high quality concrete structures, the availability of oxygen is generally so low that an electrochemical corrosion of embedded steel does not represent any practical problem /9/.

In all concrete structures, a certain amount of cracks in the concrete cover may freely expose parts of the embedded steel. In a moist marine environment, however, crack widths of up to 0.5 mm would normally not represent any corrosion problems /10/. For static cracks in submerged parts of the structure, even wider cracks may be tolerated due to a filling up of the cracks by various chemical reaction products (Calcareous depositions consisting mainly of magnesium hydroxide, calcium carbonate and corrosion products which block up the cracks and reduce rate of steel corrosion. The process is known in the literature as “self-healing of cracks”). For dynamic cracks, however, wider (in excess of 0.5 – 0.6 mm) cracks may represent a potential corrosion problem /11/.

In the submerged part of offshore concrete structures, a variety of freely exposed metallic components, such as pipes, penetration sleeves, clamps, brackets, supports and other fixtures are in metallic connection with the embedded reinforcement steel and may represent a special corrosion problem /12/. In such a case, the freely exposed metallic components represent small anodic areas in metallic connection with huge cathodic areas of the embedded reinforcement steel acting as catchments areas for oxygen. In order to control this galvanic corrosion problem, an effective cathodic protection system for all freely exposed metallic components is essential. Normally, such a cathodic protection has been based on sacrificial anodes.

3.1.3 History

Extensive surveying programs both above and below water are regularly being carried out for all offshore concrete structures in the North Sea, however no recent information on the current status of these structures is available for general public. In a State of the Art report from 1994 /13/, very little corrosion problems on embedded steel was reported; the most serious problems were related to accidental loads from ships and falling items. More detailed inspections of eleven of the oldest concrete structures installed during 1973–78 and reported in 1982, also showed their generally good conditions /14/. Inspection of the concrete platforms on the Statfjord and Gullfaks Oil Fields in 1992 also reported a generally good condition of the concrete /15/.

After 20 years of service, regular inspections of the relatively wide concrete cracks found at the foundation of Frigg CDP-1 Platform had revealed no serious corrosion in the cracked concrete /16/. After 18 years of service, corrosion monitoring based on embedded steel tubes in the Frigg TCP-2 Platform did neither report any steel corrosion /16/. Also laboratory-based investigations have shown that the risk of steel corrosion in cracked submerged concrete appears to be less severe than what originally expected /10, 11/.

Only for a few of the concrete platforms in the North Sea, more systematic investigations of chloride penetration have been carried out. All of these investigations clearly demonstrate that a certain rate of chloride penetration does take place, but only at a slower rate compared to that typically observed on conventional concrete structures /17/. Figure 3-1-1 and 3-1-2 show the chloride penetration into the concrete of Statfjord A Platform and Ekofisk Tank after 8 and 17 years of exposure, respectively. Figure 3-1-3 and 3-1-4 show the chloride penetration into the concrete of Brent B and Brent C Platforms, respectively, after 18-20 years of exposure. For Oseberg A Platform, where the concrete cover was partly less than what had been specified, serious corrosion problems occurred already at an early stage of operation. For this structure, repairs in the form of cathodic protection were carried out in 1998 /21/.

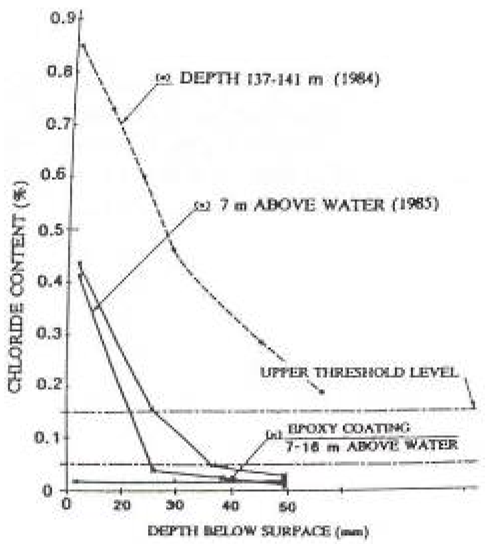

Figure 3-1-1 Chloride penetration into concrete of Statfjord A Platform (1977) after 8 years of exposure /18/.

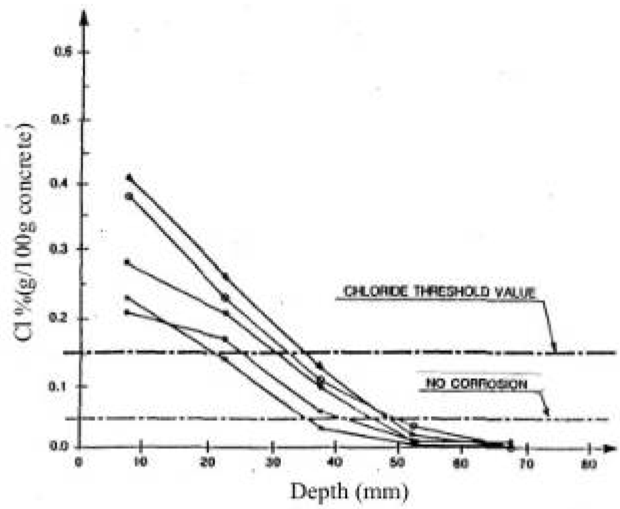

Figure 3-1-2 Chloride penetration into concrete of Ekofisk Tank (1973) after 17 years of exposure /19/.

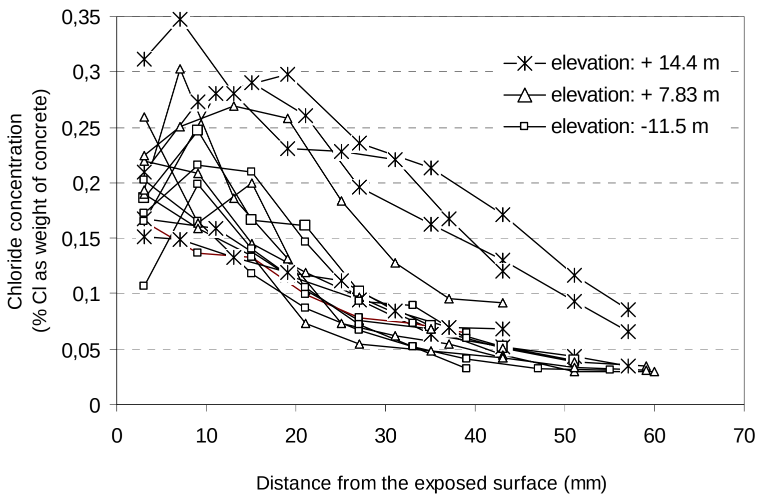

Figure 3-1-3 Chloride penetration into concrete of Brent B Platform (1975) after 20 years of exposure /20/.

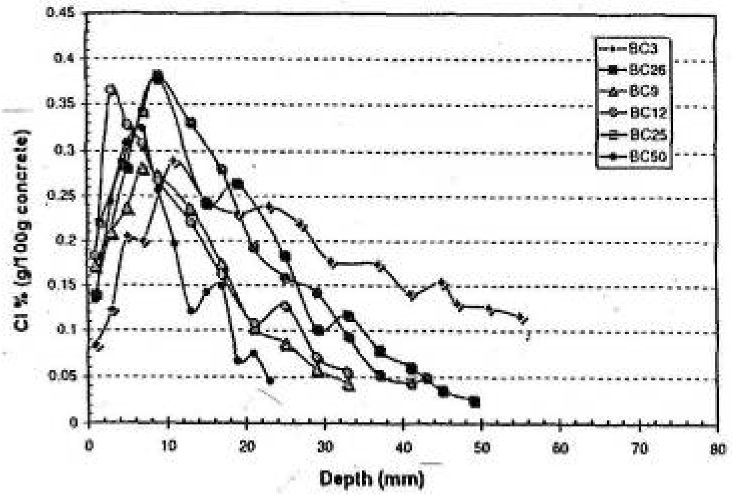

Figure 3-1-4 Chloride penetration into concrete of Brent C Platform (1976) after 18 years of exposure /17/.

For Statfjord A Platform the design specification required that the shafts of the structure should be protected in the splash zone by an epoxy coating. From Figure 3-1-1 it can be seen that such a coating had very efficiently prevented the chlorides from penetrating the concrete, and even after 15 years, this protection still appeared to be very effective /22/. However, for Heidrun Platform, Figure 3-1-5 shows that a much poorer surface coating partly applied to the legs of this platform was not so efficient in keeping the chlorides out, even after an exposure period of only two years.

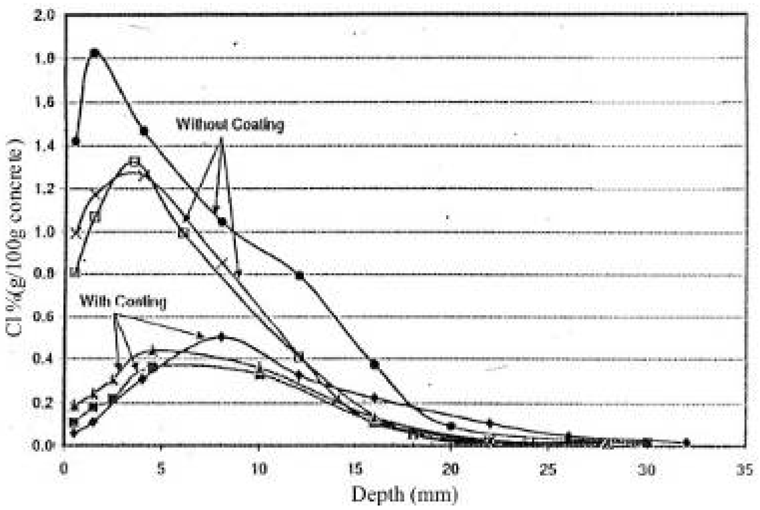

Figure 3-1-5 Effect of concrete coating on chloride penetration into concrete of Heidrun Platform (1995) after 2 years of exposure /17/.

For all concrete structures in the North Sea, a minimum concrete cover of 75 mm for the splash zone was used. Although very strict programs for QA/QC also were implemented during concrete construction, it is a typical feature of all concrete structures that both concrete covers and concrete qualities always show a high scatter and variability /23/. In spite of the limited information available on chloride penetration into the concrete structures in the North Sea, the general conditions appear to be very good.

However, for several of the structures, a certain amount of steel corrosion has already been observed and minor repairs carried out. The above results indicate that the chlorides may penetrate the specified concrete cover in the splash zone within a service period of 25-30 years. For the Brent C Platform (1975), where enough data was available to carry out a probability-based durability analysis, Figure 3.1.6 shows that a risk level of 10 % for steel corrosion is rapidly exceeded after a service period of approximately 20 years.

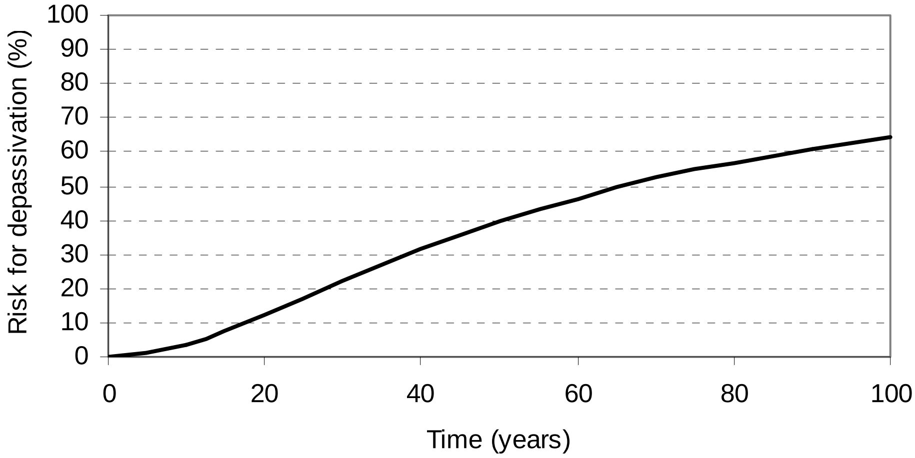

Figure 3-1-6 Development of risk for steel corrosion in the Brent B Platform (1975) /20/.

In addition to the corrosion of embedded steel, the potential corrosion problem for all the freely exposed metallic components attached to the concrete structures has already been pointed out. For majority of the platforms, a relatively high anode consumption at an early stage of operation has been observed, but after some time, the rate of anode consumption has typically been reduced to a more stable value. Current experience with rates of anode consumption appears to vary from one location to another within the same structure and also from one structure to another /24/.

Since different guidelines and recommended practice for cathodic protection of the structures have been used for the design of cathodic protection system, different design values for current drainage to the embedded steel were applied /24/. It is important, therefore, that the inspection schedule and procedure are relevant for a particular structure in order to ensure a close and systematic monitoring of the anode consumption rates.

3.1.4 State of the Art on R&D

For all the concrete structures in the North Sea, the durability specifications were based on prescriptive requirements on the composition of concrete mixes and on execution of concrete work. In order to obtain a more controlled durability and service life of new concrete structures in marine environment, a rapid development of probability-based durability design has recently taken place /25/. Durability design based on these new principles has now become the basis for new recommendations and guidelines for increasing durability of Norwegian concrete harbour structures /26, 27/. These new design procedures include the following elements:

- Probability-based durability analysis
- Evaluation of alternative strategies and protective measures
- Documentation of obtained construction quality and durability
- Preparation of a service manual for regular condition assessment and monitoring of chloride penetration with protective measures for control of this penetration.

For the existing concrete structures in the North Sea, if regular observations on the rates of chloride penetration into the structures are made, the same probability-based procedures can also be applied for a more reliable extrapolation of the further chloride penetration and risk of steel corrosion.

3.1.5 Recommendations

As several of the concrete structures in the North Sea are now approaching the intended service life of 25-30 years, current information indicates that an increasing amount of corrosion on embedded steel may be expected in the years to come. For increased service periods beyond what was originally specified, this may represent a future challenge to the operators. In order to better meet this challenge, a closer following up of the rates of chloride penetration in the splash zone of the structures is recommended. This following up should include regular measurements of chloride penetration in given critical locations of the shafts. Based on such measurements, numerical procedures for a probabilistic extrapolation of the further chloride penetration and a risk of embedded steel corrosion are available. More data on the chloride penetration into concrete is provided the more accurate and reliable such an extrapolation will be /24/.

In order to avoid unnecessary galvanic corrosion problems on the freely exposed metallic components attached to the concrete structures, a proper monitoring of the sacrificial anode systems for these components is of vital importance.

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3.2 Steel

3.2.1 Introduction

DNV has been involved in classification of offshore units, mainly Column Stabilised Units and Jack-ups since the 70'ties and 80'ties, and in the last two decades also ship-shaped units (FPSO's and drilling vessels). Through this classification activity DNV has gained a wide experience related to degradation mechanisms for floating structures built in steel.

Normal design life for classed units is 20 years. The units built in the early 70'ties have now reached an age of approx. 30 years, which is significantly more than they were originally designed for. Therefore, during the last 5-10 years an increase focus has been made to degradation mechanisms relevant for these types of units, and how they can be dealt with in particular for units exceeding the original design life.

3.2.1.1 Main principles of classification

The effect of degradation mechanisms on the safety level of floating structures is closely linked to the principles and survey scheme of classification. Classification is based on a renewal of the class certificate every 5th year, see /1/. This renewal includes a detailed survey, which includes general visual inspection, close visual inspection in way of expected critical details, and non-destructive testing (NDT) according to a pre-defined In-Service Inspection Programme (IIP) prepared by the classification society (DNV). The IIP is updated if experience for similar units indicates problem areas not known and not covered by the IIP.

In addition to the major renewal survey every 5th year, an annual survey is also carried out each year, and intermediate survey in the middle of a 5 year period to maintain the validity of the class certificate.

All together the classification survey scheme has proven to be a good tool to control the most common degradation mechanisms for floating units made of steel. The relevant degradation mechanisms are discussed in more detail below.

3.2.2 History

There have been relatively few major accidents due to degradation of floating units. The most severe accident is Alexander Kielland (1980), which lost one column due to fatigue cracking. The crack started from welded detail in way of a hydrophone (Figure 3-2-1 and Figure 3-2-2). At that time the braces were normally filled with water, which made it impossible to detect any leakage due to the fatigue cracking, and the crack could grow to a critical size, with the following rapture of the brace element.

Other minor incidents have occurred, but in general it is fair to say that the degradation of the units have been monitored and controlled through the inspection programs, and necessary action such as maintenance and repair has been carried out to maintain the overall safety of the units avoiding major accidents.

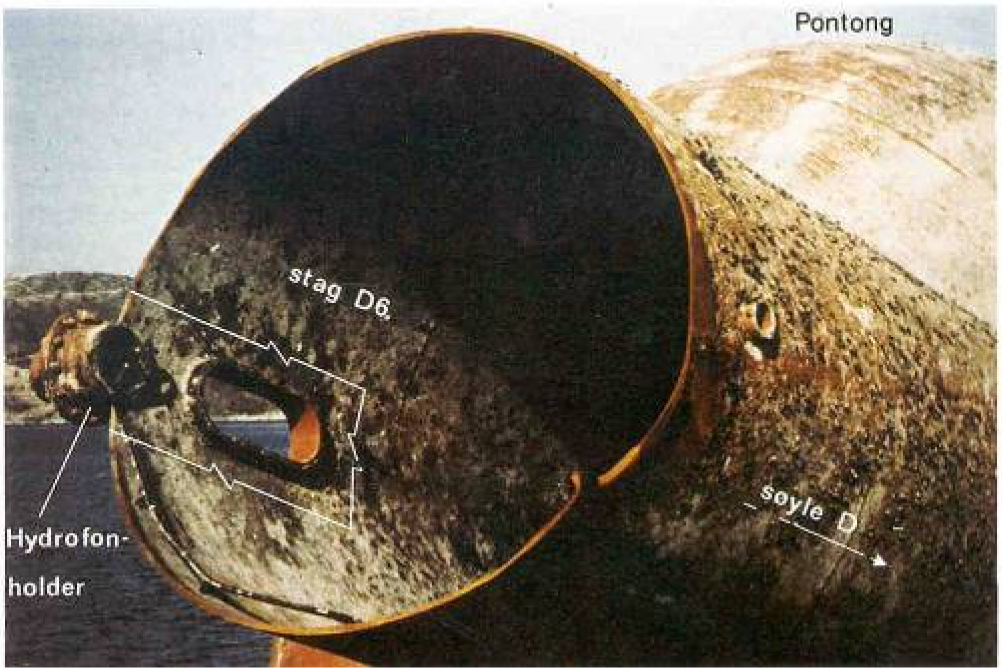

Figure 3-2-1 Alexander L. Kielland – loss of column (Source: The Alexander Kielland Accident. NOU 1981:11) /2/.

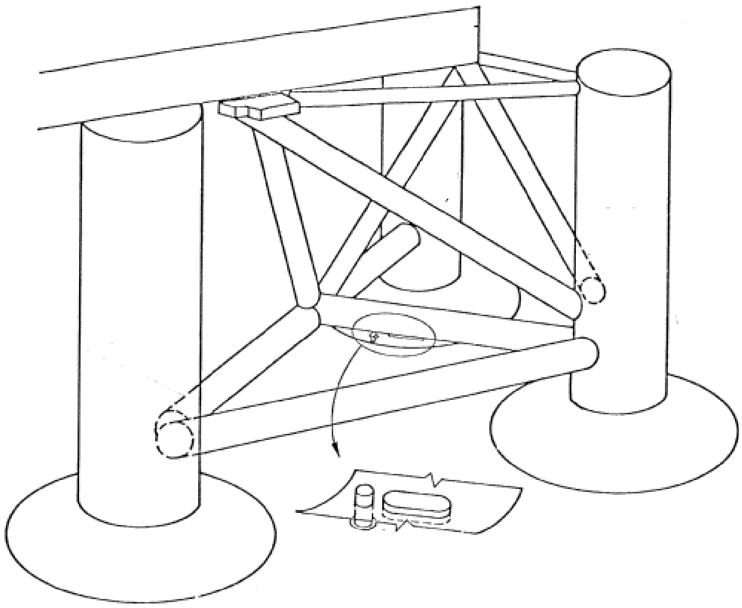

Figure 3-2-2 Alexander L. Kielland – position of crack (Source: The Alexander Kielland Accident. NOU 1981:11) /2/.

3.2.3 Main degradation mechanisms

There are two main degradation mechanisms relevant for the overall integrity of an offshore steel structure:

- Corrosion: uniform/pitting (see Section 2.1.2.3, O_2-corrosion)
- Fatigue (see Section 2.1.3)

3.2.3.1 Corrosion

In general corrosion is a visual degradation mechanism which can be monitored by scheduled inspection e.g. as specified by the classification scheme. Experience from 35 years with classification of floating offshore units is that corrosion is mainly a mechanism which causes local structural damage, but no major reduction in the overall safety level of the units as long as proper inspection schemes and maintenance is provided.

Already from the early 70'ties when the classification activity started, DNV had relatively strict requirements to corrosion protection. DNV introduced specific requirement to the quality of the corrosion protection arrangement. Most common is coating and sacrificial anodes, and to some extent impressed current. Most ship-shaped units are also built with thickness allowance to account for corrosion. Column-stabilised units and jack-ups are normally not built with thickness allowance, since coating is required in all corrosion critical areas.

3.2.3.2 Present situation – Ageing rigs

Due to the Owner's maintenance schemes and also the Rule requirements applied already from the 70'ties, most floating units following these Rules are in general in good condition. The main problem areas have been internal tanks used for trimming of the unit. Such tanks will have a frequency of filling/emptying, which gives good conditions for corrosion (Figure 3-2-3 and Figure 3-2-4). Such areas are given special focus in the class inspections. In general corrosion is not found to be a safety critical degradation mechanism provided a classification in-service inspection programme is followed and proper actions are taken based on the findings. At each renewal survey DNV checks that the rig has an acceptable condition for 5 new years in operation. The acceptance criteria are the same for new and old units and therefore the age of the unit is not important as long as necessary inspection and maintenance is carried out, see /1/.

Figure 3-2-3 General Corrosion inside ballast tank (DNV photo).

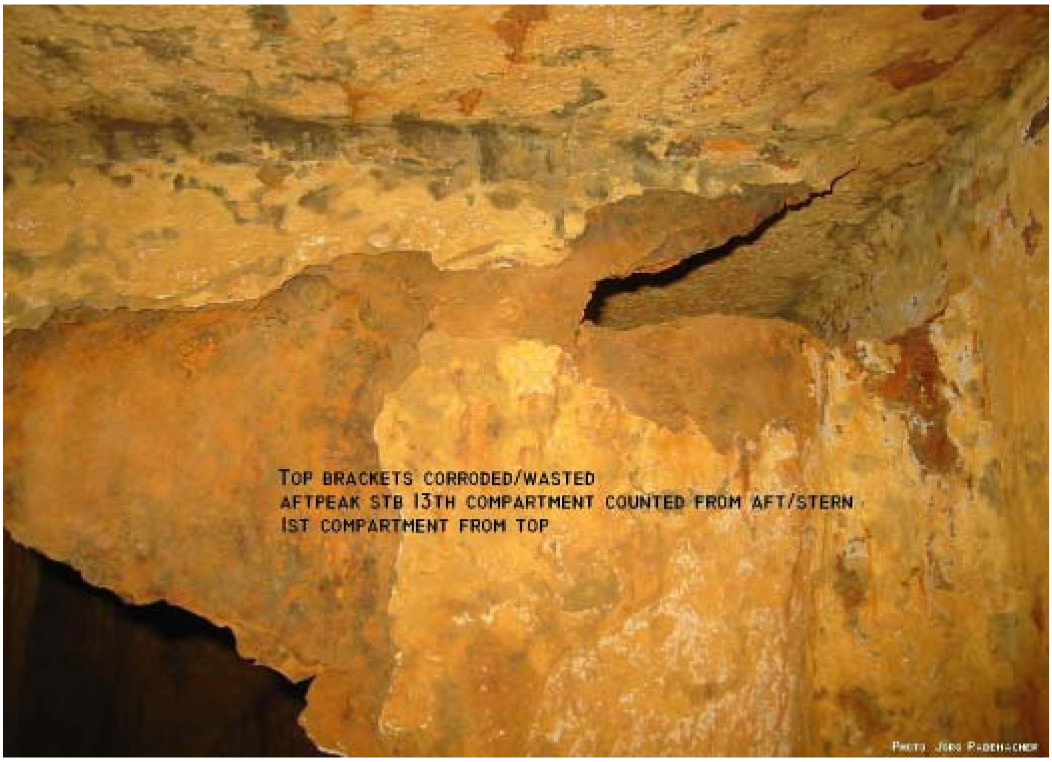

Figure 3-2-4 Heavy Corrosion inside aft peak ballast tank.

3.2.3.3 Fatigue

Fatigue (Section 2.1.3) has been the most important and most focused degradation mechanism for floating offshore structures. There are several challenges related to fatigue as a degradation mechanism and as a design parameter for ageing rigs.

Analysis Methodology

Floating units built 20-30 years ago were designed based on the methodology and design requirement valid at that time. Limited experience was available, and software and computer capacity limited the possibility to carry out detailed analysis for critical areas. Today the computer capacities allow more detailed fatigue analysis to be carried out. Such analysis together with experience (crack history) from operation of these units the last 20-30 years have revealed that the fatigue life was over-estimated for some parts of the units. This means that some details, normally local hot-spots in way of bracket toes etc., have fatigue life less than the required design life when re-calculated according the present methodology.

Load history

Floating mobile units are normally designed to operate world-wide based on the scatter diagram for the North Atlantic, assuming an equal distribution of the wave direction. The approach assumes that the unit is moved around with dominating waves from all direction. In reality some units may stay on one location for a long period with one dominating wave direction. This may cause increased fatigue damage in some areas, while other areas are less loaded.

Fabrication

The fabrication quality of fatigue critical details is of vital importance for the fatigue capacity. Local workmanship and compliance with design drawings are both important. The design calculations are based on a defined quality of the workmanship and also based on structural details as given on design drawings. Poor workmanship (e.g. substandard welding) or wrong details (e.g. details not built in accordance with design drawings) may reduce the actual fatigue capacity significantly. The crack causing the Alexander L. Kielland accident starts from a detail in way of the hydrophone which was not shown on the design drawing.

Rule development

After the Alexander L. Kielland accident the rule requirements were changed. The main changes related to fatigue and consequences of fatigue were as follows:

- all braces (referred to as slender members) shall be watertight and redundant
- water leakage detection to be installed in all braces to detect water leakage due to fatigue cracking in an early stage
- additional damage stability requirement

These new requirement focused on the rigs possibility to survive after an accident / damage, and also to detect a crack propagation as early as possible. Early detection gives time to plan an implement repairs or other compensating measures.

Present situation - Ageing rigs

DNV has introduced a Fatigue Utilisation Index (FUI) as a parameter to measure the “used” fatigue life. The parameter takes into account the number of years in operation and where it has operated.

$$FUI = \frac{\text{Used fatigue life}}{\text{Design fatigue life}}$$

More detailed definition is given in DNV-OSS-101, Ch.3 Sec. 1, I100.

Many floating units have reached their documented fatigue life (FUI>1) and are still in operation. Some years ago DNV and the PSA started to focus on this situation, and DNV introduced some additional Rule requirement for units exceeding the documented fatigue life. These requirements are described in the DNV-OSS-101, Ch.3 Sec. 1 I. The content of this new requirements are as follows:

When a floating offshore unit reach its documented fatigue life one of the following options can be selected:

1. Units with no fatigue cracks during the first 20-30 years of operation can continue to follow the existing inspection schedule. Risk-based inspection methods show that unit with no cracks maintain the safety level also after exceeding the document fatigue life as long as no cracks are detected, see Figure 3-2-5. The methodology is described in OTC 11950 (2000): "Fatigue Reliability of Old Semi-submersibles", /3/.

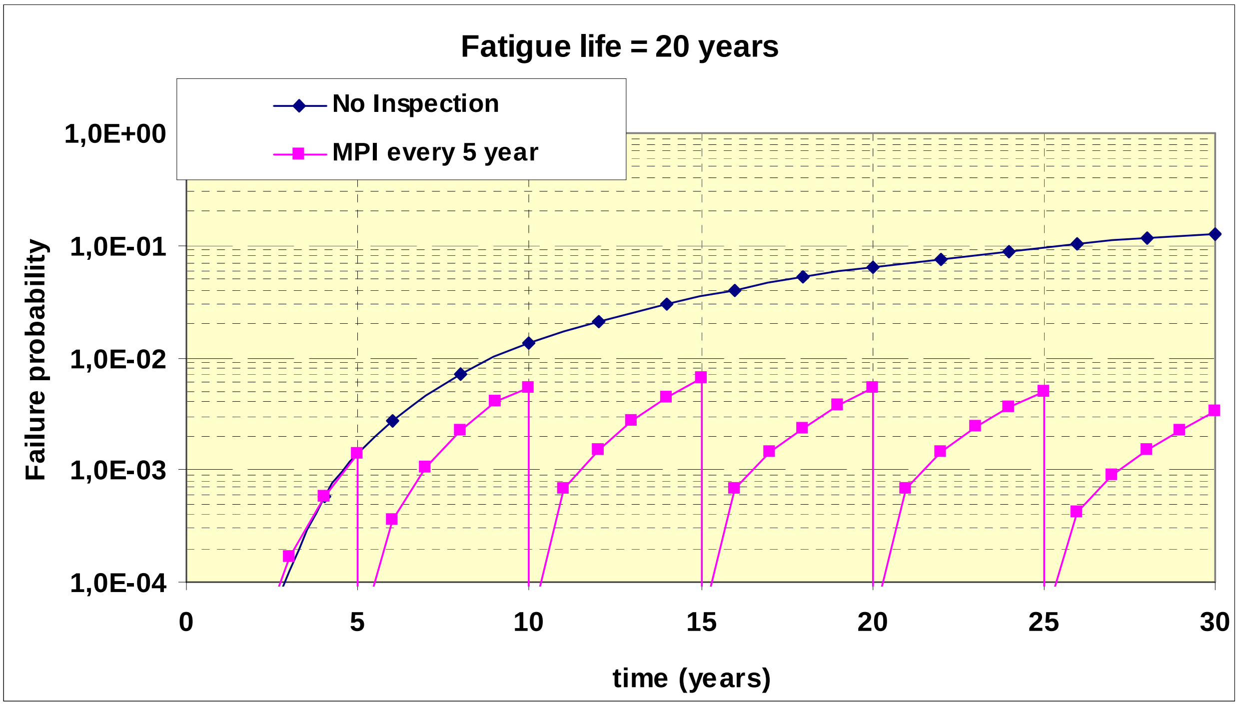

Figure 3-2-5 Failure probability.

2. For units with fatigue cracks during first 20-30 years of operation one of the following steps should be taken, as decided by the Owner of the unit:

Owner shall assess structural details in fatigue critical areas with the purpose of improving the fatigue properties of the structure. The improvement may include, grinding, replacement of steel, modification of details, implementation of risk-based inspection methods considering the detailed information available for each detail, or a combination of these actions. Inspection programmes will be updated to reflect the outcome of these investigations

or

The NDT requirements similar to the extent for renewal survey are carried out on the intermediate survey. This means that the interval between the main NDT inspections is 2.5 years instead of 5 years. This is in line with results found from Risk-base Inspection methods, assuming that most of these units have had some fatigue cracks within the first 20 years of operation.

Similar requirement have been made for column-stabilised units and jack-ups.

The following inspection methods are involved for classed units in operation:

- visual inspection - overall inspection
- close visual inspection – inspection of pre-defined details expected to be critical
- Non-Destructive Testing (NDT)
    - Magnetic Particle Inspection (MPI)
    - Ultrasonic Inspection (UT)
    - Eddy Current (EC)

Scope and extent of inspection and NDT is given in the In-Service Inspection Programs prepared for each unit classed by DNV.

3.3 References

/1/ Rules for Classification of Offshore Drilling and Support Units, DNV-OSS-101, October 2003.

/2/ The Alexander Kielland Accident. NOU 1981:11.

/3/ Fatigue Reliability of Old Semisubmersibles, OTC11950, G. Sigurdsson, I. Lotsberg, T. Myhre and K. Orbeck-Nilssen, Det Norske Veritas, 2000

4 SUBSEA PIPELINES

4.1 Introduction

As pipelines become older, the pipeline operators have several new challenges to consider, such as

- Changes in integrity, e.g. time dependent degradation mechanisms such as corrosion and fatigue (Section 2), or random mechanical damages (e.g. third party damages).
- Changes in infrastructure from the as built, e.g. increased fishing activity or heavier trawler gear.
- Changes in operational conditions, either as a natural change in well-stream condition, tie-in to other pipeline system or increased production rates.
- Required to operate beyond the design lifetime.
- Design no longer valid due to the above mentioned issues.

4.2 History

Generally, review and analysis of historically causes of pipeline failures worldwide /1, 2, 3/, indicate that ***corrosion***, specifically internal corrosion, is the most widely reported cause of failure for offshore pipelines, followed by ***maritime activities*** (e.g. anchor- or trawling- damage and vessel collisions, so called third party damage (TPD)) and then ***natural forces*** (e.g. storms and mudslides).

In the PARLOC 2001-report /1/, which includes a total number of 542 reported pipeline/riser incidents in the North Sea (at the end of 2000), it is emphasised that there is a general opinion that the incident frequencies are highest in the early years of a pipeline's life and towards the end of its life. The former has been attributed to higher vessel activity during the first years of field development and/or early appearance of flaws related to design, material, corrosion inhibition system etc. The latter is more related to changes in infrastructure from the as built, e.g. increased fishing activity or heavier trawler gear and corrosion of the system over time.

In the North Sea, the oldest pipeline is the 36" Ekofisk-Teeside oil export pipeline followed by the 36" Ekofisk-Emden gas export pipeline which came on stream in October 1975 and September 1977, respectively. Both pipelines are made of normalised CMn-steel API 5L X60 (similar to SMYS 415). One of the challenges related to the future operation of the Emden pipeline has been that the pipeline was designed and installed prior to the first issue of the NACE MR-0175. Since the H_2S content is forecasted to increase in the future, and with that the risk for sulphide stress cracking (Section 2.1.2.3) if water is present, it has been of great importance to establish whether the pipeline material is suitable for a gradual "transition" to "sour service" condition or not.

Further, both pipelines are coated with an asphalt enamel coating but without reinforcement, which was not common before in the early 80'ies. These old type of coating has shown tendencies to spall with age. The same problem is not reported for pipelines covered with asphalt enamel including reinforcement or for polypropylene coating (PP). The PARLOC 2001 report does not contain any information about coating types and type of coating damages.

The most commonly used materials for pipeline in the North Sea have been the carbon steel material grades X46, X52, X60 /3/. Later on more high strength steel as X65 and X70, and corrosion resistant steel as duplex/super duplex- and 13%Cr stainless steels, have been utilised as linepipe material. It has also during the last few years been installed quite a few CMn steel pipelines internally lined (mechanical bonding) or clad (metallurgical bonding) with CRA (e.g. AISI 316L, Incoloy 825 and Inconel 625).

4.3 Main degradation mechanisms

Threats to the pipeline system shall be systematically identified, assessed and documented throughout the operational lifetime.

This shall be done for each section along the pipeline. Examples of typical threats are:

- corrosion (internal/external)
- third party damage (TPD)
- erosion
- development of free spans causing fatigue
- buckling

As a result of the natural aging of a pipeline, corrosion and third party damages are considered to constitute the most relevant threats to the system.

Internal Corrosion is the most widely reported cause of failure for subsea pipelines (see Section 4.2). The internal corrosion includes a large variety of corrosion degradation mechanisms depending on the process medium, the material, the process condition etc. Internal corrosion in oil and gas pipelines is principally associated with CMn-steel and the following corrosion mechanisms (Section 2.1.2.3) are of main concern;

- H_2S-cracking (SSC)
- CO_2-corrosion
- Microbiologically Influence Corrosion (MIC)

Liquid water is prerequisite for any electrochemical reaction causing corrosion to occur. Internal corrosion is controlled by material selection (including clad pipe), applying a corrosion allowance (CA) on the inner surface of the CMn-steel pipe or by chemical treatment of the process fluid (e.g. corrosion inhibitor, pH-stabilisation).

External corrosion is controlled by the use of an external corrosion coating in combination with a cathodic protection (CP) system in case of coating damages (Section 2.1.2.2). The control of external corrosion will therefore depend on the type and quality of the external coating and the design of the CP-system.

- External coating**;** Corrosion protection often consists of a tight protective layer around the pipeline exterior. The external protective coating is often asphalt enamel or fusion bonded epoxy (FBE) covered with other types of plastics, as polyethylene or polypropylene, for mechanical protection or as heat insulation. Asphalt can only be utilised together with concrete coating (e.g. weight coating).

- Concrete weight coating; Concrete is applied to the coated pipeline to provide the required compaction and density. The thickness of the concrete ensures both mechanical protection and density for negative buoyancy. Concrete weight coating systems provide the following advantages; sub-sea stability, prevention of damage by e.g. ship's anchors, no trenching and less steel is required.

With the exception of the coating systems used on the oldest pipelines in the North Sea (Section 4.2), no coating damages related to "natural aging" of the coating itself have been reported /4/. The field joint coating (FJC), which are specifically high risk areas for increased bacterial activity and a subsequent increase in the external corrosion, has been a topic for debate. To DNV's knowledge, no incidents have been reported where significant wall thickness loss has been detected.

The quality and properties of modern coatings and the quality control associated with the manufacturing of coating have been considerably improved over the last years. Recommendations for the process of applying specific types of FJC/coating field repair (CFR) and 'infill' systems, and for the process of applying external coating systems for corrosion control of submarine pipelines at the coating plant are, as an example, given in DNV-RP-F102 /5/ and DNV-RP-F106 /6/, respectively. As a consequence of the quality improvement, a considerably less conservative CP-design is necessary compared with recommendations in older CP-design codes (i.e. with respect to coating breakdown factors and hence the net galvanic anode mass to be installed). This is reflected in the newer CP-design codes, DNV RP-F103 (2002) and ISO 15589-2 (2004) /7, 8/.

Most experienced coating damages are related to external impact (TPD). The risk for third party damage (i.e. mechanical damage of the pipeline) is an issue during the entire life of the pipeline, however, as mentioned in Section 4.2, the incident frequencies are highest in the early years of a pipeline's life and towards the end of is life.

All subsea systems (e.g. structures, pipelines, platforms) shall principally be provided with its own CP-system. Interaction in terms of current drain between the pipeline CP system and adjacent subsea installations electrically connected to e.g. platforms or crossing pipelines may cause excessive anode consumption of one of the structures. As the utmost consequence a reduced design life of the CP-system and thereby an under-protection of the pipeline system may occur.

4.4 Failure modes

The main failure modes for pipelines are normally considered to be;

- Leakage
- Burst
- Local Buckling / Collapse

Leakage in pipelines is often associated with the presence of local corrosion attacks (e.g. local CO_2-corrosion, pitting), but might also be a result of small cracks. Burst is more associated with a uniform wall thickness reduction or more extensive crack propagation, decreasing the pressure capacity of the pipeline. Local buckling is a failure mode confined to a short length of the pipe causing gross changes of the cross section; collapse. For subsea pipelines, this is often related to external overpressure in combination with a wall thickness reduction (e.g. as a result of corrosion).

4.5 Ensuring integrity of subsea pipelines

For a given design, corrosion monitoring, corrosion mitigating measures and inspection of the system are fundamental activities to control the integrity of a pipeline system during its design life. As illustrated in Figure 4-1, several activities are necessary to be able to control the integrity of the pipeline system.

To be able to perform an integrity assessment of a pipeline system, the data and results from the activities illustrated in the figure has to be made available. One of the challenges with older pipeline system is that historical data and also original design, fabrication and installation data and reports are lacking. This complicates the possibility of performing a reliable integrity assessment.

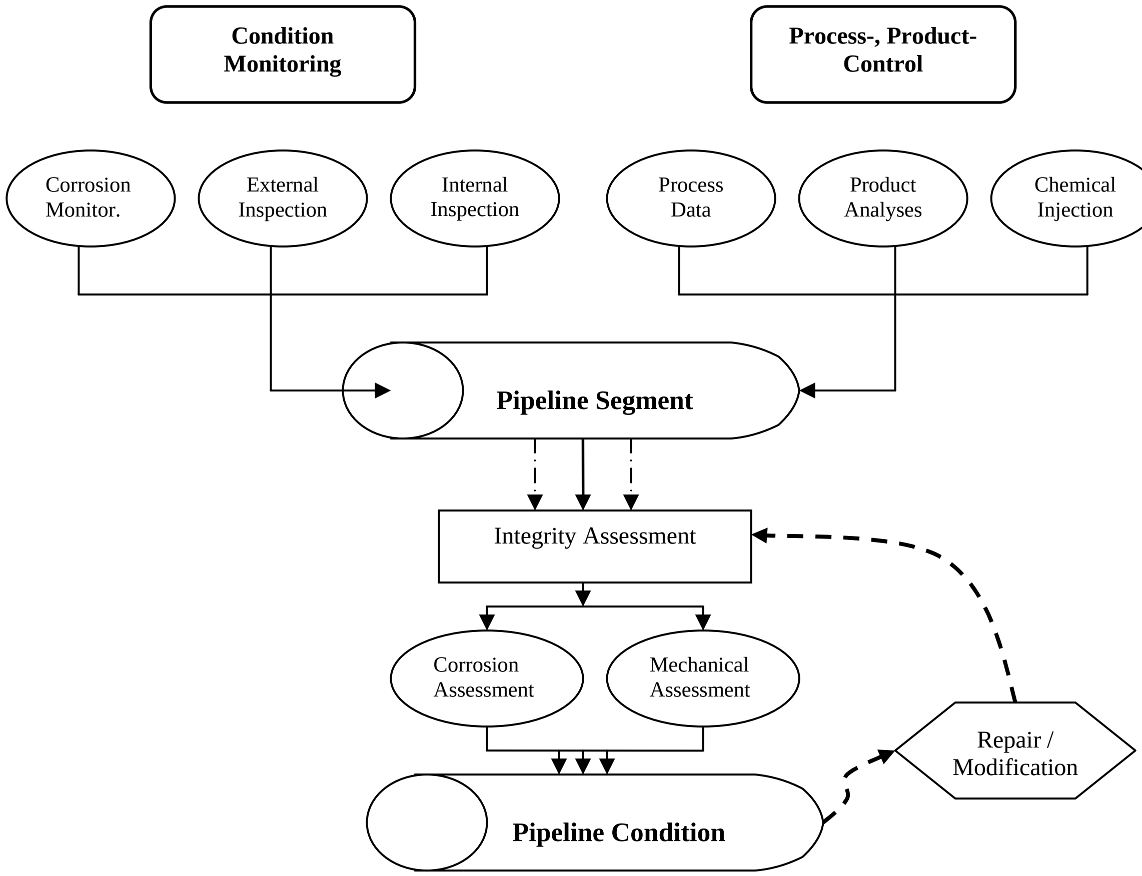

Figure 4-1 Activities necessary to control the integrity of the pipeline system.

A short description of the main activities that constitute the pipeline integrity management system is given below. More detailed requirements to managing system integrity are given in industry standards as API 1160, ASMEB31.8S and DNV-OS-F101 /9, 10, 11/.

4.5.1 Process- and product control

Process- and product-control includes the following;

- Process control (pressure, temperature, flow rate etc)
- Product sampling (CO_2, H_2S, O_2, water cut/dew point, sand etc)
- Chemical injection for corrosion prevention (corrosion inhibitors, pH-stabilisation etc.)

The process- and product control shall ensure that the condition in the pipeline is within the operational window as defined in design. If e.g. a souring of the well stream occurs, increasing the H_2S-content and with that exceeding the maximum specified limit as given in design, this information should be handed over to the responsible for pipeline integrity. For gas lines, the dew point should be monitored and for liquid lines, the water cut should be known. Further, for chemical injection, the availability (or efficiency) given in design should be known together with the precautions taken to ensure the chosen availability. As an example, a 95% inhibitor availability requires, according to NORSOK M-001, that a qualified inhibitor is injected from day one and that a corrosion management system is in place to actively monitor corrosion and inhibitor injection. Any redundancy in the system should be elucidated ensuring e.g. continuous injection of corrosion inhibitors even though one pump fails and thereby maintaining the required availability.

4.5.2 Corrosion Monitoring

The rate of corrosion dictates how long any process equipment can be safely operated. When applying corrosion monitoring techniques it is vital that the equipment is installed in locations where corrosion might occur (e.g. lowest points where liquid might accumulate in confined areas). Otherwise, the data received from such measurements will have no relevance when assessing the integrity of the system.

Typical corrosion monitoring techniques are;

- Corrosion coupons / ER-probes / LPR probes
- Sampling
- Field Signature Method - corrosion monitoring
- DDL - Deposition Data Logger
- UT of fixed spots
- Sand/Erosion

Off the techniques listed above, corrosion coupons, ER- and LPR-probes together with sampling form the core of industrial monitoring systems. The other techniques are normally found in specialised applications.
However, it should be emphasised that corrosion coupons and probes are not suitable for documenting the prospective corrosion in a pipeline but to monitor any changes in the fluid corrosivity.

4.5.3 External Inspections

Typical external inspection methods are;

- Visual Inspection (GVI/CVI) performed by divers
- Inspection performed by using Remote Operated Vehicles (ROV); Video, Sonar, Transponders, Profilers, etc.
- External Ultrasonic Testing and Thickness Measurement for verification of metal loss or cracks

During such inspections the following is typically inspected:

- The CP system - looking for excessive consumption of the anode mass.
- Indication of inadequate coverage of buried or rock dumped pipeline sections
- Visual inspection of anode consumption
- Recording of anode potential and steel potential if practically
- Field gradients at anodes
- Coating or concrete damages
- General damage to pipelines from impact (dropped object, equipment handling, anchor impact or dragging, fishing, etc.)
- Flanges and hubs – looking for leaks
- Looking for upheaval buckling or snaking

For buried pipelines, as-laid surveys along the entire length of the pipeline have to be performed prior to backfilling (buried). Significant damages to the coating and sacrificial anodes shall be recorded and the consequences for long-term performance considered. When the pipelines are buried, no further inspections of the coating or anodes are possible.

4.5.4 Internal Inspection

In-line inspections (ILI) are normally performed to verify the internal surface condition of the pipeline system but are also, depending on the chosen tool, capable of verifying the external condition with respect to corrosion and cracking.

Several types of tools for internal inspection, cleaning and batching are available on the market. An overview of types of tools associated with in-line inspection (ILI) and cleaning/batching of pipeline-systems is given in Figure 4-1.

Additionally, ILI might be performed to monitor the efficiency of the corrosion protection (e.g. external coating) and the prevention systems (i.e. corrosion inhibitor, dew-point, water cut etc). ILI is the only method that brings high confidence with respect to e.g. inhibitor availability (and efficiency) by verifying the actual condition of the pipeline internal surface.

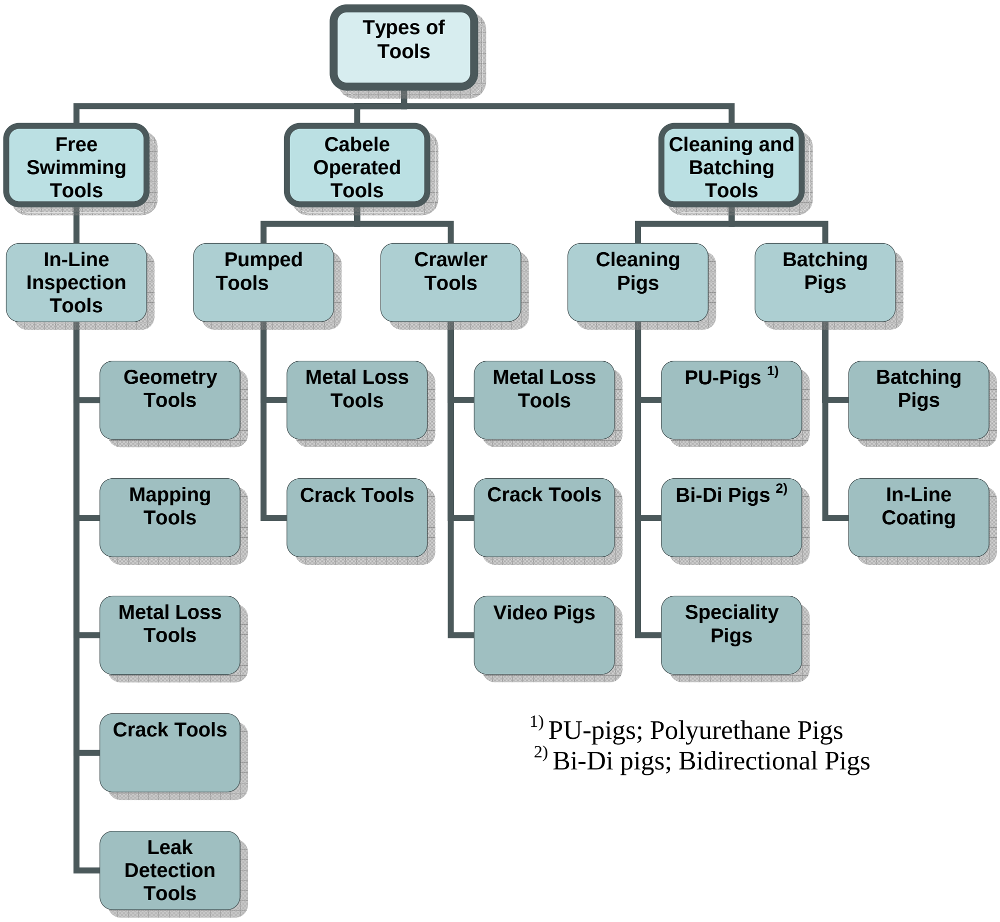

Figure 4-1 Overview of different types of internal inspection and cleaning/batching tools (source: NDT Systems & Services AG).

4.5.5 Hydrostatic pressure testing

Pressure testing is an industry-accepted method for validating the integrity of the pipeline. The pipeline is tested up to a load of approximately 1.1 x operating pressure to 95% SMYS. The pressure test is normally performed as a combined strength and leak test. Upon completion of pressure testing, the pipeline should be properly cleaned, de-watered and dried to avoid future corrosion in the system. Experience shows that local corrosion in pipeline systems is often attributed to water leftover from the pressure test.

4.6 "Corrosion Zones" associated with external corrosion.

External surfaces of subsea pipeline systems may be divided into "corrosion zones", based on the environmental parameters that determine the actual corrosivity. The physical characteristics of the corrosion zones further determine the applicable techniques for corrosion monitoring and inspection.

The following major zones may apply;

Atmospheric Zone (Offshore): For pipelines on an exporting platform and on riser platforms, a "marine atmospheric zone" will apply from the pig launcher and to the upper limit of the riser's splash zone. The unmitigated corrosion rate of carbon steel in a marine atmospheric zone is in the range 0.1 to 0.3 mm/yr at ambient temperature but may become significantly higher (0.3-1 mm/yr) for hot surfaces directly exposed to sea spray. In areas sheltered from sea spray, the unmitigated corrosion rate will approach the lower limit of the range (i.e. 0.1 mm/yr).

Closed compartments with humidity control are referred to as a "dry atmospheric zone".

Corrosion protection means:	Coating and corrosion allowance (CA)
Inspection method:	Visual inspection and in-line inspection (pigging)

Marine Splash Zone (Offshore): The marine splash zone can be defined as the area of e.g. a riser that is periodically in and out of the water by the influence of waves and tides. The actual length of the "splash zone" varies based on the actual geographical location and, moreover, from one splash zone definition to another. Extremely corrosive conditions may prevail for hot risers just above the water level; 3-10 mm/yr has been reported for risers with a wall temperature of about 100 ^{o}C. In the splash zone below the water level, the unmitigated corrosion rate is close to that in the seawater submerged zone (see below).

Corrosion protection means:	Upper zone: Coating and corrosion allowance (CA) Lower zone: : Coating, corrosion allowance (CA) and CP-system
Inspection method:	Upper zone: Visual inspection and in-line inspection (pigging) Lower zone: Visual inspection, in-line inspection (pigging) and CP-monitoring

Seawater Submerged Zone; The corrosivity of the seawater submerged zone is relatively low and unmitigated carbon-steel corrosion rates in excess of 0.1 mm/yr would only be expected for surfaces heated by an internal fluid. For ambient temperature surfaces, the unmitigated corrosion rate is below 0.1 mm/yr, although slightly higher values may apply for local pitting attacks.

Corrosion protection means:	Coating, and CP-system
Inspection method:	In-line inspection (pigging) and CP-monitoring

Offshore Buried Zone; In general, the corrosion rate in marine sediments is very low (<< 0.1 mm/yr) but high local corrosion rates (of the order of 1 mm/yr) may apply in the uppermost sediments if the bacterial activity is high. Internal heating of the pipe surface to 20 to 50 ^{o}C increases bacterial activity and hence the potential for microbiologically induced corrosion. (at higher temperatures, bacterial corrosion is inhibited). Pipeline sections covered by rock dumping, gravel and by other means are normally defined as "buried".

Corrosion protection means: Coating, and CP-system

Inspection method: In-line inspection (pigging)

4.7 Technology Status

The available technology for monitoring and inspection of pipelines have increased compared to the 70'ies and pipelines that were previously difficult or even impossible to inspect, may now be accessible. The accuracy of equipment (as MFL, UT, product measuring devises etc) has also been improved over the last years which brings a higher degree of confidence on the monitored data and, with that, a more reliable assessment of the pipeline condition (i.e. integrity assessment). An accuracy of ±10% (at 80% confidence) and ±0.5 mm (at 90% confidence) is typically reported for MFL and UT pigs, respectively.

The definition of "un-piggable" lines is given as lines that are not designed for allowing standard inspection tools to pass through, which basically requires a more or less constant bore, sufficient long radius bends and traps to launch and received pigs. Today, the inspection equipment can be tailor-made in order to overcome the situation that was previously considered "un-piggable" with respect to standard tools.

Further, since some incidents in the past can be related to lack of industry experience (as for the 13Cr stainless steel, HISC incidents associated primarily with the CP-system), the increased experience following such events will normally decrease the likelihood for similar events to occur again.

Advanced pipeline repair and rehabilitations products and services (covering steel, plastics and epoxy composite), as the Pipeline Repair System (PRS) /12/, have been developed, together with more sophisticated and robust programs for analysis and assessment of pipeline condition, allowing repair and further use of damaged pipelines.

More detailed requirements to managing system integrity are given in industry standards as API 1160, ASMEB31.8S and DNV-OS-F101.

4.8 References

/1/ PARLOC 2001 “The update of loss of containment data for offshore pipelines”, M.MacDonald Ltd for The Health and Safety Executive, The UK Offshore Operators Association and the Institute of Petroleum, July 2003

/2/ “Improving the Safety of Marine Pipelines”, Committee on the Safety of Marine Pipelines (USA), 1994

/3/ “Pipeline damages – Damages and Incidents from Petroleum Safety Authority’s CODAM database”, PSA Norway, August 2006

/4/ “Nedbryting av rørledninger over tid”, Rev.02, Det Kongelige Olje- og Energidepartement, 03.09.1998

/5/ DNV RP-F102 “Pipeline Field Joint Coating and Field Repair of Linepipe Coating”, October 2003

/6/ DNV RP-F106 “Factory Applied External Pipeline Coatings for Corrosion Control”, October 2003

/7/ DNV RP-F103 ”Cathodic Protection of Submarine Pipelines by Galvanic Anodes”, October 2003

/8/ ISO 15589-2 ”Petroleum and Natural Gas Industries – Cathodic Protection of Pipeline Transportation Systems; Part 2; Offshore Pipelines”, 2004

/9/ API Standard 1160, “Managing System Integrity for Hazardous Liquid Pipelines”, 2001

/10/ ASME B31.8S, “Managing System Integrity of Gas Pipelines”, 2001

/11/ DNV OS-F101, “Submarine Pipeline System”, 2000

/12/ DNV report No.2005-3394 “Joining Methods-Technology Summaries”, Rev.01, 12.10.2005

5 SUBSEA EQUIPMENT

5.1 Introduction

Subsea equipment will in this section be template, manifold and subsea Christmas tree (XT) system. The Subsea wellhead system is covered in Section 6, Drilling and wells.

5.2 Template

Templates are normally manufactured of low alloy carbon steel and protected against corrosion by a CP system. For the Norwegian continental shelf, the templates are integrated units with manifold and external fishing gear protection for protection of wellhead systems and the manifold.

The template is exposed to forces that develop between the wellhead foundation, wellhead thermal growth/well temperature extension and pipeline forces. The template is also seeing the gravity forces from the manifold, which the template serves as foundation for. Forces generated from wellhead growth and pipeline can over the time of operation be changing, but definitely not in a number of cycles that causes fatigue to be a problem. However, forces experienced from trawl gear might have severe effect of the function to template structure. Due to continuous development of fishing gear, both with respect to shape and mass, can effect of those be outside original design criteria of the template. Detail to template design such as handles, hatches, indicators etc. can be a potential snag point leading to sudden impact loads that might give local defects.

5.2.1 Workmanship

The templates are welded structures, normally manufactured in low alloy carbon steel. The template is being exposed to a set of different load scenarios that might have the extreme load condition during transportation/installation compared to in service condition. The building quality of the template will depend on the quality of the steel, welds, coating, electrical continuity straps and, if depth dictates, punctured members. They are all falling in under quality assurance that can be part of manufacturing procedures and Factory Acceptance Test (FAT). The overall quality and ability to sustain any future condition of changed loads is up to what quality the template is designed for. The latter element, i.e. all future load conditions, might be hard to detect upfront.

5.2.2 Degradation mechanism

The template, as a low alloy carbon steel structure shall be protected against corrosion through cathodic protection assisted with high quality coating. Normally epoxy based paint according to NORSOK is used. The template design also often includes anodes to protect wellhead and the current drainage through the well shall be accounted for in CP design. Failure mechanism with respect to corrosion would therefore be related to:

1) Lack of electric continuity to all protected parts.
2) Coating not intact leading to excessive anode consumption.
3) Inadequate anode design and/or quality; anodes may fall off after some time in operation etc.

Degradation mechanism to the structural integrity to the template can be related to unfavourable effect of load combinations and operation outside original design criteria. This can be explained by change in foundation due to sea bed erosion, new tie-ins or subsequent rock dumping outside what original designed for. Severe global failure of a template structure would be by external applied forces which can be: BOP impact loads, conductor string installation or external forces created by impact with equipment from non oil and gas production equipment, such as anchors and fishing gear.

5.2.3 Inspection programme

Inspection programme should typically cover following items:

- The CP system – looking for excessive consumption of anode mass and damaged/missing anodes.
- Recording of anode potential and steel if practically possible.
- Coating damages, both due to general degradation and as effect of contact with fishing gear.
- General damage to structure from fishing gear.
- Hatches, handles and other elements that serve a function or can generate a snag point.
- Inspect earth cabling used to ensure electrical continuity.

5.3 Manifold

For the Norwegian sector of the North Sea the manifold is an integral part of the template and it can be a retrievable unit from the template. The manifolds today are often manufactured from prefabricated pipes and components in 22Cr duplex materials. According to pressure requirements and requirements to flexible pipe design elements can also be manufactured of 25Cr Super Duplex materials. The mechanical behaviour and mechanism of these materials are as described in Section 2.1.2.2 External corrosion (i.e. HISC).

After incidents, both on the Norwegian continental shelf as other places, failure in components of this material have led to a number of activities in order to get an understanding of failure mechanism as well as developing design guidelines (DNV RP F-112 /2/) to avoid this failure in the future. Current recommendation of usage of 22Cr Duplex and 25Cr Super Duplex materials have today a more conservative approach with respect to allowable working stress level in the material. The design guidelines today includes routines and checks to allowable stress level that is not covered by former industry practice to pipe stress and component calculations.

5.3.1 Workmanship

The manifold is a safety critical element and involves a number of mechanical components (valves, connectors, and sensors) that can fail. These components and the manifold piping itself are manufactured from advanced materials. The manifold piping are normally with welded connections and the valve bodies are welded in the manifold piping. Sensors tend to be bolted by approved flange styles. Welding is a critical process both with respect to onsite pressure testing as well as in service condition. Welds gives changes to material properties and residual stresses are inherently present at welds (post weld heat treatment will reduce the residual stresses, if carried out). Workmanship and control here is important both to welding qualification and non destructive testing, and the components should be designed to cater for latest development in design rules for HISC sensitive materials. The overall quality plans for manufacturing and testing must follow good practice, and testing shall be conducted according to the correct sequences. It is of importance that industry practice of pressure test of components and welds are done prior to coating. Piping insulation shall be according to verified and accepted procedures as well as being quality checked against these.

5.3.2 Degradation mechanism

Degradation mechanism for the manifold can be summarized to material aspects as well as functional aspects. The manifold piping is manufactured of different materials with different corrosion resistance. Typical material selection can be Duplex (22Cr ferritic-austenitic steel), Super Duplex (25Cr ferritic-austenitic steel), 6Mo for process bore piping, whilst control system piping can be of the less noble 316 (austenitic steel).

With respect to the external environment, all piping will be subjected to CP, not only for its own protection, but also to protect other components as valves, actuators, connectors and instruments. Also the latter components can be of different materials, and should in general through engineering processes, be made of materials that have robustness to HISC.

If the process piping is insulated to reduce cooling effect from ambient seawater, the insulation material shall be sufficiently resistant to degradation when submerged in seawater. Degradation can be a combination of absorbed seawater, with elevated temperature due to hot produced fluids. Degradation of insulation can also be mechanical damage. This will be from operation and contact with foreign objects, or it can be due to deflection and strain absorbed by pipe material which causes the insulation material to crack. Due care shall be paid so unwanted effect of damage to coating in combination with reduced CP. This condition can lead to accelerated degradation to the metallic pipe material when seawater is exposed to bare material and the CP potential is not sufficient.

Valves, valve actuators and instruments are all components that will be manufactured from different materials, where major bodies and shells can be made from low alloy carbon steel. It is therefore of importance that these components are electrically connected so CP protection is ensured. When degradation is discussed, this will also involve moving components. Especially override mechanisms and sliding stems can loose some of its intended purpose due to effects of degradation to material, marine growth, and calcareous deposits. This can lead to increased friction and wear and subsequent leakages to sea from e.g. actuator housings.

Other components that are of importance are clamps and connectors. Careful design with suitable material selection should normally avoid potential problems.

With respect to the internal environment, internal corrosion and erosion are considered the most relevant failure mechanisms.

5.3.3 Inspection programme

Manifold inspection shall cover following checks:

- General condition to pipe coating
- General condition to components, valves, actuators and instruments
- Inspect for leakages at valves and components
- Inspect all connector to trees and pipelines
- Pressure test
- Visual inspection to detect foreign objects

5.4 Subsea Christmas tree (XT)

Subsea XT components are generally manufactured from low alloy carbon steel. Wetted surfaces to produced fluids are normally with corrosion resistant alloy (Inconel 625 etc). The tree itself is a construction with a number of pressure containing components bolted together with flange type connections. It is of importance that these components are assembled together after formalized procedures to safeguard quality.
The externals of the tree are protected against corrosion by a CP system together with a coating, normally an epoxy based coating. It is of importance that the metallic material which is used is not exceeding strength and hardness requirements for the industry. Also, as mentioned, since the tree consists of several components bolted together it is of importance that all components are electrically connected to ensure cathodic protection.
Internal corrosion is as mentioned in Section 2.1.2.3 and should within normal use not be of concern. However, the internals of the tree can be exposed to well stimulation chemicals, especially after interventions that can be of aggressive nature, even when exposed to the CRA.
Some barriers may contain non metallic seals. It is reasonable to believe that all such seals could not be documented at time of design to actual working design life.

5.4.1 Workmanship

The subsea XT are a critical safety barrier as well as it is probably the most advanced product that is permanently installed. The XTs are built of advanced components and there is a mixture of low alloy steel components as well as high alloy stainless steel materials. Common for both are that they are sensitive to thermal effects from welding so it is important that all heat treatment and welding activities are done after qualified and approved procedures. In contradiction to the manifold all components on a XT are connected by bolted connections, or by a flange type connection. Compared to a welded pressure container there is no formal certification of how to assemble a bolted connection. It is therefore of importance that a proper good bolt torque procedures exist and are understood by the personnel that is doing the job. The XT are during its FAT extensively tested, but long term effect to the assembly will not be discovered, either in qualification test or in FAT. I.e. there is a discussion in the industry to whether the tree is experiencing vibration. However, regardless of vibration or its potential frequency, it is important that slim members, such as small bore piping are supported sufficiently such that long term effect of vibration does not lead to breakage.

5.4.2 Degradation mechanism

Degradation of the tree is dependant to what it is subjected to in service. Normal service should be accounted for in detail design, both with respect to exposure to produced fluids, injection chemicals as well as external exposure to seawater with its parameters. Such parameters can be external temperature, location, and seawater depth. Degradation can also be divided into material degradation and functionality to components.

For the tree, the major building blocks are made from low alloy carbon steel with relatively high strength. Those components are in most cases coated by an epoxy based coating. For some projects the sensitivity to flow assurance is based on strict requirement to temperature insulation. If this is the case the entire tree block should be insulated, and not only limited to the external process piping. Degradation of coating or insulation can lead to reduced availability in production. High temperature to epoxy coated surfaces can lead to excessive degradation of coating. This shall however be accounted for in the system CP design. For temperature insulation-material care shall be taken to seawater absorption, mechanical wear from impact of external components, and loss of binding between insulation and steel material. The latter can lead to severe local corrosion if the combination is unfavourable wrt seawater access and lack of CP effect.
High strength material, both of carbon steel as well as sophisticated stainless qualities, shall be selected with sufficient margin to avoid HISC, and the operation stress level shall be below certain values.
Valves, actuators, connectors and instruments are manufactured from a mixture of different metallic materials. The large shells and bodies are usually manufactured from low alloy carbon steel, and smaller components are likely to have some stainless type material, e.g. Duplex and Super Duplex. Override mechanisms and override stems that extend from actuator housing to external seawater atmosphere can be a potential degradation to functionality. E.g. marine growth, calcareous deposits can lead to failure of the sealing element between actuators and seawater allowing seawater into areas not tolerant to this. Also severe friction can, due to mentioned effects, lead to problems in operating e.g. valves or other mechanisms.

The internals of the tree shall in normal service not be of concern. Through material selection the tree shall be robust. However, it seems to be situations were the well has been treated with chemicals than can have severe effect to tree internals when not flushed out satisfactory. These chemicals (which can be acid) can remain in dead end pockets, in seal grooves etc. and cause severe local damage. Effect to soft seals can be unknown.

5.4.3 Inspection program

Inspection of Christmas tree shall cover following checks:

- The CP system – looking for excessive consumption of anode mass
- Recording of anode potential and steel if practically
- Coating damages, both due to general degradation and as effect of hot surfaces
- General damage to structure from fishing gear
- General condition to pipe coating
- Inspect for leakages at valves, connectors, sensors and components
- Pressure test
- Visual inspection to detect foreign objects

5.5 References

/1/ DNV-RP-F112 Design of duplex stainless steel subsea equipment exposed to cathodic protection, Draft issue April 2006

6 DRILLING AND WELLS

6.1 Introduction

As drilling equipment ages, the operator has several new challenges to consider, such as:

- Changes in integrity, e.g. time dependent modes such as corrosion and fatigue, or random modes such as mechanical wear.
- Accidental events (incl. well settlements)
- Requirements to operate beyond the design lifetime.
- Design and Approval no longer valid due to the above mentioned issues.

6.2 History

Generally, review and analysis of historically causes of drilling equipment failures worldwide, indicate that ***mechanical wear***, is the most widely reported cause of failure for offshore drilling equipment and wells, followed by corrosion and fatigue damage.

6.3 Main degradation mechanisms

Threats shall be systematically identified, assessed and documented throughout the operational lifetime. This shall be done for each component and for the system. Examples of typical threats are:

- mechanical wear
- corrosion
- Erosion
- buckling
- fatigue

6.4 Drilling system and well system

A typical schematic layout of the drilling equipment is shown in Figure 6-1.

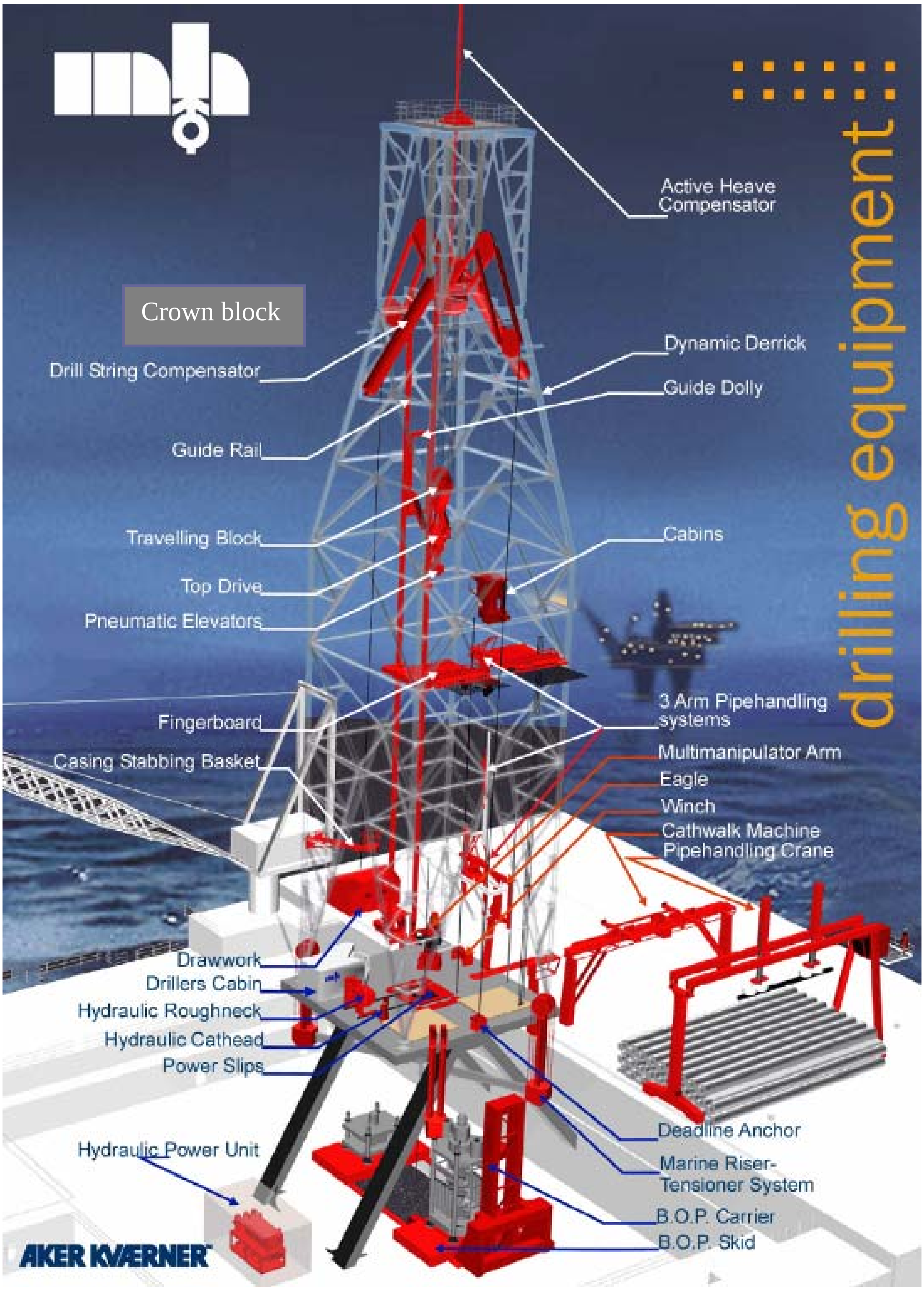

Figure 6-1 Typical schematic layout, drilling equipment (Aker Kværner).

6.4.1 Pipe handling

The pipe handling system often experiences mechanical wear due to frequent use. Components experiencing such wear should be inspected periodically. Furthermore, impacts and vibration during pipe handling might lead to loss of pretension in bolted connections. Loss of pretension increases the probability for fatigue damage in the connections. The pretension of the bolted connections should be checked periodically during the design life.

6.4.2 Main hoisting system

The brake capacity of the brakes on the draw work might get reduced due to oil contamination on the discs. An important aspect of draw work brakes is the friction between the brake pads and the discs. The friction factor used in the brake design calculations is to DNV's experience often non-conservative.

The main brake is an electric motor, and the driller training using the brake system is often not satisfactory, which leads to potential hazardous situations. This also includes that some drilling rigs uses the emergency brake on a daily basis in the drilling operation, which reduces the safety level in the main hoisting system.

If the pads are not "run-in" properly, a reduced brake capacity will be experienced. This may not be noticed during normal operations, but may lead to an accident in an emergency braking operation. The disc springs of the callipers are subjected to low-cycle fatigue which reduces the brake capacity after 1-2 years of operation. The fabrication tolerances of the callipers are sometimes too inaccurate, which give a higher airgap for the disc brakes and following loss of brake effect on one of the sides.

The travelling block/crown blocks are often experiencing mechanical wear in sheave grooves during their life time.

The safety level of the draw work and the crown block should be at the same level. The crown block safety level is less than the draw work, which has lead to hazardous situations during accidental events where details in the crown block have collapsed.

In general, the development of the drilling operations goes toward higher hook loads, resulting in a need for corresponding increased brake capacity.

6.4.3 Control systems

Experience shows that Control systems sometimes contain programming errors, leading to logical errors in the system and planned operations fail.

After a few years in operation, whole or parts of a control system might be upgraded. The quality control of the new control system is often not as extensive as when the drilling system was initially designed, manufactured and tested. Experience shows that such upgrading sometimes lead to unwanted events. Safety assessments when upgrading control systems should be increased to prevent uncontrolled situations/operations. In the draw work, several control systems from different vendors are often not sufficiently correlated. This aspect gives a source for failure during operation. To prevent this source for failure, one of the vendors should be given the main responsibility with respect to total quality assurance, documentation and interface aspects.

6.4.4 Iron Roughneck and pipe racking system

The iron roughneck as well as the pipe racking system experiences extensive mechanical wear, and are often replaced/upgraded after 4-5 years.

6.4.5 Steel wire ropes

Steel wire ropes are subject to continuous wear and fatigue loading. On drawwork and riser tension system there is implemented a ton*mileage measuring device. After a specific ton*mileage the cut and slip is carried out. It is normal procedure to send part of the used wire to a test laboratory in order to calibrate the acceptance level for cut& slip. Worn steel wire ropes may be reduced in diameter due to roll out and fatigue cracks may be introduced.

If steel wire rope are subject to high temperature from for example flaring the wire grease may be lost which lead to more rapid degradation of the wire than expected.

Damage to wire, caused by kinking, running over sharp edges or due to bad spooling on winches are common causes of replacement for wires in winches.

Winches which lack spooling devices may experience incorrect spooling. This effect gives uneven contact on the wire and leads to higher wear on the wire.

6.4.6 Sealed machinery

In general, we have experienced that closed sealed machinery sometimes are not properly sealed, and contamination is able to enter and cause extensive mechanical wear on the machinery. This is hidden errors which are not found during external visual inspection. Typical examples: splines and bearings.

6.4.7 Drill string

The drill string often experience fatigue cracking due to ageing and poor control with the number of load cycles. The handling and use of the drill string give small damages which lead to corrosion, which accelerates fatigue (i.e. corrosion fatigue as described in Section 2.1.4) and thereby give a fatigue weakness and damage. High differential pressures combined with eroding environment might lead to wash-out of the drill string as shown in Figure 6-2.

Figure 6-2 Typical example wash-out of drill string (DNV photo).

6.4.8 Drilling riser

The drilling riser is often designed with a smaller thickness amendment due to corrosion than other risers and structures, but should be subject to a higher control regime. Reduced wall thickness due to corrosion and erosion leads to a reduced tension capacity. Through the riser management system on the rig, there should be a system to rotate the position of the riser elements periodically to be in the splash zone as well as in the highest loaded positions in turn. There should also be a load record of the riser stack giving an overview of the loading of each component.

6.5 Blow Out Preventer (BOP)

The gaskets of the BOP are often subject to a continuous mechanical wear during the drilling operation as well as periodic testing of the BOP. In some kinds of BOPs, the shear ram can wear out after only approx. 15 runs. In such cases, it is of vital importance that the operator has exact control of the history of the BOP, and that the critical gaskets are replaced before leaks occur. Degradation and damage of some of the BOP gaskets can increase significantly due to special operations such as drilling through casing where steel swarf from the drilling operation passes the BOP. The feasibility of the gaskets is periodically controlled by pressure tests demonstrating that the functionality of the BOP is maintained.

6.6 Subsea Wellhead

The Subsea wellhead is normally manufactured from relatively high strength low alloy CMn steel. Sealing surfaces are normally inlay welded with CRA. The wellhead are welded to a casing pup piece, often manufactured by API 5L grade materials. Current analysis shows that the intended design and mechanical behaviour are dependant of the level and quality of cement fill between 20” casing and 30” conductor. The different mechanical behaviour is of particular importance for the wellheads capacity to take riser fatigue loads. Current design load as defined by NORSOK U-001 are not sufficient to describe the capacity to a wellhead system. Recent work by DNV also indicates that the wellhead does not provide the same level of conservatism as the common industry standard for completion/work over risers.
The potential damage to Subsea wellheads are related to drilling or workover mode when the wellheads are subjected to a riser load. The failure mode is related to fatigue, both in welds as well as at stress risers in base material. As described in Section 2.1.3 in this document, a fatigue failure will have an initiation stage with slow propagation before reaching a limit where remaining cross section is overloaded and a rapid failure develops. The wellheads are normally not accessible for inspection, and hence it is difficult to detect cracks that are in initiation stage.
Recent assessments of fatigue lifetime show that the wellheads are utilised at a level exceeding already used time with riser exposure. This is particularly important when assessments of old wellhead systems are done with respect to IOR programmes which may lead to extended riser exposure.

6.6.1 Workmanship

The Subsea wellheads are manufactured of relatively high strength CMn steel. As the wellheads can show short fatigue life, it is important that they are designed with profiles that give low stress concentration factors. All welding and heat treatment must be done according to approved procedures and subsequently inspected by NDT. The fatigue resistance of welds is very dependant of the weld configuration. A weld that is grinded after welding will have a better fatigue life compared to a non machined weld. However, it is of importance that the weld between 18 ¾” wellhead housing to 20” casing weld gives limitation to final surface treatment after welding. This is due to restricted access for personnel, due to long assembly and small internal diameter.

The cement job in the annuli between 20” and 30” casings is difficult to check. Cement will be contaminated as well as it is of low strength type with low viscosity. During curing it will shrink and give less support. Thermal expansion due to temperature variations will also decrease the supporting effect from the cement. This can be unfavourable with respect to the fatigue life of the 20” casing and its welds.

6.7 Conductors

The conductors are normally manufactured from low alloy carbon steel and are dependant of cathodic protection against external corrosion. As mentioned above for Subsea wellheads, the conductors are also welded to nominal thickness API 5 L grade steel materials. This weld together with stress risers from changes in cross sectional area are all elements that can give reduced fatigue life capacity to the wellhead system. Fatigue loads are experienced by the conductor when it is subjected to completion/workover risers- as well as to a drilling riser set up. The conductor capacity is dependant of quality and support externally provided by grouting and strength capacity to surrounding sea bed. The conductor capacity is also influenced by lateral support from a template system. In other words, a satellite conductor without support from a template structure is exposed to higher external loads. This leads to lower static capacity and shorter fatigue life.

As mentioned above for Subsea wellheads, the conductors' mechanical behaviour are not always as expected through design. It is a combined unit with wellhead and they are having a mechanical interaction that depends on factors mentioned above, such as external grouting and internal cement quality in the annuli between 20" and 30".

For combinations of the factors above, the wellhead/conductor capacities to riser fatigue load can be relatively small. In some cases they are calculated to as low as shorter than a normal duration of an intervention campaign. Therefore the wellhead and conductor show lower conservatism than what is acceptable for a riser set-up.

It is important to include such evaluations into the process when IOR programmes or other programmes are discussed that lead to extended riser exposure to existing wellhead system.

The conductors are easier to inspect compared to the wellhead housing. However, the most severe failure mode, fatigue might be impossible to inspect for. Also due to most critical weld might be in an area where external inspection is impossible due to external frames, and internally due to all casings that is installed.

6.7.1 Workmanship

The conductor is constructed of mild steel. It is not pressure containing so it is only seeing mechanical load. Robustness to fatigue life is very dependant on the quality of the weld. Compared to wellhead housings, the weld between conductor housing and conductor casing is possible to both grind and inspect. However, it is not certain that this have been accounted for at the time of design and manufacture to old systems. External grouting and possible fracturing of the seabed is factors that can lead to reduced capacity to both static bending loads as well as fatigue load. A template installed conductor is normally better in those respects than satellite wells.

6.8 Production casing in wells

Depending on the well behaviour, the Production casing and other components in the completion string will be subject to corrosion and erosion. For wells with a high degree of sand production, erosion (see Section 2.1.1) will be a relevant degradation mechanism. For wells subject to gas lift, frequent start/stop, water production or varying injection of water/gas, corrosion will be the main degradation mechanism. Geotechnical scenarios like settlements, dislocations, etc., can introduce additional shear and compression loads, and should be taken into account in the design. Detection of damaged production casing is primarily performed by pressure surveillance, and there are a number of quality control and surveillance methods of production casing with respect to thickness measuring.

6.9 High pressure tubing and manifolds for circulation of drill mud

In high pressure systems, erosion will be the main degradation mechanism. Especially in choke- and kill manifolds during well control situations, very large erosion rates will appear due to the mixture of gas and drilling slurry. Also in high-pressure piping handling drill mud and cement, erosion is the main degradation mechanism, especially in bends and branches.

6.10 Recertification of well control equipment

The content of this section is mainly extracted from ref. /1/. It is DNV's interpretation of the PSA regulations that a major overhaul/inspection with verification of Blow Out Preventers and other pressure control equipment used for Drilling, Completion and Workover operations, should be performed at least every five years.

In addition, the need to recertify equipment can occur due to several other causes:

- Change of intended use / loading aspects
- Increasing original design life
- Repair of equipment

The purpose of this inspection is to verify and document that the equipment condition and properties are within the original acceptance criteria.

The extent of inspection may be influenced on the following parameters:

- Repair history
- Maintenance history
- Operational history
- Manufacturers guidelines
- Change in rules and regulations or company's governing documents

The following activities shall be included in the recertification process:

- Review of original documentation with special focus on traceability.
- Review of maintenance history/records, to verify the amount of use and extent of maintenance
- Stripping/dismantling of equipment
- Visual inspection.
- NDT
- Dimensional check of selected components/review of dimensional check reports.
- Change out of seals, treads etc.
- Reassembly – recoating - preservation
- Load/pressure testing and functional testing.

The acceptance criteria for the various inspections performed shall be based on the manufacturer's initial qualification programs and engineering documentation, as well as internationally recognized codes and standards.

DNV does not recommend recertifying equipment unless the acceptance criteria applied gives a certain confidence with regards to margins to failure. It must be possible to verify that the functional, performance and safety margins of the equipment are within the original acceptance criteria.

6.11 References

/1/ DNV-OTG-06 Recertification of well control equipment – service description, September 2005

7 MOORING SYSTEM

7.1 Introduction

A Mooring Integrity JIP carried out by Noble Denton Europe /1/ has concluded as follows:

- The interface between the surface vessel and the mooring line requires particular attention for all types of FPS.
- Carefully planned innovative[1] inspection, making use of all possible tools, has been demonstrated to be able to detect problems relatively early before they become a potential source of failure.
- At present no in-water techniques exist to check for possible fatigue cracks.
- On two North Sea FPSs chain wear and corrosion have been found to be significantly higher than what is specified by most mooring design codes. This wear seems to be more pronounced on less heavily loaded leeward lines compared to the more loaded windward lines.
- At present there is little data available which indicates how the break strength of long term deployed mooring components will be reduced by wear, corrosion including pitting and the possible development of small fatigue cracks.
- A possible contributory mechanism for the relatively high line failure rate among drilling semi-submersibles has been identified. This is believed to be due to rigs thinking they have set up balanced pre-tensions, when in fact this has not been achieved. One reason can be that the load cells on the windlasses are not calibrated properly. If the tension meters are well positioned, working properly and their calibration is in date, a likely cause of unbalanced line tensions is partial seizure of the gypsy wheels. This can be confirmed by a simple line Payout/Pull-In test. If this reveals that some of the gypsy wheels are partially seized, an attempt should be made to free them up. However, if the unit is on station it may not be feasible to undertake such work in situ. In such a case the line tensions out with the fairlead should be determined by other measures such as:
    - o ROV or possibly diver monitoring of the chain angles where they emerge from the fairleads
    - o Acoustic monitoring of the x, y and z positions of specific connectors on the mooring lines

  From these measurements it is possible to back calculate the actual line tensions as long as this is done in calm conditions with minimal tidal variations.

7.2 Fairleads and chain stoppers

Typical problems with fairleads are malfunction of bearings, excessive wear and tear in fairlead wheels and pockets due to insufficient support for chain. This may be caused by low tension and/or the fairlead is not rotating with the chain, which may be caused by bearing problems. Excessive wear and tear has been discovered in cases were the chain is terminated in chain stoppers underneath the unit.

[1] The mooring systems of mobile offshore units are inspected onshore within a period of 5 years. For units permanently installed at a location the inspection has to be carried out offshore and it is important to use all possible available tools. In situ water inspection techniques are continuing to improve, but further developments are needed to provide dimensional data on links all around the inter-grip area and to improve the marine growth cleaning off speed. For further information see /1/.

7.3 Chain

Historically, chain manufactured in the 1980's, especially Grade 4 chain suffered with quality control problems and subsequent brittle fracture problems. Brittle fracture is the term used for a rapid failure of material, which involves low ductility. It is partly dependent on the type of steel used, or the processing that it has been subject to, and is exacerbated by stress-raisers or cracks in the material. It can result in the failure of chains at relatively low tensions. The problem with brittle fracture is that the propensity of the material to fail in this way is not obvious to the naked eye and can only be quantified by destructive testing. Steel is more susceptible to brittle fracture when the yield strength is high or where the operational temperatures are low. The welding process, and subsequent heat treatment, used to form the chain link during manufacture must be very carefully controlled to prevent brittle fracture problems with chain. This said the metallurgical and manufacturing issues appear to have been largely resolved such that modern high strength chain can now be consistently produced.

Much of this problematic chain has now been removed or scrapped but the associated problems have had an impact on the industry over the years /2/. It is also possible that there is a residual amount of this chain around.

The prime cause of line failure now appears to be with the connecting shackles or with links that have been mechanically damaged. Common modes of failure in chain systems therefore include:

- Mechanical damage to links
- Missing or loose studs
- Failure of connecting links
- Brittle fracture of links (not so common with improved quality control of chain)
- Fatigue

Missing or loose studs has no directly influence on the breaking strength, however the stress distribution in the link is changed and the footprint will represent an area where fatigue cracks can develop and result in fatigue failure of the link. Control with stud pressing is essential and instances have been seen where studs have been pressed without having been correctly seated in the imprints. Also, studs have been expanded by excessive amounts with detrimental effect on the links. Ideally, stud pressing should result in light contact with the link. There is no limitation regarding how many times a stud can be pressed.

The following tolerances regarding studs apply /8/:

- Axial stud movement up to 1 mm is acceptable.
- Axial stud movement greater than 2 mm is unacceptable.
- Links are to be removed or studs are to be pressed using an approved procedure.
- Acceptance of axial stud movement from 1 to 2 mm must be evaluated based on the environmental conditions of the unit's location and expected period of time before the chain is again available for inspection.
- Lateral movement up to 4 mm is acceptable provided there is no realistic prospect of the stud falling out.
- Welding of studs is not acceptable.

In Figure 7-1 the line failure was caused by fatigue. The fatigue initiation has caused the chain link fracture and growth has been probably due to overloading of the chain link. Most likely the overloading has been probably due to twisting of the chain link, in addition to high cyclic axial tension loading of the mooring chain.

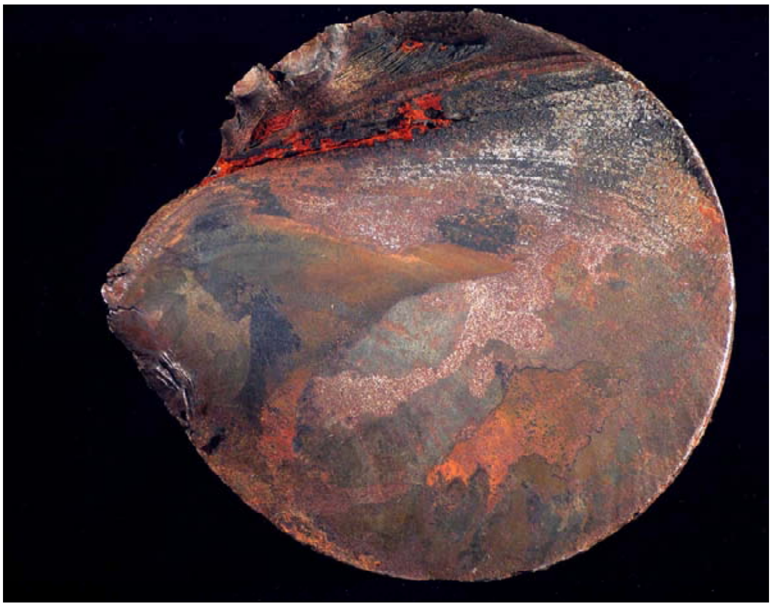

Figure 7-1 Fatigue (DNV Photo).

The chain links shown in Figure 7-2 have suffered bacterial corrosion along one of the straight sides. The corrosion rate was estimated to be of 2.5 mm/year at the contact area between chain and seabed based on this examination. Even though the damage was confined to the straight sides of the chain links (areas away from those with the highest potential stresses), a corrosion rate of this order can obviously affect life of a mooring system that is designed to be in operation during a 20 year period. In Figure 7-3 a crack was detected, which was initiated in the foot print area.

Figure 7-2 Bacterial corrosion (DNV classed FPSO offshore West Africa).

Figure 7-3 Crack initiated in the stud foot print (DNV photo).

Figure 7-4 shows severe wear and tear in a long term mooring system.

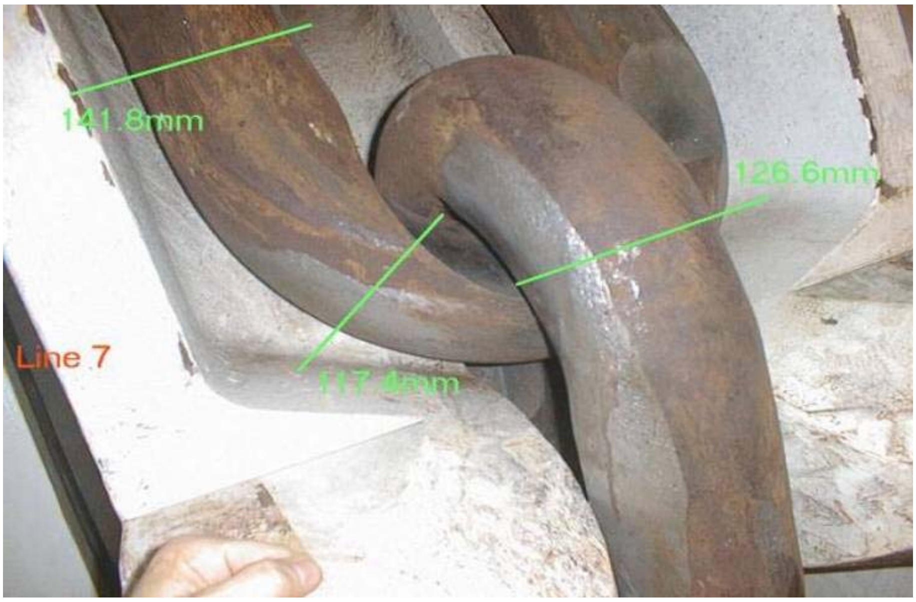

Figure 7-4 Wear and tear in chain links (Photo Norsk Hydro).

7.4 Recertification of chain

Normally recertified chain is not accepted in long term mooring. However, for drilling units it is more common to rent used chain if the unit's own mooring system is not sufficient for a new location. Recertification of chain shall be carried out applying the same inspection requirement as for a complete periodical survey, which include visual examination, extensive non-destructive testing, dimension control and pressing of studs.

A recertified chain for mobile offshore units shall pass the requirements for renewal survey given in DNV Instruction to surveyor /8/. It is generally not possible to state that a recertified chain is as good as a new equivalent chain. However, the recertified chain is found good enough for 5 year in operation, since a renewal survey is required every 5 years. Recertified chain is normally not accepted for permanent installations.

7.5 Synthetic fibre ropes

Typical failure mechanisms are:

- Ingress of particles
- Mechanical damage during installation and hook up activities
- Mechanical damage caused be fishing activities
- Creep

Ingress of particles such as sand or clay into the load bearing part of the fibre rope will reduce the breaking strength of the fibre rope significantly. This problem can be avoided by installing a filter underneath the outer jacket of the fibre ropes. DNV has qualified such filters for Marlow Ropes /3/ and ScanRope /4/.

Mechanical damage during installation, hook up and ROV operations must be avoided. Further, trawling has caused failure of fibre ropes. DNV has developed a recommended practice /5/ to assess the rest capacity of a damaged fibre rope.
The purpose of this recommended practice is to provide assessment basis for the suitability of a polyester mooring rope to remain in service, after it has been mechanically damaged by external objects. The recommended practice is applicable to any "parallel-subrope" type of rope. The inputs required to perform the necessary calculations are provided by the rope manufacturer. This information is given in the Manufacturer's Report. The damage assessment is based on the subrope-to-rope relationship, since the subrope is the primary building block of the rope. Subrope-to-rope assessment is required since the effect of damage is highly dependent on the damage distribution. This implies that for a given loss of area, the resulting rope strength and fatigue performance will vary depending on the distribution of the damage.

Synthetic ropes have become an accepted alternative to chain and steel wire rope mooring lines in recent years. At present, polyester fibre is the most widely used synthetic fibre for this purpose. High modulus polyethylene (HMPE) is an alternative to polyester, with many favourable properties. However, HMPE is more susceptible to creep than polyester, and this behaviour requires careful assessment as part of the design process for HMPE mooring lines.
Creep is a form of permanent elongation of synthetic fibres. The creep rate increases with increasing specific load and temperature. Creep can ultimately lead to failure of a mooring line. Creep need not be a serious problem if it is properly accounted for in the design of a mooring line.

7.6 Steel wire ropes

There are three types of steel wire ropes used in mooring systems (see Figure 7-5) with different expected service life:

- Six and multi strand, normally used by mobile offshore units.
- Spiral strand with and without plastic sheathing, normally used by permanent installed units.
- Half and fully locked coil with or without plastic sheathing, normally used by permanent installed units.

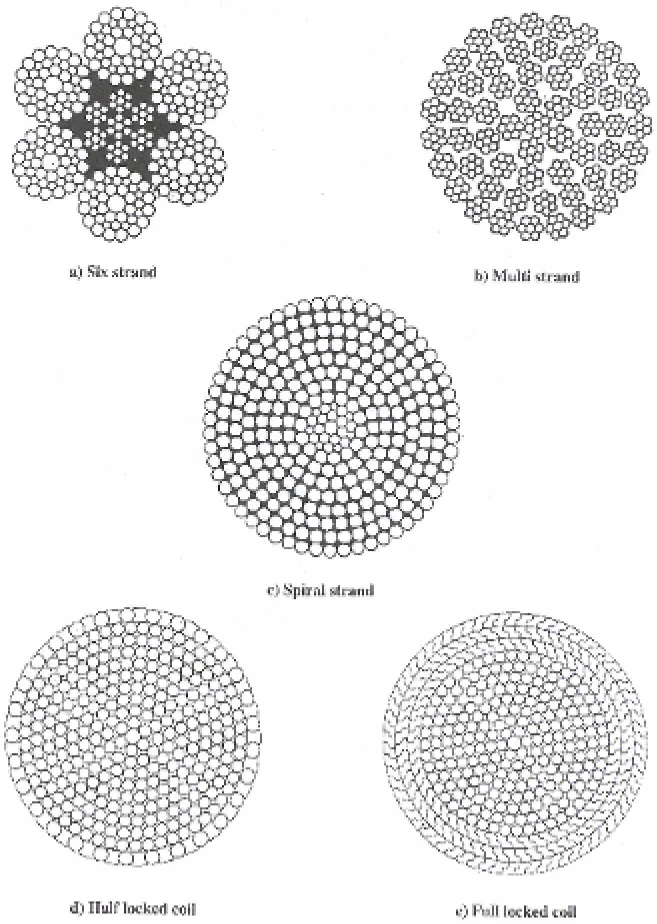

Figure 7-5 Constructions of Steel wire ropes /6/.

Failure of wire in mooring lines is caused by one of three causes:

- Mechanical damage to the wire
- Corrosion/Wear
- Fatigue
- Chasing operations can also cause bends and kinks in mooring wires.
- Bends may not be serious enough to replace the wire rope; however, kinks will seriously reduce strength.

With respect to design life the following table from /6, 7/ can be used as a rough guideline:

Choice of steel wire rope construction		
Field design life (years)	*Possibilities for replacement of wire rope segments*	
	Yes	No
< 8	A/B/C	A/B/C
8 – 15	A/B/C	A/B
> 15	A/B	A
A) Half locked coil/full locked coil/spiral strand with plastic sheathing B) Half locked coil/full locked coil/spiral strand without plastic sheathing C) Six strand/multi strand		

A common design requirement is that wire rope segments in mooring lines are to be protected against corrosion attacks throughout the design life. The wire rope is therefore assumed to be fully protected such that its fatigue life approaches that in air. This is normally ensured by the following measures or combinations thereof:

- Sacrificial coating of individual wires.
- Application of blocking compound on each layer of the strand during stranding. The compound should fill all crevices in the wire rope, strongly adhere to individual wire surfaces and have good lubrication properties.
- Cathodic protection by spinning zinc or other sacrificial anode alloy wires in one of the outer 3 layers of wire rope during manufacture.
- Surface sheathing of the wire rope by an extruded plastic jacket in order to prevent ingress of sea water and flushing out the blocking compound.

The ends of each wire rope segment are normally terminated with sockets. Free bending at the sockets outlet can reduce the wire rope fatigue life. To avoid premature fatigue failure, a bend limiting device is often incorporated at these locations. Such a device is designed to smoothly transfer the loads from the sockets to the rope. To prevent ingress of water in the socket a sealing system may be incorporated in the device.

7.7 References

/1/ Floating production system - JIP FPS mooring integrity. Prepared by Noble Denton Europe Limited for the Health and Safety Executive 2006

/2/ P. J. Donaldson, B.Sc. Master Mariner, MRINA M. Brown B.Sc., M.Sc., M.B.A., C.Eng., MRINA M. Pithie, Master Mariner: “Design and integrity management of mobile installation moorings”. Noble Denton Europe Ltd, 2004

/3/ DNV Report No. BGN-R3100270 “Test of Soil Filter”, July 2000

/4/ DNV Report No. 2005-3502 “Testing of Soil Filter”

/5/ DNV-RP-E304 “Damage Assessment of Fibre Ropes for Offshore Mooring”

/6/ DNV Certification Note No.2.5 “Certification of Offshore Mooring Steel Wire Ropes, May 1995

/7/ DNV-OS-E301 “Position Mooring”, October 2004

/8/ DNV Instruction to surveyors for Classification of Mobile Offshore Units, I-C3.4 Mooring System

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APPENDIX

A
SUMMARY

SUMMARY			
Area	**Relevant constructions, systems or equipment on installations**	**Relevant degradation mechanisms**	**Typical failure modes**
Load bearing structures, concrete	- All concrete on the structure including the skirt below seabed.	- Chemical seawater attack - Freezing and thawing - Expansive alkali reactions - Bacterial corrosion - Chloride penetration of the concrete - Galvanic corrosion of the reinforcement steel	- Cracking - Fracture
Load bearing structures, steel	- Floating offshore unit - Jack-up rig - Column-stabilised units	- Corrosion, fatigue	- Local structural damage, fatigue cracking
Subsea pipelines	- Service/process conditions - Free spans - Coating - Asphalt enamel coating - Weight coating of pipeline - Atmospheric zone, marine splash zone, seawater submerged zone offshore buried zone	- Corrosion (internal) - Sulphide stress cracking - Fatigue - Third party damage (TPD) - Spalling - Spalling - Corrosion (internal/external)	- Leakage - Burst - Local buckling/collapse - Reduced wall thickness
Subsea equipment	- Template - Manifold - Subsea XT system	- Third party damage (fishing gear) - External corrosion - External/internal corrosion - HISC - Mechanical damage to coating - Erosion - Wear - External corrosion - HISC	- Coating damage - Cracking - Reduced wall thickness

Drilling and wells	- Pipe handling system - Main hoisting system - Iron roughneck and pipe racking system - Steel wire rope - Sealed machinery - Drill string - Drilling riser - BOP - Subsea wellhead - Conductors - Production casing in wells - High pressure tubing and manifolds for circulation of drill mud	- Mechanical wear - Fatigue - Low cycle fatigue - Mechanical wear - Mechanical wear - Fatigue - Wear - Wear - Fatigue - Corrosion fatigue - Corrosion - Erosion - Mechanical wear - Fatigue - Fatigue - Corrosion - Erosion - Erosion	- Fatigue cracking - Reduced wall thickness
Mooring system	- Fairleads and chain stoppers - Chain - Synthetic fibre ropes - Steel wire ropes	- Wear and tear - Bacterial corrosion - Mechanical damage to links - Wear and tear - Brittle material (welding related) - Fatigue - Ingress of particles - Mechanical damage during installation or due to fishing activity - Creep - Mechanical damage to wire - Corrosion/Wear - Fatigue	- Failure of mooring line - Reduced wall thickness - Fatigue cracking

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POLITICO

ENERGY & ENVIRONMENT

California's latest power grid problems are just the beginning

State officials knew ahead of the recent heat wave that the grid was on shaky ground.

Residents on the Central Coast had tussled for decades over the role of Diablo Canyon, but until recently, they thought the debate was over. | Michael A. Mariant/AP Photo

By **CAMILLE VON KAENEL**
09/23/2022 04:30 AM EDT

SACRAMENTO, Calif. — An epic heat wave this summer brought California's power grid to the brink of collapse, and put its governor on defense as he touted the state's nation-leading climate goals.

In Democratic Gov. Gavin Newsom's telling, the state kept the lights on because of its efforts to bolster renewable energy.

"Went right up to the edge of breaking our grid, but it didn't," Newsom said at a Clinton Global Initiative event this week, describing this month's scorcher to dignitaries gathered in New York City for Climate Week at the U.N. "This transition worked."

The reality, however, is a lot messier.

California's recent decisions to postpone the closure of its last nuclear plant and to extend the life of some natural gas-fired facilities highlight what officials and experts say is the fact that the state with the most ambitious energy goals is far from achieving them.

Growing demand for electricity and the fickle nature, for now, of greener technologies such as wind and solar are making it hard to progress toward the state-mandated goal of a grid that's 100 percent emissions-free by 2045. Renewables provided 36 percent of the state's power supply on average so far this year.

Those constraints were behind the recent decision by the Legislature, at Newsom's urging, to postpone the retirement of the Diablo Canyon nuclear plant despite the fact that activists thought they'd secured its closure — and the governor himself once supported the idea.

Those constraints were behind the recent decision by the Legislature, at Gavin Newsom's urging, to postpone the retirement of the Diablo Canyon nuclear plant despite the fact that activists thought they'd secured its closure — and the governor himself once supported the idea. | Michael A. Mariant/AP Photo

The 10 days of triple-digit temperatures across the state this month sent power demand surging to a record level, bringing state regulators close to ordering rolling blackouts, a potentially deadly move and a political disaster.

It was the realization of a nightmare scenario a top state energy official said he's been considering for months.

"Oh, my lord, we are in a very bad situation compared to even the worst case that we anticipated," Siva Gunda, vice chair of the California Energy Commission, said he recalls thinking in the spring, when supply chain delays and a tariff on solar imports — compounded by severe drought — started to look like a multi-year power crisis.

The possibility of rolling blackouts became a shadow looming over California Democrats, even those who felt uneasy about keeping Diablo Canyon open. Some talked publicly about how outages contributed to the impeachment of then-Gov. Gray Davis at the beginning of the century.

What's needed now, officials say, is even more investment by the state akin to the Marshall Plan that rebuilt Europe after World War II.

"Enough isn't being done right now" to avoid a worrying gap in the power supply in the future, said state Sen. John Laird, a Democrat from Santa Cruz who has argued the state needs massive new investment in renewable energy and batteries to move off fossil fuels.

"We have to make sure that we have more wind, we have more solar, we really develop offshore wind, get out of the way of some of the developed renewables so that they come on the grid," Laird said.

The stakes are existential. After the Enron electricity trading scandal and the Western energy crisis that followed 20 years ago, the state's reputation was in tatters and a governor got recalled. But California quickly built itself up as a model for embracing wind and solar power — becoming the measure against which other states compare their own climate ambitions.

Now, by a state estimate, California will need to deploy renewable energy at five times its average pace to meet its mandated goal of 100 percent emissions-free power by 2045. All that while contending with rising temperatures, drought and wildfire.

That uncomfortable reality gave Laird some sleepless nights, he recalled in an interview. He is close friends with people who fought for decades to shut the nuclear plant, worried there might be an accident along the seismically active Central Coast, among other concerns.

He eventually voted to keep Diablo Canyon open, a difficult decision he said was driven by projections that California would not have enough new wind and solar power in time to make up for its closure. The last-minute scramble by Newsom and the Legislature could postpone the plant's demise until 2030, reversing a deal made six years ago between green groups, labor and regulators to close it in 2025. The nuclear plant provides up to ten percent of the state's power.

Laird sees the delay, which requires federal approval, as a stopgap measure that shouldn't get in the way of a massive build-up of renewable energy.

"We need it either way," he said about the infusion of renewable power. "If Diablo is extended, we need it. If it's not extended, we need it. And one of the reasons we're here is that not enough was brought online."

The state senator's district is a microcosm of California's energy transition. Criss-crossed by high-voltage transmission lines, the region not only includes the heavily fortified Diablo Canyon, nestled out of sight among rolling green hills along the ocean, but also the first proposed offshore wind farm on the West Coast. Projects like this are meant to help the state get rid of all carbon emissions — if they can be built on time.

Workers along the Central Coast have a long history of building energy projects and see the state's grid challenges as an opportunity, Dawn Ortiz-Legg, a San Luis Obispo County supervisor, said in an interview in Morro Bay, the sleepy fishing village slated to host the floating offshore wind turbines 20 miles off its shore. The county official previously helped turn homebuilders into workers who constructed large solar farms that helped lift the county out of recession.

Behind Ortiz-Legg rose three fog-shrouded, iconic smokestacks at a shuttered power plant that once burned coal and gas. Now, in an apt metaphor, they are scheduled to come down to make way for a proposed battery facility that would store renewable power.

Ortiz-Legg said she agreed with postponing the closure of the nuclear power plant — but only temporarily. Now, she's calling for state leaders to act with more urgency to make sure Diablo Canyon can be closed, according to the new plan, in 2030.

"It's really important to note that in 2001, 2002, when California had its energy crisis, gas plants were permitted in 20 days," she said.

The tight timeline is making Jane Swanson skeptical. A nearby resident, she has fought the nuclear plant since the 1970s out of concerns about nuclear waste and seismic safety, as part of the organization Mothers for Peace. She was looking forward to Pacific Gas & Electric, which runs the plant, closing it down and now fears the future will bring more broken promises.

“You can’t believe a word PG&E says,” Swanson said. “You can’t believe a word politicians say. So I have no faith that any future agreement or end date will be observed because we’ve had those dates and they’re not being observed.”

State energy regulators last year ordered utilities to add more clean energy to the grid over the next three years than ever before. But getting the solar panels and wind turbines built and plugged in depends largely on the whims of the global economy.

California energy officials went into crisis mode this spring when supply chain issues from the pandemic and a new retroactive tariff on solar panels cast a pall over the industry, delaying the renewable energy projects they had been counting on before Diablo Canyon was set to retire.

No one disputes that an additional margin of safety is needed, said Ralph Cavanagh, the energy co-director for the Natural Resources Defense Council. Still, he argued, the governor locked down on Diablo Canyon prematurely instead of looking at alternatives like bolstering energy efficiency and pulling more energy from other Western states.

Republicans in California, meanwhile, prefer the nuclear power option.

“If the plant gets decommissioned, we don’t have enough juice to keep the lights on and keep air conditioners working and keep people’s EVs charged,” said Assemblymember Jordan Cunningham, the Republican who represents the Central Coast and thinks the plant should keep running even longer.

Cunningham also largely supports California’s climate goals and helped shape the ambitious offshore wind target for the state.

“I think in five years’ time, we’ll be in a better place, with renewables coupled with storage that we need to run a modern electricity grid, but we’re just not really quite there yet," he said.

Rep. Salud Carbajal, a Democrat representing the area, has yet to take a position on the plant's extension. But he urged Newsom to consider the local implications of keeping it open, including seismic safety.

Residents on the Central Coast had tussled for decades over the role of Diablo Canyon, but until recently, they thought the debate was over. Employees made retirement plans. Construction at the plant slowed as PG&E began preparing to shut the twin reactors down. Conservationists eyed the lands around the plant.

An extension would upset those plans.

In an interview, Carbajal said he's also wondered whether it will get in the way of developing nearby offshore wind farms

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he said, referring to the debate over Diablo Canyon. "There's a lot of moving

Loading

David Shabazian, Director
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California Department of Conservation
801 K Street, MS 24-01
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October 1, 2021

RE: Response to CalGEM Questions for the California Oil and Gas Public Health Rulemaking Scientific Advisory Panel

Director Shabazian and Supervisor Ntuk,

Please find attached the responses from the California Oil and Gas Public Health Rulemaking Scientific Advisory Panel to the written questions sent by the California Geologic Energy Management Division (CalGEM) on August 31, 2021.

We would be glad to answer any further questions that may arise.

Best Regards,

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CalGEM Questions for the California Oil and Gas Public Health Rulemaking Scientific Advisory Panel

CalGEM requests the California Oil and Gas Public Health Rulemaking Scientific Advisory Panel assistance with the following questions:

1. ***How would the panel characterize the level of certainty that proximity to oil and gas extraction wells and associated facilities in California causes negative health outcomes? Is there a demonstrated causal link between living near oil and gas wells and associated facilities and health outcomes?***

We have focused our review on epidemiological studies carried out in multiple oil and gas regions, including Colorado, which has a similar regulatory context as California. Given that similar environmental health hazards and risks are intrinsic to both conventional and unconventional oil and gas development (OGD), including exposure pathways, chemicals associated with hydrocarbon reservoirs, use of ancillary equipment, and non-chemical stressors (See section on "Similarities and Differences Between Unconventional and Conventional OGD"), the California Oil and Gas Public Health Rulemaking Scientific Advisory Panel (Panel) concludes that the full body of epidemiologic literature is relevant to assess the human health hazards, risks and impacts of upstream OGD in California.

Our Panel concludes with a high level of certainty[1] that the epidemiologic evidence indicates that close residential proximity to OGD is associated with adverse perinatal and respiratory outcomes, for which the body of human health studies is most extensive in California and other locations.

Studies on Oil and Gas Development and Perinatal Outcomes

Perinatal outcome studies provide the largest [19 studies][2] and strongest body of evidence linking OGD exposure during the sensitive prenatal period with adverse health effects. The majority of studies that examine perinatal effects found increased risk of adverse birth outcomes in those most exposed to OGD (measured using metrics including, but not limited to proximity, well density, and production volume). It should also be noted that adverse perinatal outcomes, including preterm births, low birth weight, and small-for-gestational age births

[1] In this document, the statement, "a high-level of certainty" is based on the professional judgement of all California Oil and Gas Public Health Rulemaking Scientific Advisory Panel (Panel) members in their assessment of the scientific evidence. In terms of panel process, all Panel members agree with the responses to the questions in this document. Any Panel member could have written a dissenting opinion, but no one requested to do so. This document reflects the perspective of the Panel members and not necessarily the opinions of their employers or institutions.

[2] Apergis et al., 2019; Busby & Mangano, 2017; Caron-Beaudoin et al., 2020; Casey et al., 2016; Currie et al., 2017; Cushing et al., 2020; Gonzalez et al., 2020; Hill, 2018; Janitz et al., 2019; Ma, 2016; McKenzie et al., 2014, 2019; Stacy et al., 2015; Tang et al., 2021; Tran et al., 2020, *Forthcoming*; Walker Whitworth et al., 2018; Whitworth et al., 2017; Willis et al., 2021.

increase the risk of mortality and long-term developmental problems in newborns (Liu et al., 2012; Vogel et al., 2018) as well as longer term morbidity through adulthood (Baer et al., 2016; Barker, 1995; Carmody & Charlton, 2013; Frey & Klebanoff, 2016)**.**

Perinatal Outcomes Associated with Conventional and Unconventional Oil and Gas Development

While many perinatal outcome studies outside of California focus on unconventional OGD (e.g., high-volume hydraulic fracturing), a recent review of the literature (Deziel et al., 2020), highlighted the need for an updated assessment of the health effects associated with OGD more generally, as both conventional and unconventional OGD operations present health risks, especially to those living in close proximity. This bolsters conclusions reached by the authors of the 2015 independent scientific study of hydraulic fracturing and well stimulation in California led by the California Council on Science and Technology (CCST) (Long et al., 2015) pursuant to Senate Bill 4 (2013, Pavley). Recent studies in California have reported associations between exposure to OGD and adverse birth outcomes, considering wells under production using enhanced oil recovery including cyclic steam injection, steam flooding and water flooding -- methods that do not meet the definition of unconventional development (Gonzalez et al., 2020; Tran et al., 2020, *Forthcoming*). Similar findings regarding adverse birth outcomes have been reported while examining unconventional OGD in Colorado, Oklahoma, Pennsylvania and Texas (Apergis et al., 2019; Casey et al., 2016; Cushing et al., 2020; Gonzalez et al., 2020; Hill, 2018; McKenzie et al., 2019; Stacy et al., 2015; Walker Whitworth et al., 2018; Whitworth et al., 2017). In the California independent scientific study on well stimulation pursuant to Senate Bill 4 (2013, Pavley), the authors concluded that while hydraulic fracturing introduces some specific human health risks, the majority of environmental risks and stressors are similar across conventional and unconventional oil and gas operations (Long et al., 2015; Shonkoff et al., 2015). Further, a handful of epidemiological studies explicitly examine potential differences in associations between conventional or unconventional oil or natural gas development and adverse outcomes. For example, Apergis et al. (2019) reported statistically significant reductions in infant health index within 1 km of both conventional and unconventional drilling sites in Oklahoma. In summary, the Panel concludes with a high level of certainty that human health studies focused on unconventional and conventional OGD are relevant to consider in the California context where conventional development is most prevalent.

Consistency Across Perinatal Epidemiology Studies

We have a high level of certainty in the findings in the body of epidemiological studies for perinatal health outcomes because of the consistency of results across multiple studies that were conducted using different methodologies, in different locations, with diverse populations, and during different time periods (see **Table 1** below). Most of these studies entail rigorous, high quality analyses (i.e., study designs that establish temporality based on large sample sizes, control for potential individual and area-level confounders, apply rigorous statistical modelling techniques, and conduct sensitivity analyses to assess the robustness of effects). A variety of pollutants (e.g., $PM_{2.5}$ and air toxics) and other OGD stressors are associated with these same adverse birth outcomes (Dzhambov & Lercher, 2019; Nieuwenhuijsen et al., 2017; Shapiro et al., 2013), which further strengthens the evidence of the link between OGD and adverse perinatal outcomes. Therefore, the totality of the epidemiological evidence provides a high level of certainty that exposure to OGD (and associated exposures) cause a significant increased risk of poor birth outcomes.

Further, imprecision in exposure assessment or non-differential exposure misclassification in some of the epidemiological studies is more likely to attenuate observed relationships, thus leading to an underestimate of the true adverse impacts of OGD on birth outcomes (**Figure 1**). In environmental epidemiologic studies, researchers often use surrogates to estimate exposures or assign individuals to exposure categories; these surrogates have some measurement error associated with them. When these errors in assigning or classifying participant exposures are similar between exposed and unexposed or those with or without the health outcome, this is referred to as non-differential exposure misclassification. This type of "noise" in the data tends to dilute or attenuate the true exposure-response relationship, as illustrated by the hypothetical dashed line in **Figure 1**, which has a shallower slope compared to the hypothetical "true" solid line.

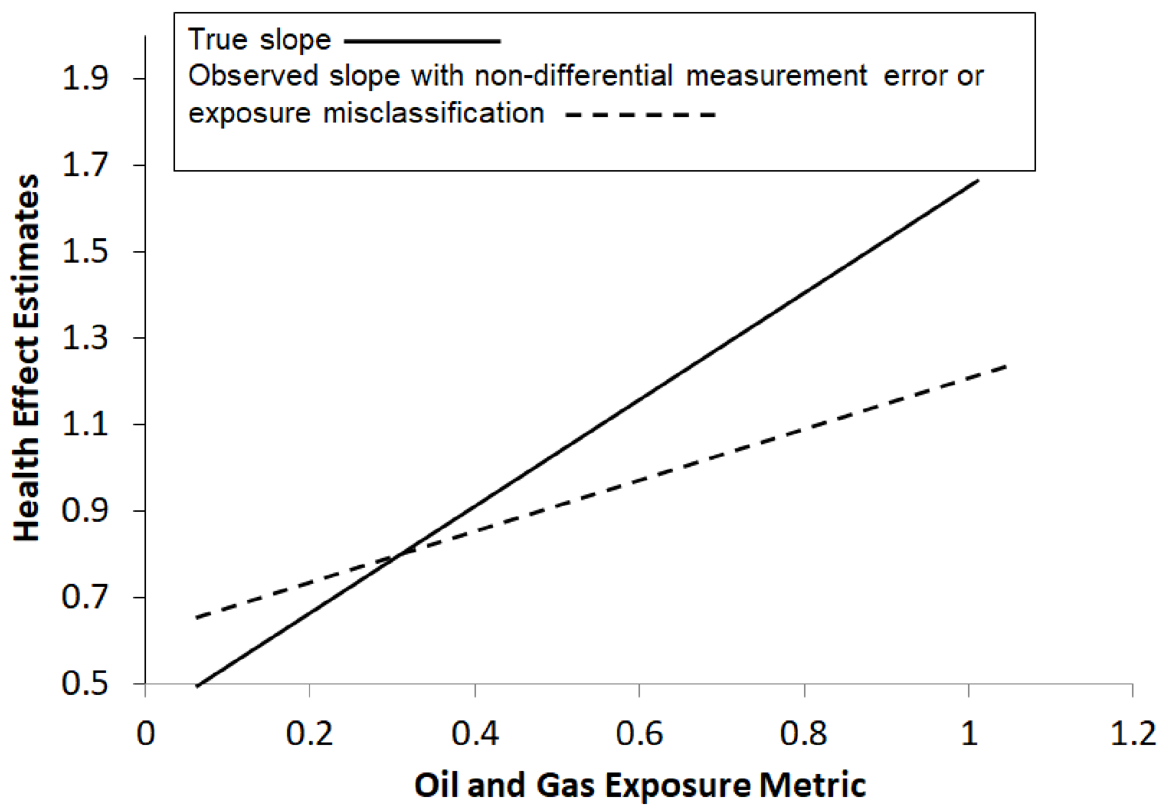

Figure 1. Effect of imprecise exposure estimates on a hypothetical exposure-response relationship (Source: Adapted from Seixas & Checkoway, 1995).

Respiratory Risks and Impacts from Oil and Gas Development

Respiratory health outcomes are the second most studied health outcomes in the epidemiological literature examining OGD, with eight peer-reviewed studies published to date. Two peer-reviewed studies in California found an association between OGD and self-reported and physician-diagnosed asthma, reduced lung function, and self-reported acute respiratory symptoms (e.g., recent wheeze) (Johnston et al., 2021; Shamasunder et al., 2018). Six studies in other oil and gas regions (Pennsylvania and Texas) reported an association between OGD and asthma exacerbations, asthma hospitalizations, and respiratory symptoms (Koehler et al., 2018; Peng et al., 2018; Rabinowitz et al., 2015; Rasmussen et al., 2016; Willis et al., 2018, 2020).

Epidemiological studies, by design, often use aggregate measures of exposure to account for multiple potential stressors and pathways associated with OGD (e.g., air pollution, noise pollution, groundwater and/or drinking water contamination). Many criteria air pollutants (e.g., particulate matter, ozone, nitrogen oxides) and hazardous air pollutants emitted from OGD have a well-established body of scientific literature indicating that exposure to these pollutants causes an increased risk of development and exacerbation of respiratory disease (Bolden et al., 2015; Ferrero et al., 2014). We reiterate the relevance of studies on both conventional and unconventional OGD for respiratory health outcomes. For example, (Willis et al., 2020) found that both conventional and unconventional natural gas development at the ZIP code level was associated with pediatric asthma hospitalizations in Texas.

Comparing The Body of Perinatal and Respiratory Outcome Studies Against The Bradford Hill Criteria for Causation

Below, we demonstrate how the body of epidemiological studies on the relationship between OGD and perinatal and respiratory outcomes meets the nine Bradford Hill Criteria for Causation (Hill, 1965; Lucas & McMichael, 2005). The Bradford Hill Criteria are used to evaluate the strength of epidemiological evidence for determining a causal relationship between an exposure and observed effect. These criteria are widely used in the field of epidemiology and public health practice to guide decision-making. After considering these criteria, the Panel concludes with a high level of certainty that there is a causal relationship between close geographic proximity to OGD and adverse perinatal and respiratory outcomes (**Table 1**).

Table 1. Application of the Bradford Hill Criteria for Causation to the peer-reviewed epidemiological literature on oil and gas development and perinatal and respiratory health outcomes.

Criteria for Causation (Bradford-Hill)	Description of Criteria	Perinatal Health Studies	Respiratory Health Studies
Strength of Association	Environmental studies commonly report modest effects sizes (i.e., relative to active tobacco smoking or alcohol consumption). A small magnitude of association can support a causal relationship, a larger association may be more convincing.	Reported effect sizes are in ranges similar to other well-established environmental reproductive and developmental hazards, such as $PM_{2.5}$ (Dadvand et al., 2013; C. Li et al., 2020). Some studies, particularly those in California, have found stronger effect estimates for OGD exposures among socially marginalized groups (Cushing et al., 2020; Gonzalez et al., 2020; Tran et al., 2020, *Forthcoming*).	Reported effect sizes are in ranges similar to other well-established environmental respiratory hazards. For example, effect sizes in reductions in lung function by Johnston et al. (2021) are similar in magnitude to reductions in lung function associated with secondhand smoke exposure among women (Eisner, 2002) and reductions in lung function among adults living near busy roadways (e.g., (Kan et al., 2007).
Consistency	Consistent findings observed by different persons in different places with different samples strengthens the likelihood of an effect.	Adverse birth outcomes have been observed in multiple studies using multiple methods in different populations at different times and locations (e.g., California, Pennsylvania, Colorado, Texas). While there is some variation in findings by specific perinatal outcomes, the overall body of evidence is highly consistent in supporting the association between OGD and adverse perinatal outcomes.	Various respiratory health outcomes are evaluated in the literature. For asthma -- the most commonly studied respiratory health outcome -- studies across California, Pennsylvania and Texas consistently show an association between OGD and asthma-related metrics (asthma prevalence, exacerbations, pediatric hospitalizations) (Koehler et al., 2018; Rasmussen et al., 2016; Shamasunder et al., 2018; Willis et al., 2018, 2020) .

Criteria for Causation (Bradford-Hill)	Description of Criteria	Perinatal Health Studies	Respiratory Health Studies
Specificity	Causation is likely if there is no other likely explanation.	All peer-reviewed birth outcome studies included in our review controlled for other potential confounders by (i) accounting or adjusting for other individual-level or area-level factors (e.g., other air pollution sources, neighborhood socioeconomic status) in the analysis (Casey et al., 2016; McKenzie et al., 2014; Tran et al., 2020, *Forthcoming*). Other studies applied statistical modeling approaches such as difference-in-difference that accounts for temporal and spatial trends that may confound observed effects (Willis et al., 2021).	Most respiratory health studies have controlled for other potential explanatory or confounding factors by (i) accounting or adjusting for other individual-level (e.g., smoking status) or area-level factors (e.g., other air pollution sources) in the analysis (Johnston et al., 2021; Koehler et al., 2018; Peng et al., 2018; Rabinowitz et al., 2015; Rasmussen et al., 2016; Willis et al., 2018, 2020), or in the study design, such as utilizing a difference-in-difference methodology (Peng et al., 2018; Willis et al., 2018).
Temporality	Exposure precedes the disease.	Most birth outcomes studies have proper temporal alignment between exposure and outcome and use a retrospective cohort, case control or other study design that allows retroactive assessment of exposures to OGD occurring before the onset of disease. They do not consider exposure that occurred at the time of disease or oil and gas wells drilled after the disease.	Some respiratory health studies do not allow for assessments of exposure that predate disease. However, of the studies with the proper temporal alignment (Johnston et al., 2021; Koehler et al., 2018; Peng et al., 2018; Rasmussen et al., 2016; Willis et al., 2018), authors report statistically significant associations between OGD and oral corticosteroid medication orders, asthma hospitalizations and asthma-related emergency department visits.

Criteria for Causation (Bradford-Hill)	**Description of Criteria**	**Perinatal Health Studies**	**Respiratory Health Studies**
Biological Gradient (Dose-Response)	Greater exposure leads to a greater likelihood of the outcome.	Some studies have found dose-response relationships based on oil and gas production volume categories or metrics of inverse distance weighting and/or oil and gas well density in California and elsewhere (Casey et al., 2016; McKenzie et al., 2014, 2019; Tang et al., 2021; Tran et al., 2020).	Larger reductions in lung function observed with decreased distance from active oil development sites (Johnston et al., 2021).
Plausibility	The exposure pathway and biological mechanism is plausible based on other knowledge.	Individual health-damaging chemical pollutants are well-understood to be emitted from OGD (e.g., $PM_{2.5}$, benzene) and established as contributing to increased risk for the same adverse perinatal outcomes observed in the epidemiology studies. Stressors associated with OGD (e.g., psychosocial stress; (Casey et al., 2019) can also contribute to increased adverse perinatal outcomes.	Many air pollutants associated with OGD are well-known to contribute to respiratory morbidity and mortality, including exacerbations of existing respiratory conditions (Guarnieri & Balmes, 2014).
Coherence	Causal inference is possible only if the literature or substantive knowledge supports this conclusion.	In particular, the body of peer-reviewed literature is converging towards singular directions for adverse perinatal outcomes.	The body of peer-reviewed literature points in a singular direction for adverse respiratory health outcomes.

Criteria for Causation (Bradford-Hill)	Description of Criteria	Perinatal Health Studies	Respiratory Health Studies
Experiment	Causation is a valid conclusion if researchers have seen observed associations in prior experimental studies.	N/A- Human population-based experimental studies are not available due to ethical issues.	N/A- Human population-based experimental studies are not available due to ethical issues.
Analogy	For similar programs operating, similar results can be expected to bolster the causal inference concluded.	Pollutants well known to be emitted during OGD including benzene, toluene and 1,3 butadiene are listed as reproductive or developmental toxicants under Prop 65 and thus are recognized as such by the State of California (CalEPA OEHHA, 2021). EPA's current Integrated Science Assessments of particulate matter and tropospheric ozone conclude that the evidence is suggestive of, but is not sufficient to infer, a causative relationship between birth outcomes, including preterm birth and low birth weight, and $PM_{2.5}$ and long term ozone exposures (US EPA, 2019, 2020). Additionally, increased stress during pregnancy can alter fetal growth and length of gestation (Fink et al., 2012).	EPA's current Integrated Science Assessments of particulate matter and tropospheric ozone conclude that there is: a casual relationship between respiratory outcomes, including asthma and short term ozone exposure; and likely a causal relationship between respiratory outcomes, including asthma and: short and long term $PM_{2.5}$ exposure; and long term ozone exposure (US EPA, 2019, 2020).

Similarities and Differences Between Unconventional and Conventional Oil and Gas Development

Though definitions of conventional and unconventional OGD may differ across different regulatory and policy landscapes, the majority of OGD in California is often considered conventional, involving vertical drilling at shallower depths into target geologies that hold migrated hydrocarbons. These attributes of development are often considered in contrast to unconventional OGD, which can involve horizontal directional drilling in deeper wells to access source rock formations by increasing the permeability of these tight formations using mostly hydraulic fracturing. In addition, these unconventional operations are often accompanied with greater masses of material inputs (e.g., water, chemical additives, proppants) and a greater magnitude of liquid and solid waste outputs (e.g., flowback fluids and produced water). It should be noted, however, that hydraulic fracturing that takes place in California often uses fluids (gels) with higher concentrations of well stimulation chemicals than those fluids used in high-volume slick water hydraulic fracturing of source rock in other parts of the United States (Long et al., 2015).

However, many environmental and health hazards and risks are intrinsic to both conventional and unconventional OGD (Hill et al., 2019; Jackson et al., 2014; Lauer et al., 2018; Stringfellow et al., 2017; Zammerilli et al., 2014). $PM_{2.5}$ and nitrogen oxides emissions result from the use of diesel-powered equipment and trucks and hazardous air pollutants such as benzene, toluene, ethylbenzene and xylene (BTEX) occur naturally in oil and gas formations, regardless of the type of extraction method employed. Noise pollution, odors, and landscape disruption are inherent to OGD. Investigations in other oil and gas states have noted radioactivity on particles downwind from unconventional oil and gas wells (Li et al., 2020b) and in sediment downstream of water treatment plants that treat waste from conventional as well as unconventional oil and gas operations (Burgos et al., 2017; Lauer et al., 2018).

In California, policy, regulatory and scientific emphasis has been placed on well stimulation activities, including hydraulic fracturing, matrix acidizing and acid fracturing. The 2015 Independent Scientific Assessment on Well Stimulation in California, which focused primarily on well stimulation activities pursuant to Senate Bill 4 (2013, Pavley), reported the following key conclusion: *"The majority of impacts associated with hydraulic fracturing are caused by the indirect impacts of oil and gas production enabled by the hydraulic fracturing"* (Long et al., 2015). Indirect impacts relevant to human health for the purposes of the study included: "proximity to any oil production, including stimulation- enabled production, could result in hazardous emissions to air and water, and noise and light pollution that could affect public health" (Long et al., 2015). Additionally, a recent evaluation of chemical usage during OGD in California found significant overlap in chemical additives used for well stimulation (including hydraulic fracturing) and those used in routine activities, such as well maintenance (Stringfellow et al., 2017).

2. What are the air pollutants released from these activities that cause negative health outcomes? How do we know exposure to these is likely from oil and gas extraction wells and associated facilities, as opposed to other sources?

The wells, valves, tanks and other equipment used to produce, store, process and transport petroleum products at both unconventional and conventional OGD sites are associated with emissions of toxic air contaminants, hazardous air pollutants and other health-damaging non-methane VOCs (Helmig, 2020; Moore et al., 2014). Diesel engines used to power on-site equipment and trucks at unconventional and conventional OGD sites directly emit health-damaging hazardous air pollutants, fine particulate matter ($PM_{2.5}$), nitrogen oxides and volatile organic compounds (VOCs) (CalEPA OEHHA, 2001). Many VOCs and nitrogen oxides are precursors to ground level ozone (O_3) formation, another known health harming pollutant. Hazardous air pollutants that are known to be emitted from OGD sites include benzene, toluene, ethylbenzene, xylenes, hexane and formaldehyde--many of which are known, probable or possible carcinogens and/or teratogens and which have other adverse effects for non-cancer health outcomes (CalEPA OEHHA, 2008, 2009; Moore et al., 2014). In the San Joaquin Valley Air Pollution Control District, OGD activities are responsible for the majority of emissions of multiple toxic air contaminants including acetaldehyde, benzene, formaldehyde, hexane and hydrogen sulfide (**Figure 2**) (Brandt et al., 2015; Long et al., 2015).

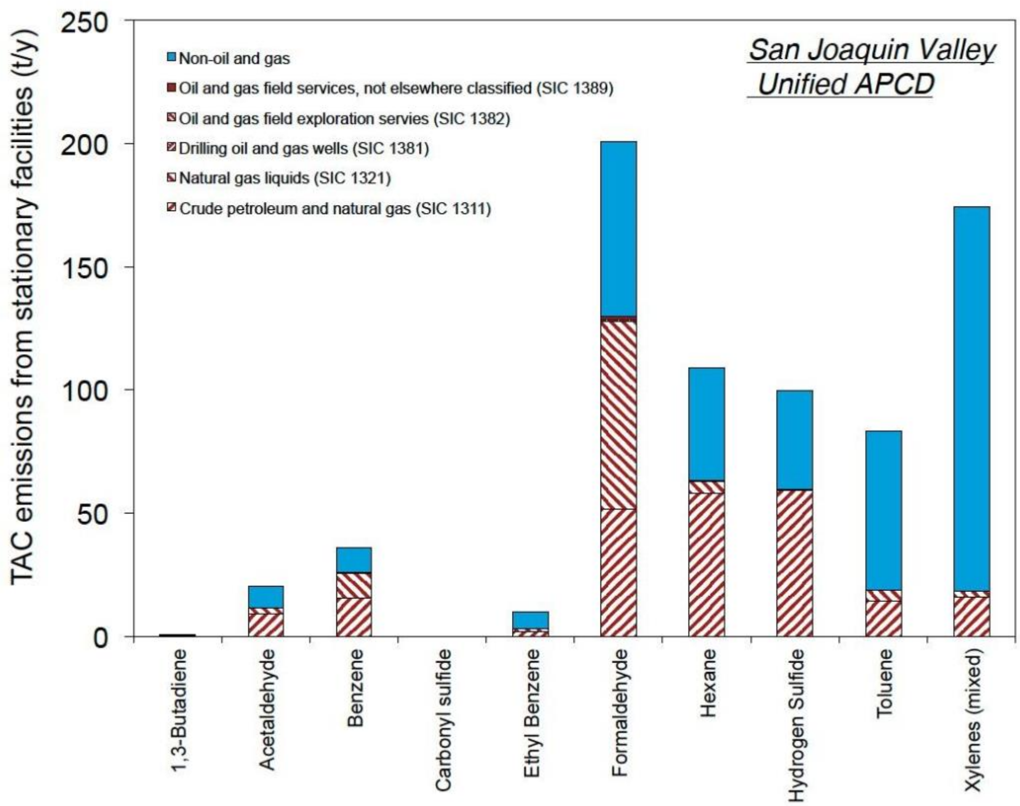

Figure 2. **Toxic Air Contaminant emissions from stationary facilities in the San Joaquin Valley Air Pollution Control District (Source: (Brandt et al., 2015).**

A recently published study using statewide air quality monitoring data from California investigated whether drilling new wells or increasing production volume at active wells resulted in emissions of $PM_{2.5}$, nitrogen dioxide (NO_2), VOCs, or O_3 (Gonzalez et al., 2021)**.** To assess the effect of oil and gas activities on concentrations of air pollutants, the authors used daily variation in wind direction as an instrumental variable and used fixed effects regression to control temporal factors and time-invariant geographic factors. The authors documented higher concentrations of $PM_{2.5}$, NO_2, VOCs, and O_3 at air quality monitoring sites within 4 km of pre-production OGD well sites (i.e., wells that were between spudding and completion) and 2 km of production OGD well sites, after adjusting for geographic, meteorological, seasonal, and time trending factors. In placebo tests, the authors assessed exposure to well sites downwind of the air monitors and observed no effect on air pollutant concentrations. **Table 2** summarizes the increases in each pollutant for each additional upwind well site by distance.

Table 2. Summary of air pollutant concentrations measured between 2006-2019 at 314 air quality monitoring sites in the EPA Air Quality System for California (Gonzalez et al., 2021*)*.

Distance	$PM_{2.5}$ µg/m^3*	NO_2 ppb	VOCs (ppb C)*	O_3 (ppb)
Estimated increase for each additional upwind pre-production well site				
Within 2 km	2.35 (0.81, 3.89)	2.91 (0.99, 4.84)	No increase	no increase
2-3 km	0.97 (0.52, 1.41)	0.65 (0.31, 0.99)	No increase	0.31 (0.2, 42)
3-4 km	no increase	no increase	no increase	0.14 (0.05, 0.23)
Estimated Increase for each 100 BOE of total oil and gas upwind production volume				
1 km	1.93 (1.08, 2.78)	0.62 (0.37, 0.86)	0.04 (0.01, 07)	no increase
1-2 km	no increase	no increase	no increase	0.11 (0.08, 0.14)

*No $PM_{2.5}$ or VOC monitoring sites with 1 km of pre-production well sites; BOE, barrels of oil equivalents.

These multiple stressors, along with other physical factors such as noise and vibration, are consistently found in exposure studies to be measurably higher near oil and gas extraction wells and other ancillary infrastructure in California. <u>As such, the Panel concludes with a high level of certainty that concentrations of health-damaging air pollutants, including criteria air pollutants and toxic air contaminants, are more concentrated near OGD activities compared to further away.</u>

3. ***Does the evidence evaluated clearly support a specific setback? If so, what is this setback distance and what oil and gas extraction activities would it specifically apply to? What is the supporting evidence?***
    a. ***How does this evidence justify the recommended setback distance, as opposed to another distance?***

Existing epidemiologic studies were not designed to test and establish a specific "safe" buffer distance between OGD sites and sensitive receptors, such as homes and schools. Nevertheless, <u>studies consistently demonstrate evidence of harm at distances less than 1 km, and some studies also show evidence of harm linked to OGD activity at distances greater than 1 km.</u> In addition, exposure pathway studies have demonstrated through measurements and modelling techniques, the potential for human exposure to numerous environmental stressors (e.g., air pollutants, water contaminants, noise) at distances less than 1 km (e.g., Allshouse et al., 2019; Holder et al., 2019; McKenzie et al., 2018; DiGiulio et al., 2021; Soriano et al., 2020), and that the likelihood and magnitude of exposure decreases with increasing distance.

b. ***What are the health benefits from this setback? Can the panel quantify them or recommend a methodology CalGEM can use to quantify them? Can the panel establish that these health benefits can only be achieved with the setback? Or can they also be achieved with mitigation controls?***

Figure 3 presents a hierarchy of strategies to reduce human health hazards, risks and impacts from OGD activities. **Table 3** presents the advantages and disadvantages of each strategy from an environmental public health perspective.

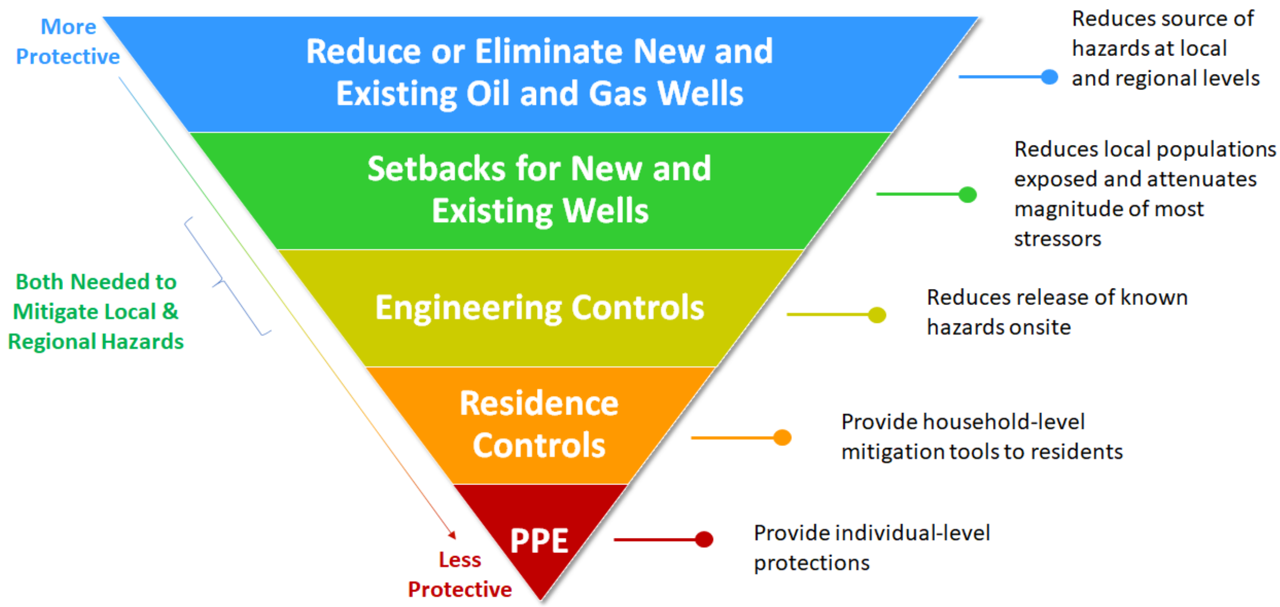

Figure 3. Hierarchy of strategies to reduce or eliminate public health harms for OGD activities. Note: the use of the term "wells" includes the ancillary infrastructure used to develop, gather and process oil and gas in the upstream oil and gas sector.

At the top of Figure 3 is the most health protective strategy: to stop drilling and developing new wells, phase out existing OGD activities and associated infrastructure, and properly plug remediate legacy wells and ancillary infrastructure.

If the development of oil and gas is to continue, the greatest health benefits would be gained from a strategy that includes the next two controls in the hierarchy depicted in Figure 3: the elimination of new and existing wells and ancillary infrastructure within scientifically informed setback distances and the deployment of engineering emission controls and associated monitoring approaches that lead to rapid leak detection and repair for new and existing wells and ancillary infrastructure. Because air pollutant concentrations and noise levels decrease with increasing distance from a source, adequate setbacks can reduce harm to local populations by reducing exposures to air pollutants and noise directly emitted from the OGD activities. However, setbacks do not reduce harms from OGD contributions to regional air pollutant levels, such as secondary particulate matter and ozone, or greenhouse gases, such as methane, which are nearly always co-mingled with health-damaging air pollutants (Michanowicz et al., *Forthcoming*). Engineering controls that reduce emissions at the well site are also necessary to reduce these harms.

Engineering controls include cradle-to-grave noise and air pollution emission mitigation controls on OGD infrastructure including new, modified and existing infrastructure, and proper abandonment of legacy infrastructure, prioritizing those nearest to residential sites and schools and those associated with the highest emissions, leaks and other environmental hazards.

However, engineering controls can fail and engineering solutions may not be available for or economically feasible to handle all of the complex stressors generated by OGD, including multiple sources and types of air pollution, noise pollution, light pollution, water pollution, and other stressors. <u>Therefore, neither setbacks or engineering controls alone are sufficient to reduce the health hazards and risks from OGD activities -- both approaches are needed in tandem</u>.

Finally, we note that while outside of CalGEM's jurisdiction, setbacks for new construction of housing or schools at a certain distance from existing or permitted OGD sites (commonly referred to as reverse setbacks), should be considered.

Table 3. Advantages and Disadvantages of Oil and Gas Development Control Strategies from an Environmental Public Health Perspective.

Control Strategy	Description	Advantage	Disadvantage
Elimination	Eliminate or reduce new and existing wells and ancillary infrastructure in combination with proper plugging and abandonment of wells and other legacy infrastructure.	Eliminates the source of nearly all environmental stressors (e.g., air and water pollutants, noise); protects local and regional populations	None.
Setbacks	Increase the distance between OGD hazards and sensitive receptors.	Reduces risk of exposures to populations living near OGD sites; environmental stressors are generally attenuated with increasing distance.	Setbacks alone without coupled engineered mitigation controls allow continued release of hazards and therefore does not adequately address air pollutant and greenhouse gas emissions from OGD and their impacts on regional air quality and the climate.
Engineering Controls	Reduces or eliminates release of specific hazards on site.	Reduces or eliminates certain hazards and therefore can have local and regional environmental public health benefits.	Tends to be disproportionately focused on air pollutant emissions. Often not feasible to apply engineering solutions to multiple, complex stressors each requiring different control technologies (e.g. noise, air and water impacts, social stressors) and lacks the important factor of safety provided by a setback when engineering controls fail.
Residence Controls	Provides households with devices to reduce hazard at the home (e.g., water filter, light-blocking shades, air filters).	Reduces intensity of certain hazards to nearby communities at the household level.	Places burden on individuals and households to use devices properly and to maintain and regularly replace controls to maximize effectiveness. Not feasible to apply devices to address numerous, complex stressors.
Personal Protective Equipment	Provide individuals with devices to reduce exposure (e.g., respiratory masks, ear plugs, eye masks).	Reduces intensity of exposure of certain hazards to nearby individuals.	Places burden on individuals to use PPE consistently and properly and is not feasible for the complex stressors.

Attributable Risk Calculations

One method to estimate health harms from OGD is to use the measures of association from the epidemiologic literature and population counts to calculate the excess number of specific health outcomes. This is what is known as an attributable risk method. We may be able to derive these estimates in the final report for birth outcomes using estimates of population counts for women of reproductive age in California living near OGD sites. We will also attempt to derive similar estimates for respiratory outcomes by using age appropriate population counts near OGD sites. This attributable risk method can allow us to estimate the number of adverse perinatal or respiratory cases that are attributable to OGD exposures and could be attenuated through the implementation of elimination or setback strategies.

c. ***Can the panel quantify or recommend a methodology CalGEM can use to quantify the health benefits associated with mitigation controls?***

The Panel was not tasked to estimate health benefits of various setbacks and mitigation strategies, which pose significant methodological challenges and would require considerable time and effort. Among the challenges is the need to consider the benefits of reducing multiple stressors -- multiple air pollutants and other chemicals, noise, vibration, light, subsurface contamination, etc.

Known Health Benefits of Reducing Air and Noise Pollution

There is a significant body of literature and available tools that address the potential health benefits that can be achieved by reducing air and noise pollution exposures. The National Institute of Environmental Health Sciences has linked air pollution and specifically $PM_{2.5}$ to respiratory disease, cardiovascular disease, cancer, and reproduction harm and provides references supporting these links (NIEHS (National Institute of Environmental Health Sciences), 2021). Schraufnagel et al. (2019) examined in detail the health benefits of air pollution reductions in different geographic regions. Friedman et al. (2001) showed that improvements in air quality in preparation for the 1996 Atlanta Olympics resulted in significantly lower rates of childhood asthma events, including reduced emergency department visits and hospitalizations. Avol et al. (2001) demonstrated that children in southern California who moved to communities with higher air pollution levels had lower lung function growth rates than children who moved to areas with lower air pollution levels. Gauderman et al. (2015), examining the impact of reductions in $PM_{2.5}$ and nitrogen dioxide in the Los Angeles air basin, found that children who grew up after air quality improvements had less than ½ the chance of having clinically low lung function results. Ha et al. (2014) found $PM_{2.5}$ exposures in all trimesters to be significantly and positively associated with the risk of all adverse birth outcomes.

In an analysis of noise exposure reductions. Based on sound levels measured and/or modeled across the US together with an EPA exposure- response model for levels exceeding EPA standards, Swinburn et al. (2015) found that a 5-dB noise reduction scenario in communities with noise exceeding EPA standards would reduce the prevalence of hypertension by 1.4% and coronary heart disease by 1.8%. The types of health-benefit studies noted here provide a basis for conducting a health-benefits analysis using a tool such as US EPA's Environmental Benefits Mapping and Analysis Program—Community Edition (BenMAP-CE) (US EPA, 2021).

Possible Approaches to Quantify Health Benefits

CalGEM could obtain estimates of the health benefits achieved from different mitigation strategies individually or in combination with tools such as the Community Multiscale Air Quality Model (CMAQ) (Binkowski & Roselle, 2003) and/or other exposure assessment tools and link model output to EPA's BenMAP-CE (US EPA, 2021). However, these models and approaches are only focused on air quality and noise. It should also be noted that a significant drawback of using BenMAP-CE for this application is that it only considers impacts from criteria air pollutants and not from toxic air contaminants or other emerging air pollutants.

BenMAP-CE estimates the number and economic value of health impacts resulting from changes in air pollution concentrations. BenMAP-CE estimates benefits in terms of the reductions in the risk of premature death, heart attacks, and other adverse health effects. BenMAP-CE requires as input, pollutant concentrations at a scale that matches with population data. These concentrations can be obtained from a model such as CMAQ (Binkowski & Roselle, 2003) or from a monitoring network. BenMAP-CE takes the concentration fields for a base case and then for a pollution reduction (or increase) to assess health benefits (or detriments). BenMAP-CE then estimates changes in health endpoints, allowing the user to specify the concentration–response function and either use built-in population and baseline mortality rates or specify them as inputs.

It should be noted that in order to use a model such as BenMAP-CE to assess health benefits of setbacks and mitigation controls at well sites across California would involve a significant level of time and effort in data collection and model executions. In addition, these models are limited to characterizing the health benefits of criteria air pollutant reductions, but do not account for other OGD related exposures such as toxic air contaminants, other chemical exposures and exposures to other stressors through other environmental pathways (e.g., water and noise). Additionally, and importantly, the lack of spatially resolved emissions data from upstream OGD introduces challenges when assessing local- and sub-regional scaled health impacts that would be required for calculating benefits of specific policies such as setbacks and emission control. As such, <u>attempts to quantify benefits using BenMAP-CE are likely to underestimate them</u>.

4. ***CalGEM is aware of health risk assessments, health impact assessments, air exposure studies, and workforce safety studies that have been conducted but were not evaluated as part of your preliminary advice. How do these studies align with your causation determination, any recommended setback distance, and recommendations on health benefits quantification?***

The Panel determined early in its deliberations that it would limit the studies assessed in its report to those in the peer-reviewed scientific literature. This criterion ensures that studies have been evaluated by scientists who have not been involved with the study but have expertise in the relevant topic area and/or the methods used to carry out analyses, prior to publication. The peer-review process helps to ensure that high quality data and scientific interpretations are at the core of the science-policy decision-making process. Authors of peer reviewed studies are more likely to have been questioned about their methods, data interpretations, and conclusions, leading to greater confidence in the results.

In addition, the Panel was not tasked with assessing occupational studies. If CalGEM staff are aware of any peer-reviewed studies that were not included in our preliminary advice, we encourage them to send the Panel references so that we can evaluate them for inclusion in the final report. We intend to scan the literature again to assess whether relevant studies have been published since we completed the draft report. <u>Should additional peer-reviewed studies be identified, the Panel will evaluate them to determine if they align with the scope of the report and should be added</u>.

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Aquatic Toxicology 230 (2021) 105714

Contents lists available at ScienceDirect

Aquatic Toxicology

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

Hepatobiliary PAHs and prevalence of pathological changes in Red Snapper

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A R T I C L E I N F O

Keywords:
Liver
Pathology
DWH
Fish
Polycyclic aromatic hydrocarbons
Gulf of Mexico
Microscopic hepatic changes

A B S T R A C T

Red Snapper *(Lutjanus campechanus)* were collected throughout the Gulf of Mexico (GoM) from 2011 to 2017 and analyzed for biliary (n = 496) fluorescent aromatic compounds (FACs), hepatic (n = 297) polycyclic aromatic hydrocarbons (PAHs) and microscopic hepatobiliary changes (MHC, n = 152). Gross and histological evaluations were conducted with liver tissues to identify and characterize pathological changes. This is the first report to interrelate hepatobiliary PAH concentrations and MHCs in Red Snapper. Hepatic PAHs measured in GoM Red Snapper ranged from 192 to 8530 ng g^{-1} w.w. and biliary FACs ranged from 480 to 1,100,000 ng FAC g^{-1} bile. Biliary FACs in Red Snapper collected along the west Florida Shelf and north central region declined after 2011 and were relatively stable until a sharp increase was noted in 2017. Increases in the PAH exposures are likely due to a number of sources including leaking infrastructure, annual spills, riverine input and the resuspension of contaminated sediments. In contrast, hepatic PAH concentrations were relatively stable indicating Red Snapper are able to maintain metabolic clearance however this energetic cost may be manifesting as microscopic hepatic changes (MHCs). Virtually all (99 %) of the evaluated Red Snapper had one to nine MHCs with an average of five coinciding changes in an individual fish. The observed changes were broadly classified as inflammatory responses, metabolic responses, degenerative lesions, nonneoplastic proliferation and neoplastic lesions. Biliary FACs were associated with parasitic infection and intracellular breakdown product accumulation such as intra-macrophage hemosiderin, lipofuscin and ceroid laden prevalence. Whereas, hepatic PAHs were associated with increased myxozoan plasmodia prevalence. This study evaluates relationships between hepatobiliary PAH concentrations and biometrics, somatic indices, condition factors and microscopic hepatic changes in Red Snapper located in the north central GoM. Together, these results may be signaling increased disease progression in Gulf of Mexico Red Snapper more than likely resulting from chronic environmental stressors including elevated PAH exposures and concentrations.

1. Introduction

Red Snapper *(Lutjanus campechanus)* are the dominant species of the mid-shelf reef complex distributed along the continental shelf throughout the United States (U.S.) and Mexican waters of the Gulf of Mexico (GoM) and the southeast U.S. Atlantic coast. This is a long-lived demersal species (≤50 years) that significantly supports both recreational and commercial fisheries. The latest stock assessment of GoM Red Snapper indicates the stock is no longer overfished but continues to rebuild from the severely overfished and depleted conditions experienced over the past few decades. Due to the ecological and economic importance of Red Snapper in the GoM, this species was the focus of a number of impact-related studies following the *Deepwater Horizon* (DWH) oil spill in 2010. Post-DWH, Red Snapper exhibited shifts in diet and trophic level (Tarnecki and Patterson, 2015), declines in growth (Herdter et al., 2017) and abundance (Pulster et al., 2020c), and an increased frequency of epidermal lesions (Murawski et al., 2014).

The most toxic component of oil, polycyclic aromatic hydrocarbons (PAHs), are of great interest due to their association with a wide range of deleterious health effects in fish (Collier et al., 2013; Pulster et al., 2020c). Common mechanisms of toxicity include metabolic activation of PAHs generating reactive intermediates or metabolites, the activation of the aryl hydrocarbon receptor (AHR) and AHR-dependent alterations in gene expression. Persistent AHR activation by planar xenobiotics, including many PAHs, potentially mediates various toxicological events in tissues as well as vascular and immune systems (Curtis et al., 2011). Toxic chemicals and other xenobiotics are primarily metabolized in the liver of vertebrates. The morphological features of a fish liver have the

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https://doi.org/10.1016/j.aquatox.2020.105714
Received 8 June 2020; Received in revised form 24 November 2020; Accepted 2 December 2020
Available online 7 December 2020
0166-445X/

same general circulatory (i.e., hepatic arterioles, portal veins, hepatic veins) and biliary conduits as their mammalian counterpart (Wolf and Wolfe, 2005). Most fish species have a single-lobed liver with two main circulatory regions of the vasculature. However, there are subtle interspecies differences in the sinusoidal structure and considerable interspecies differences in the length and position of the ducts making up the biliary system (Gingerich, 1982). The conformation of fish hepatocytes are often described as double-layered cords representing blind-ended, anastomosing and branching tubules with a lack of apparent organization. Furthermore, histological findings tend to reveal a more vacuolated (higher glycogen and/or lipid content) compared to mammals (Gingerich, 1982; Wolf and Wolfe, 2005). All fish have a resident population of hepatic macrophages as part of the reticuloendothelial system. While they can be histologically unapparent in healthy fish, unique species anatomic variation or activation of the reticuloendothelial system can present as individualized or small nests of resident pigmented macrophage aggregates (PMAs). Nonresident perisinusoidal macrophages and hepatic stellate cells (Ito cells, perisinusoidal cells, fat-storing cells, lipocytes) can also be seen in the normal hepatic structure.

Physiologically, the same basic metabolic functions of the liver are also similar between fish and mammals, including energy metabolism and storage, storage of nutrients, synthesis of enzymes and other cofactors, bile formation and excretion (food digestion), hormone production (e.g., vitellogenin), and xenobiotic metabolism (Wolf and Wheeler, 2018; Wolf and Wolfe, 2005). Fish have a well-developed biotransformation system capable of converting xenobiotics (e.g., PAHs) into water-soluble forms that can be easily eliminated but they lack a highly developed deoxyribonucleic acid (DNA) repair system to combat DNA damage caused by xenobiotics prior to elimination (Varanasi et al., 1987). For instance, the PAH metabolic intermediates of benzo[*a*]pyrene (e.g., 7,8-oxide benzo[*a*]pyrene and 7,8-dihydrodiol benzo[*a*]pyrene) have demonstrated genotoxic and carcinogenic properties by forming DNA adducts which induce mutations leading to cancer. The liver is also a common target for both cytotoxicity and tumorigenesis in part due to biotransformation reactions that can enhance toxicity and carcinogenicity of metabolites (Hinton et al., 2001; Wolf and Wheeler, 2018). Additionally, the enterohepatic cycling mechanism in fish prolongs the removal of certain compounds and thus creates the potential for increased organ accumulation and toxicity (Gingerich, 1982). Resultingly, biliary PAH metabolites in fish serve as a sensitive indicator of recent oil exposure (e.g., days to weeks), whereas, PAHs in tissues serve as an indicator of long-term (months to years) chronic exposure.

Histopathological alterations or microscopic hepatic changes (MHCs) at both the molecular and cellular level of an organism have been documented in various organs (e.g., liver, kidney, gill) of fish following oil exposures (Brown-Peterson et al., 2015; Katsumiti et al., 2008; Marty et al., 2003; Whitehead et al., 2012). Examples of MHCs observed in fish liver following exposures to oil include hepatic intravascular congestion, lipid changes, necrosis, cystic degeneration, foci of cellular alteration (FCA), and neoplasia (Brown-Peterson et al., 2015; Oliva et al., 2013; Reddy et al., 1999). These MHCs, as well as others described in the literature, provide definitive biological end-points of historical exposures and has led to their increased usage as indicators of environmental contamination (Oliva et al., 2013).

Previous studies have reported chronic and increasing hepatobiliary PAH concentrations in GoM fishes (Pulster et al., 2020d), including ecologically and economically important demersal species including Golden Tilefish *Lopholatilus chamaeleonticeps* (Snyder, 2020), hakes *Urophycis* spp.(Struch et al., 2019), and ten species of grouper (Pulster et al., 2020a). This documented increase in PAH exposure has contributed to growing concern regarding the potential adverse health effects in fish as well as the continued degradation of the GoM ecosystem as a whole. In order to help understand the impacts of chronic PAH exposures on Red Snapper, we developed this study with the objectives to (1) survey hepatobiliary PAH concentrations throughout their range in the GoM; (2) evaluate temporal trends in PAH exposures in the DWH impacted area; and (3) correlate hepatic PAH concentrations with hepatobiliary health of Red Snapper in the GoM.

2. Materials and methods

2.1. Field collections

Red Snapper ($n = 566$) were collected from 72 locations during May through September of 2011–2017. Fishing was conducted using demersal longline sampling gear along transects throughout the GoM with transect locations divided into six regions of the GoM designated as the north central (NC), West Florida Shelf (WFS), Yucatan Shelf (YS), Bay of Campeche (BC), southwest (SW), and northwest (NW) regions (Fig. 1). Biometrics were recorded upon retrieval (Table A1), bile and hepatic tissues were collected from a subsample of Red Snapper for PAH analyses and histopathology. The demersal longline sampling design and protocols have been previously described in detail (Murawski et al., 2018; Pulster et al., 2020a) and brief sampling specifics can be found in the Appendix.

2.2. Biliary and hepatic PAH analysis

Untreated bile samples (3 μL) from Red Snapper ($n = 496$) were analyzed for naphthalene, phenanthrene and benzo[*a*]pyrene metabolite equivalents (also known as fluorescent aromatic compounds, FACs) using high performance liquid chromatography fluorescence detection (HPLC-F). Biliary FACs are expressed as ng FAC g^{-1} bile and reported as the sum of all three PAH metabolite equivalents (TFAC) rounded to two significant figures. Homogenized hepatic tissues ($n = 297$) were prepared for analysis using QuEChERS extraction and clean-up methodology (Bond Elut, Agilent Technologies, Santa Clara, CA, USA). Target PAHs in liver tissues were confirmed and quantified using a gas chromatograph (GC, Agilent 7890B) coupled to a triple quadrupole mass spectrometer (MS/MS, Agilent 7010) operating in multiple reaction monitoring (MRM) mode. Hepatic PAHs are reported as the sum of 46 PAH analytes ($T_{46}PAH$) rounded to three significant figures expressed as ng g^{-1} (w.w.). The 2–3 ring and 4–6 ring PAHs and alkylated homologs were summed as low molecular weight (LMW) and high molecular weight (HMW) PAHs, respectively. Further details on specific methods, instrument parameters and quality assurance protocols are provided in the Appendix.

2.3. Health proxies

Providing Red Snapper hepatic samples had sufficient mass for both PAH analysis and lipid extractions, lipid content was measured using a modified Folch method described in Pulster et al. (2020b). Fulton's Condition Factor [K = (total weight (g) / standard length (cm^3) x 100] and hepatic (HSI), gastrointestinal (GaSI) and gonadal (GSI) somatic indices [SI = organ weight (g) / total weight (g)] were calculated for each individual fish.

2.4. Histopathology

Gross and histological evaluations were performed on 152 hepatic samples from 17 locations in the NC ($n = 91$), NW ($n = 57$), and SW ($n = 4$) regions of the GoM collected 2012–2017. In the field, three representative segments were taken from the middle and distal portions of the liver lobes from each individual. Microscopic examination was performed using a Nikon Eclipse 80i (Nikon, Minato, Tokyo, Japan) and photomicrographs were taken with an Accu-scope Excelis HD (Commack, NY). Specifics on the tissue preparation and analysis are located in the Appendix.

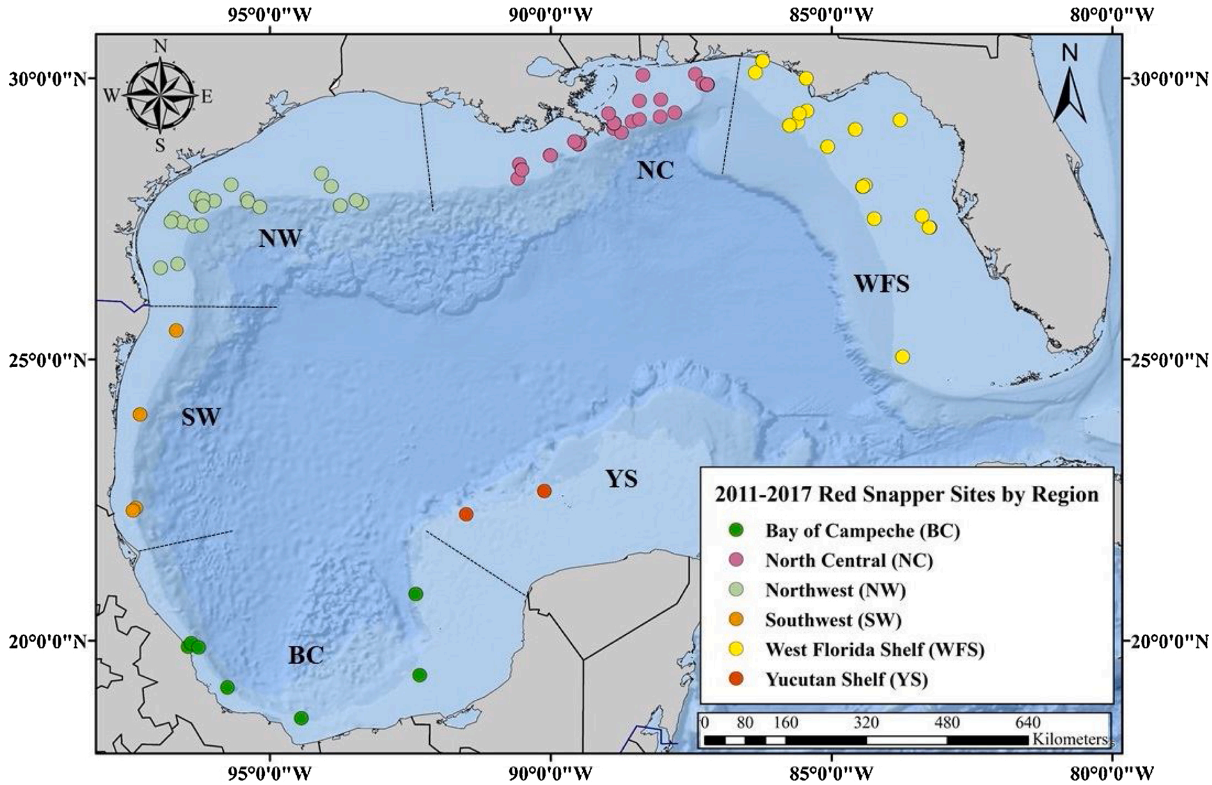

Fig. 1. Collection locations for Red Snapper (n = 566) used in this study. Dashed lines indicate the delineations between regions.

2.5. Statistical analyses

All statistical analyses were performed using JMP Pro Version 14.3 (SAS Institute) and MATLAB R2019b (Update 3, MathWorks) with the Fathom Toolbox. The Appendix provides more in-depth detail on the methodology used for data analyses.

3. Results

3.1. Associations between PAH concentrations and biometrics

Hepatic PAHs and biliary FACs in Red Snapper ranged from 192 to 8530 ng g^{-1} w.w. and 480 to 1,100,000 ng FAC g^{-1} bile, respectively (Table 1, Fig. 2). No significant relationships were found between biliary FACs and liver lipid content (Table 2). Weak (r = 0.30–0.50) to

Table 1
Mean (± standard deviation) and total biliary fluorescent aromatic compounds (TFAC) and hepatic polycyclic aromatic hydrocarbon (T_{46}PAH) concentrations measured in Red Snapper collected in the Gulf of Mexico, 2011-2017.

Region	Year	Biliary FACs (ng FAC g^{-1} bile)				Hepatic PAHs (ng g^{-1} w.w.)		
		Nap	Phn	B[a]P	TFACs	LMW	HMW	T_{46}PAH
Bay of Campeche (BC)	2015	22,000 ± 11,000	N/A	170 ± 80	22,000 ± 11,000	1870 ± 1190	19.1 ± 6.38	1890 ± 1190
	2016	44,000 ± 49,000	11,000 ± 9000	110 ± 70	55,000 ± 58,000	1022 ± 546	9.84 ± 3.08	1031 ± 546
	2015–2016	28,000 ± 26,000	11,000 ± 9000	160 ± 80	31,000 ± 32,000	1480 ± 1012	14.9 ± 6.91	1500 ± 1016
North Central (NC)	2011	120,000 ± 78,000	21,000 ± 12,000	280 ± 140	140,000 ± 90,000	1203 ± 977	9.39 ± 7.09	1210 ± 979
	2012	60,000 ± 27,000	13,000 ± 4300	220 ± 150	74,000 ± 30,000	1240 ± 701	6.69 ± 2.50	1250 ± 701
	2013	56,000 ± 26,000	13,000 ± 6000	450 ± 330	69,000 ± 32,000	2640 ± 2600	9.21 ± 5.74	2650 ± 2600
	2014	76,000 ± 70,000	N/A	440 ± 270	76,000 ± 70,000	955 ± 375	5.23 ± 2.13	960 ± 375
	2015	65,000 ± 61,000	N/A	340 ± 270	65,000 ± 61,000	1540 ± 1110	9.99 ± 5.18	1550 ± 1110
	2017	140,000 ± 130,000	26,000 ± 17,000	210 ± 120	170,000 ± 150,000	1094 ± 300	4.54 ± 1.59	1099 ± 301
	2011–2017	80,000 ± 73,000	17,000 ± 11,000	350 ± 270	90,000 ± 81,000	1480 ± 1290	8.21 ± 5.12	1490 ± 1290
Northwest (NW)	2016	110,000 ± 85,000	19,000 ± 15,000	350 ± 530	130,000 ± 92,000	2330 ± 1460	10.2 ± 5.97	2340 ± 1470
	2017	220,000 ± 170,000	40,000 ± 22,000	240 ± 190	250,000 ± 190,000	N/A	N/A	N/A
	2016–2017	130,000 ± 120,000	23,000 ± 18,000	330 ± 480	150,000 ± 130,000	2330 ± 1460	10.2 ± 5.97	2340 ± 1470
Southwest (SW)	2016	130,000 ± 120,000	40,000 ± 44,000	170 ± 80	170,000 ± 150,000	1310 ± 988	8.43 ± 5.69	1320 ± 993
West Florida Shelf (WFS)	2011	N/A	N/A	N/A	N/A	724 ± 389	14.7 ± 6.91	738 ± 390
	2013	31,000 ± 24,000	5000 ± 2800	110 ± 90	36,000 ± 26,000	N/A	N/A	N/A
	2014	31,000 ± 22,000	8000 ± 5000	130 ± 70	37,000 ± 24,000	403 ± 113	6.44 ± 1.89	409 ± 113
	2015	10,000 ± 6900	N/A	100 ± 130	11,000 ± 7000	727 ± 397	12.1 ± 3.33	740 ± 397
	2017	210,000 ± 91,000	49,000 ± 23,000	250 ± 180	260,000 ± 110,000	N/A	N/A	N/A
	2011–2017	98,000 ± 110,000	27,000 ± 27,000	170 ± 150	120,000 ± 130,000	648 ± 367	12.5 ± 6.71	661 ± 369
Yucatan Shelf (YS)	2015	38,000 ± 43,000	N/A	74 ± 90	38,000 ± 43,000	659 ± 359	6.49 ± 5.15	666 ± 353

FAC = fluorescent aromatic compounds; Nap = Naphthalene, Phn = Phenanthrene, B[a]P = benzo[a]pyrene, TFAC = sum of Nap + Phn + B[a]P FACs, LMW = low molecular weight PAHs, HMW = high molecular weight PAHs, T_{46}PAH = total of 46 PAHs.

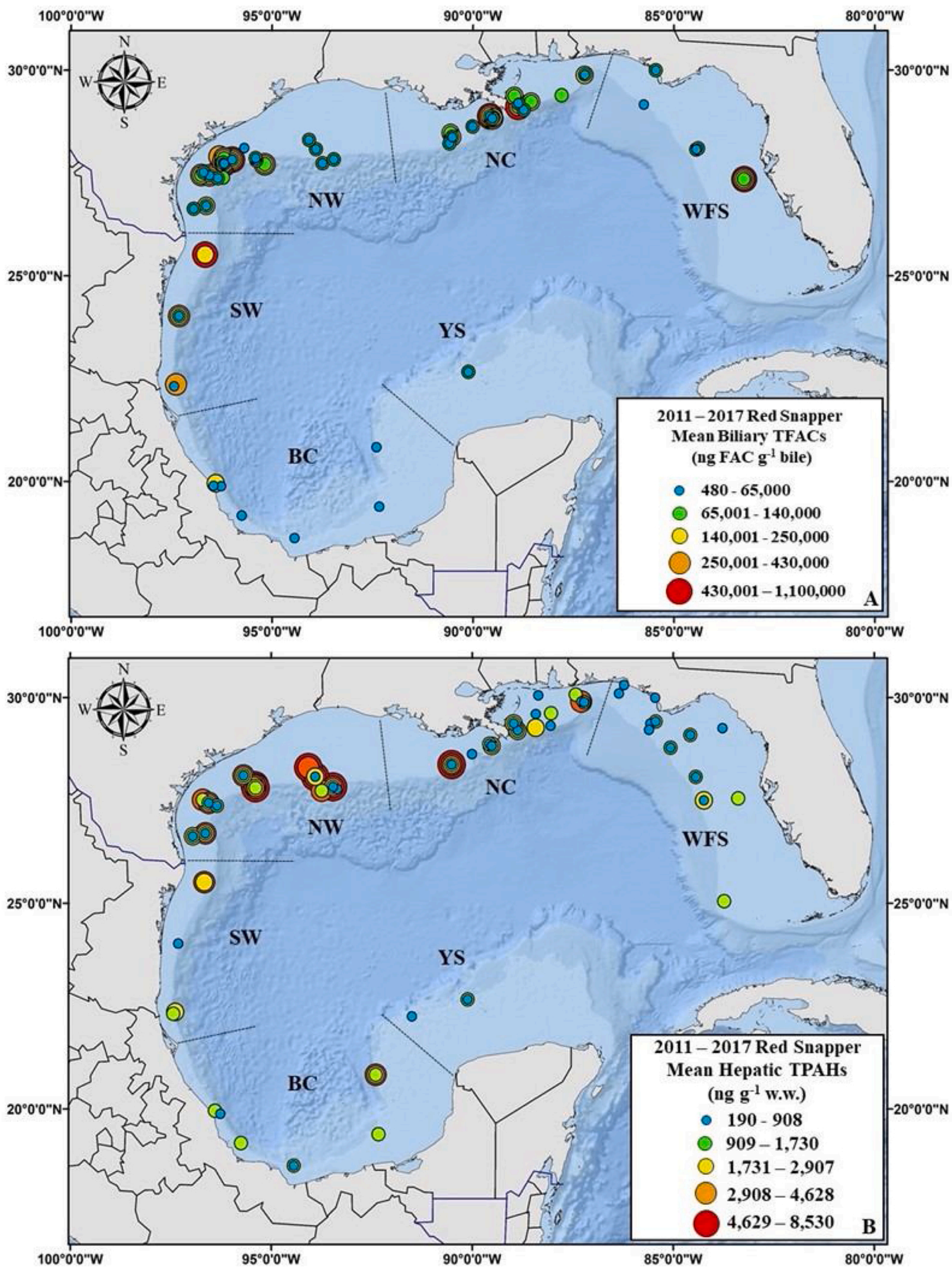

Fig. 2. Mean biliary total fluorescent aromatic compounds (TFACs, A) and hepatic total petroleum hydrocarbons (TPAHs, B) in Gulf of Mexico Red Snapper, 2011-2017. The dashed lines delineate between six regions designated as the West Florida Shelf (WFS), north central (NC), northwest (NW), southwest (SW), Bay of Campeche (BC), and Yucatán Shelf (YS).

moderate (r = 0.50−0.70) negative associations were found between hepatic lipids and both wet weight (r = -0.32, p < 0.0001) and lipid normalized (r = -0.49, p < 0.0001) PAH concentrations, therefore only wet weight concentrations are reported. Weak to very weak (r < 0.30) relationships were found between hepatobiliary PAHs and biometrics, somatic indices and condition factor (Table 2). There was also a weak positive relationship detected between biliary and hepatic PAHs concentrations (r = 0.21, p = 0.003). Weak to strong positive associations were detected between the HSI and GSI (r = 0.484, p = 0.007), HSI and GaSI (r = 0.904, p = 0.001) and GSI and GaSI (r = 0.416, p = 0.006) in Red Snapper collected Gulf-wide.

In the NW region, the total weight of female Red Snapper was significantly larger than males (p = 0.03). There were no other significant differences observed between sex and length (p = 0.55), weight (p = 0.29), or condition factor (p = 0.43) for Red Snapper collected Gulf-wide or within a particular region (all years combined). The total lengths (TL) of Red Snapper used in this study ranged between 31 and 101 cm (63 ± 13 cm) (Table A1). Based on the TL in this dataset, the estimated age range is between 2 and 25 years or older, with a mean of approximately 6 years old (Herdter et al., 2017; Patterson et al., 2001). Strong positive relationships were observed between standard length and total weight (r = 0.912, p = 0.001).

There were no differences in hepatobiliary PAH concentrations between male and female Red Snapper even though mean (± standard deviation) hepatic lipid content was significantly higher in males (0.49 ± 0.24 g, n = 169) compared to females (0.40 ± 0.18 g, n = 129, p = 0.003) for all Gulf-wide data combined over years and regions. Regionally, sex differences in hepatobiliary concentrations were only detected in the NC region, with males having significantly higher biliary FAC (104,000 ± 98,000 ng FAC g^{-1}, p = 0.021) and hepatic (1730 ±

Table 2
Prevalence (%) of microscopic hepatic changes (MHC) in Red Snapper collected by region (sample size in parentheses) in the Gulf of Mexico (GoM). Regions include the north central (NC), northwest (NW) and southwest (SW) GoM. The region with the highest prevalence for each MHC is bolded.

Microscopic Hepatic Change	NC (n = 91)	NW (n = 57)	SW (n = 4)	Gulf-wide (n = 152)
Lymphocytic inflammation	**60**	58	0	58
Parasites/nematodes/metazoans	**58**	54	25	55
Pigmented macrophage aggregates	57	**60**	25	57
Granuloma(s)	**57**	51	50	55
Cholangiofibrosis	47	**68**	25	55
Lipid-type vacuolar change	**46**	32	0	39
Glycogen-type vacuolar change	35	39	**50**	37
Edema	**34**	0	0	20
Myxozoan plasmodia	12	**21**	0	13
Bile duct hyperplasia	**10**	2	0	7
Hepatocellular atrophy	**9**	4	0	7
Hepatocellular necrosis	**8**	7	0	7
Ito cell hyperplasia	1	**9**	0	4
Hyperplastic nodule	**1**	0	0	1
Foci of cellular alteration (FCA)	0	**2**	0	1

1580 ng g^{-1} w.w., p = 0.009) PAH concentrations compared to females (75,000 ± 53,000 ng FAC g^{-1}; 1160 ± 665 ng g^{-1} w.w.).

3.2. Gulf-wide spatial trends in biometrics and PAH concentrations

For all years combined, regional differences using permanovas were detected in the standard lengths (F = 6.18, p = 0.001), total fish weight (F = 2.85, p = 0.019), HSI (F = 6.13, p = 0.001), condition factors (F = 17.70, p = 0.001) and hepatic lipid content (F = 36.42, p = 0.001). Pairwise permanovas determined the lipid content (p = 0.015) of NC Red Snapper were significantly larger than those along the WFS. The HSI (p = 0.015) and GSI (p = 0.015) of Red Snapper collected in the NC region were significantly larger than those collected in the NW region. Additionally, Red Snapper in the NC region had a lower condition factor compared to the NW (p = 0.015), WFS (p = 0.015), SW (p = 0.045) and BC (p = 0.015) regions. Red Snapper collected along the WFS had significantly larger HSI than those collected in the NW (p = 0.015) and SW (p = 0.015) regions as well as significantly larger GSI than the NW (p = 0.03).

In 2013 (p = 0.002), 2014 (p = 0.012), and 2015 (p = 0.018) mean biliary FACs measured in Red Snapper from the NC region were significantly higher than Red Snapper collected along the WFS. In 2015, mean biliary and hepatic concentrations in the NC region were also significantly higher than fish collected in the BC (p = 0.024) and WFS (p = 0.042) regions, respectively. No other spatial trends were detected in hepatobiliary PAH concentrations.

3.3. Temporal trends in the North Central Region

The current study expands upon an earlier study documenting the decline in biliary FACs in Red Snapper collected in the NC region of the GoM between 2011 and 2013 (Snyder et al., 2015) by providing an extended time series for biliary FAC concentrations (2014–2017) and including PAH levels for hepatic tissues (2011−2017). Temporal differences detected in the biometrics of Red Snapper collected in the NC region are summarized in Table A1. The HSIs of Red Snapper collected in the NC region during 2014 were significantly higher than Red Snapper collected in the same region during 2011 (p = 0.015), 2012 (p = 0.015), 2013 (p = 0.03), 2015 (p = 0.015), and 2017 (p = 0.015). The 2012 GSIs were significantly lower than in 2014 (p = 0.015) and 2017 (p = 0.015). There were no significant differences observed between years for the GaSIs in the NC region. In general, all somatic indices for Red Snapper in the NC region increased between 2011 and 2014 (HSIs: 814 %, GaSIs: 190 %, GSIs: 292 %) followed by decreases by 2017 (HSIs: -88 %, GaSIs: -68 %, GSIs: -59 %). Condition factors in 2017 were significantly higher than 2011 (p = 0.015), 2012 (p = 0.03), 2013 (p = 0.015) and 2015 (p = 0.03) and exhibited an overall 43 % increase between 2011 and 2017.

Mean biliary FACs for NC Red Snapper were significantly higher in 2011 compared to 2012 (p = 0.015), 2013 (p = 0.015), 2014 (p = 0.045) and 2015 (p = 0.015; Table 1). The 2017 mean biliary FAC concentration was significantly higher than 2012 (p = 0.015). In general, mean biliary FACs in the NC region declined 51 % between 2011 and 2013, and remained relatively stable until 2015, when a sharp increase (162 %) in concentrations was observed in 2017.

Although there were no significant temporal trends observed in hepatic PAH concentrations in Red Snapper collected in the NC region of the GoM (Table 1), annual differences were detected in the hepatic profiles using discriminant analysis (Hotelling-Lawley p < 0.0001, Fig. 3). Approximately 82 % of the variability in the profiles is explained by year as a grouping variable for the factor 1 (χ^2 (3815) = 556, p < 0.0001) and 2 (χ^2 (2974) = 540, p < 0.0001) loadings, consisting of primarily the LMW and HMW compounds, respectively (Fig. 3 and A1).

3.4. Temporal trends on the West Florida shelf

There were no significant sex-related differences in standard lengths, total weight, organ weights, condition factor, hepatic lipid content, biliary FACs or hepatic PAHs in the WFS region for all years combined or within a particular year. Mean biliary FACs were significantly higher in 2017 compared to 2013 (p = 0.006), 2014 (p = 0.006) and 2015 (p = 0.006) (Table 1). Mean hepatic PAH concentrations were significantly higher in 2011 compared to 2014 (p = 0.001) (Table 1). Overall, mean biliary FACs in Red Snapper along the WFS increased 625 % between 2013 and 2017 whereas hepatic PAH concentrations decreased 45 % between 2011 and 2014, followed by an 81 % increase by 2015.

There were no significant annual differences in the HSIs or GSIs in Red Snapper collected along the WFS, however the GaSIs increased 64 % between 2011 and 2014 (p = 0.01) and then decreased 31 % by 2015 (p = 0.21). Condition factors of Red Snapper along the WFS declined 22 % between 2011 and 2013 (p = 0.04) and then increased 36 % by 2017 (p = 0.010).

3.5. Gross evaluation

Red Snapper (n =152) collected from three regions of the GoM (NC,

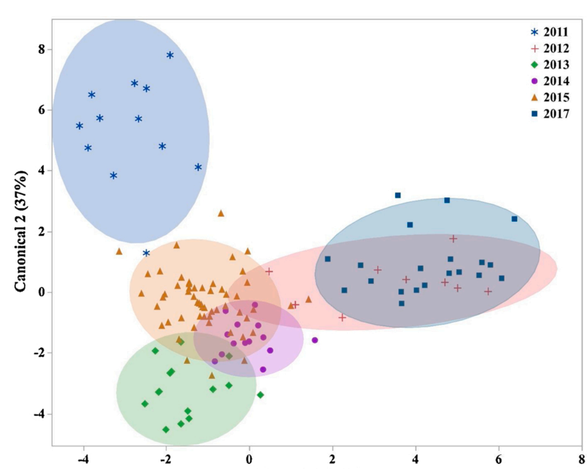

Fig. 3. Discriminant analysis plot illustrating distinct clustering of PAH composition profiles by year in liver tissues of Red Snapper located in north central Gulf of Mexico. Ellipse indicate 90 % data coverage for each year.

NW, SW) between 2012 and 2017 were evaluated for gross and histological changes in their hepatic tissues. Gross evaluation of Red Snapper hepatic tissues (n = 152) revealed minimal morphologic variation. In general, hepatic tissues were comprised of homogenous soft tissue with a smooth capsular surface and no lobular delineation. Interspersed throughout the soft tissue were blood vessels as well as biliary ducts. Blood vessels were often prominent and contained dark brown blood clots. The parenchyma ranged from light tan to dark tan and rare samples had pinpoint to 1 mm dark brown or copper foci scattered in the hepatic capsule or throughout the parenchyma.

3.6. Histological evaluation

Histological evaluation of the hepatic tissues revealed a spectrum of hepatocellular morphologic variation in association with vacuolar change ranging from mild to severe and only one fish had a liver with no microscopic hepatic changes (MHCs). The normal microscopic anatomy of the liver is described in more detail in the Appendix. Microscopic hepatic changes were noted in 99 % of the fish evaluated (Table 2). Each of the fish identified with changes had at least one MHC and as many as nine were noted for an individual fish. The 15 types of MHCs observed in GoM Red Snapper can be broadly categorized into: (1) inflammatory responses (granulomas, chronic inflammation, lymphocytic inflammation, PMAs, fibrosis, edema, parasites); (2) metabolic responses (hepatocellular vacuolization, atrophy); (3) degenerative responses (necrosis); (4) nonneoplastic proliferative changes (FCAs, bile duct hyperplasia, Ito cell hyperplasia); and (5) neoplastic lesions (hyperplastic nodule). A single occurrence of hepatocellular hyperplasia was noted in the NC region (Fig. 4a).

Lymphocytic inflammation was commonly observed in the peripancreatic connective tissue, adventitial tissue of the bile ducts, wall of vessels, or randomly distributed throughout the hepatic cords (Fig. 4b). There was a moderate correlation between lymphocytic inflammation and parasites (R^2 = 0.49, p = 0.0190). Fifty-eight percent of the Red Snapper identified with lymphocytic inflammation (n = 88) had concurrent granulomas containing organisms that were interpreted as larval nematodes (Fig. 4c). Granulomas were characterized by a center of necrotic cellular debris that variably contained multinucleated nematode larvae surrounded by concentric layers of flattened macrophages or a thin layer of fibrous connective tissue (Fig. 4c-d). Rare sub adult to adult nematodes were observed within the connective tissues adjacent to the pancreas or hepatic serosa (Figure A3).

Intra-biliary duct myxozoan plasmodia were observed in 13 % of the Red Snapper (Figure A3). Plasmodia were serpiginous, filled with flocculent eosinophilic material and had various stages of developing myxospores. Parasitic plasmodia were observed in samples from the NC and NW regions. There was moderate cholangiofibrosis (bile duct fibrosis) in the affected fish (Fig. 4d). Cholangiofibrosis was variably

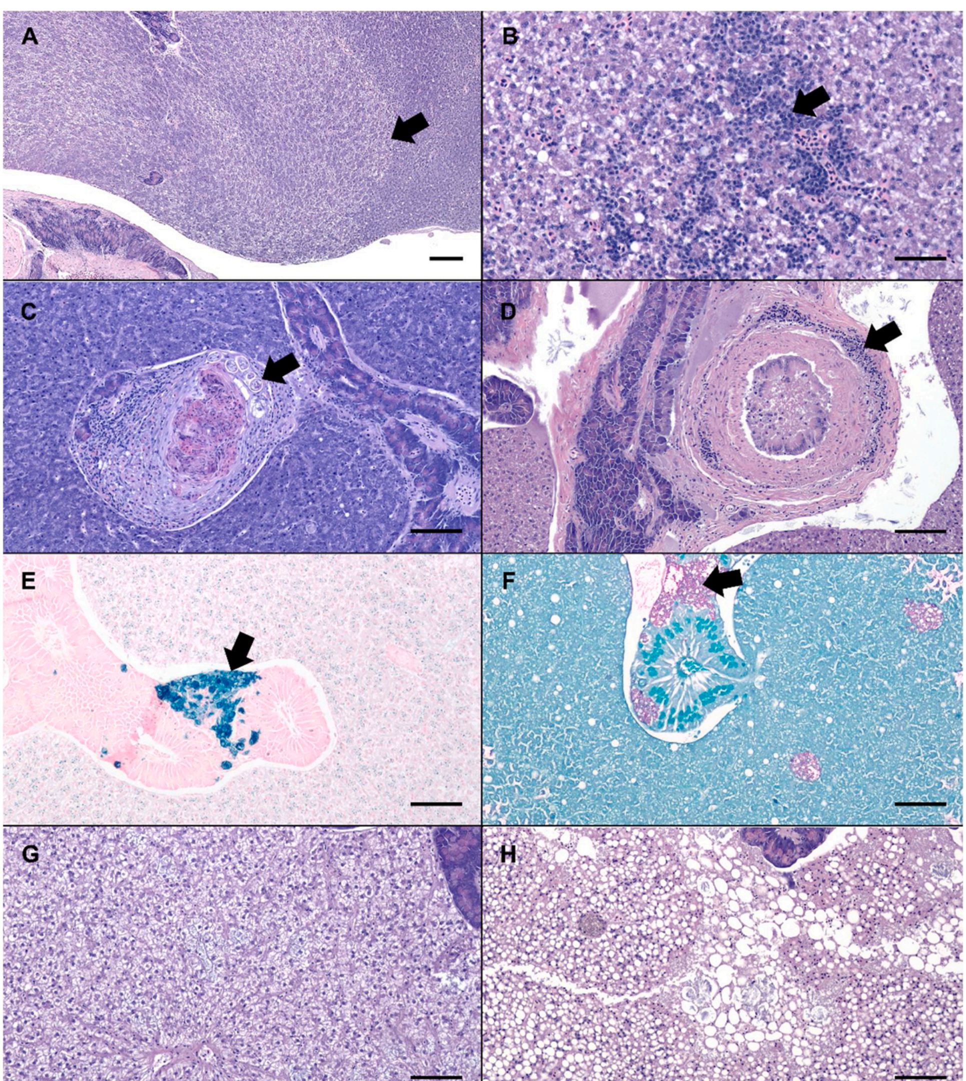

Fig. 4. Photomicrograph of (A) hepatocellular nodular hyperplasia (arrow) (H&E stain, 40X magnification, 100 µm scale bar); (B) focus of lymphocytic inflammation (arrow) (H&E stain, 20X magnification, 25 µm scale bar); (C) peripancreatic multinucleated nematode larvae encysted in fibrous connective tissue at the periphery of a peripancreatic granuloma (arrow, H&E stain, 20X magnification); (D) bile duct necrosis with adventitial lymphocytic inflammation (arrow) and moderate peribiliary fibrosis (cholangiofibrosis, H&E stain, 100X magnification); (E) pigmented macrophage aggregates in the peripancreatic tissue as well as pigment within hepatocytes staining positive for hemosiderin (arrow, Perl's Iron stain, 100X magnification); (F) pigment within macrophage aggregates staining positive for PAS (arrow, PAS stain, 100X magnification; (G) severe glycogen type vacuolar change (H&E stain, 100X magnification); and (H) severe lipid type vacuolar change (H&E stain, 100X magnification). Scale bar for plates C-H is 50 µm; H&E: Hematoxylin and Eosin stain.

observed and severity often correlated with inflammation of the ducts or intraluminal organisms and was well-delineated by Masson's trichrome stain as bright blue fibers.

Pigmented macrophage aggregates (PMA) contained foamy basophilic cytoplasm pigmented with intracellular lipid droplets or hemosiderin/lipofuscin/ceroid (Fig. 4e). Pigment within some macrophages stained positive with Perl's Iron and PAS stain for hemosiderin (Fig. 4e-f) while for others the iron staining was negative (Figure A4). Sudan Black IV and AFB of the Perl's negative aggregates stains were both negative indicating for some samples the intracellular pigments were not lipofuscin, lipid, triglycerides, or lipoprotein (Figure A5−6).

Intra-hepatocellular cytoplasmic vacuolization consistent with lipid (39 %) and glycogen (37 %) type vacuolar change were also prevalent in Red Snapper Gulf-wide and ranged from minimal to severe (Fig. 4g). Lipid-type vacuolar change was characterized by intracytoplasmic, discrete round micro or macrovesicles that had distinct borders. Glycogen-type vacuolar change was characterized by mild cellular swelling and diffuse replacement of the normal eosinophilic granular hepatocytic cytoplasm by wispy, vacuolated cytoplasm (Fig. 4h).

Hepatocellular atrophy was only observed in Red Snapper collected in the SW (2016) and NC (2017) regions. Atrophic hepatocytes were decreased in size due to shrunken cytoplasmic contents (Figure A7). Atrophied hepatocytes often had deeply basophilic granular cytoplasm. Due to cytoplasmic loss, nuclei appeared closer in proximity. Most of the samples that had atrophy were considered minimal to mild except for one fish collected near the Mississippi delta that was moderate in severity.

3.7. Biometric associations with MHCs

While lipid-type vacuolar change had a moderate positive relationship with liver lipid content ($R^2 = 0.52$, $p = 0.012$), lipid-type vacuolar changes had a moderate inverse correlation with glycogen-type vacuolar changes ($R^2 = -0.68$, $p = 0.0005$). Male Red Snapper collected in the NW region ($\chi^2_{(1,\ n\ =\ 55)} = 9.53$, $p = 0.002$) and Gulf-wide ($\chi^2_{(1,\ n\ =\ 150)} = 3.81$, $p = 0.037$) had a higher prevalence of glycogen-type vacuolar change compared to females. Atrophy was more prevalent in females compared to males collected Gulf-wide ($\chi^2_{(1,\ n\ =\ 150)} = 4.99$, $p = 0.027$) and in the NC region ($\chi^2_{(1,\ n\ =\ 91)} = 7.546$, $p = 0.008$). The sex difference in atrophy was largely driven by the small sample size of males ($n = 2$) compared to females ($n = 8$). In the NC region, males also had a higher prevalence of bile duct hyperplasia ($\chi^2_{(1,\ n\ =\ 91)} = 3.86$, $p = 0.0486$) and myxozoan plasmodia ($\chi^2_{(1,\ n\ =\ 91)} = 5.490$, $p = 0.017$) compared to females. No other sex-related differences were detected in the observed pathological changes either within a region or Gulf-wide. There were no correlations between standard length and weight (proxy for age) of Red Snapper with any of the observed MHCs, therefore, precluding age as a significant variable in propelling these changes.

3.8. PAH associations with MHCs

There were no relationships between biliary FACs and pathological changes observed for all data combined. However, there were significant differences within a region. Biliary FACs were moderately to strongly correlated with the prevalence of parasites ($R^2 = 0.67$, $p = 0.0498$) in the NW region and intracellular hemosiderin, lipofuscin and ceroid laden macrophages ($R^2 = 0.72$, p = 0.008) in the NC region. Additionally, the hepatic PAH concentrations were strongly associated with the prevalence of myxozoan plasmodia in Red Snapper Gulf-wide ($R^2 = 0.72$, $p = 0.001$), as well as in the NW ($R^2 = 0.76$, $p = 0.018$) and NC ($R^2 = 0.64$, $p = 0.025$) regions.

3.9. Regional patterns in MHC prevalence

The NC GoM Red Snapper had a higher prevalence of edema ($\chi^2_{(2,\ n\ =\ 152)} = 25.33$, $p < 0.0001$) and lipid-type vacuolar change ($\chi^2_{(2,\ n\ =\ 152)} = 6.79$, $p = 0.033$) compared to those in the NW and SW regions. The most severe cases of lipid- and glycogen-type changes were fish collected in the NC and NW region of the Gulf. Cholangiofibrosis was more prevalent in fish located in the NW ($\chi^2_{(2,\ n\ =\ 152)} = 6.923$, $p = 0.031$) GoM compared to those from the NC region.

3.10. MHC patterns in the north central time series

Both decreasing and increasing trends were observed in the various MHCs observed in Red Snapper (Fig. 5). Cholangiofibrosis (-29 %, $p = 0.011$) and edema (-81 %, $p < 0.0001$) decreased over time (2011−2017) in fish from the NC region of the GoM (Figure 56a). Conversely, considerable increases over time were observed for PMAs (60 %, $p = 0.008$), hepatocellular atrophy ($p < 0.0001$), and hemosiderin, lipofuscin and ceroid laden macrophages (140 %, $p = 0.001$; Fig. 5b). No other temporal trends were detected in MHCs in Red Snapper collected in the NC region of the GoM.

4. Discussion

In the GoM, this is the most comprehensive assessment of hepatobiliary PAH concentrations and microscopic hepatic changes in Red Snapper collected between 2011 and 2017. There were no trends

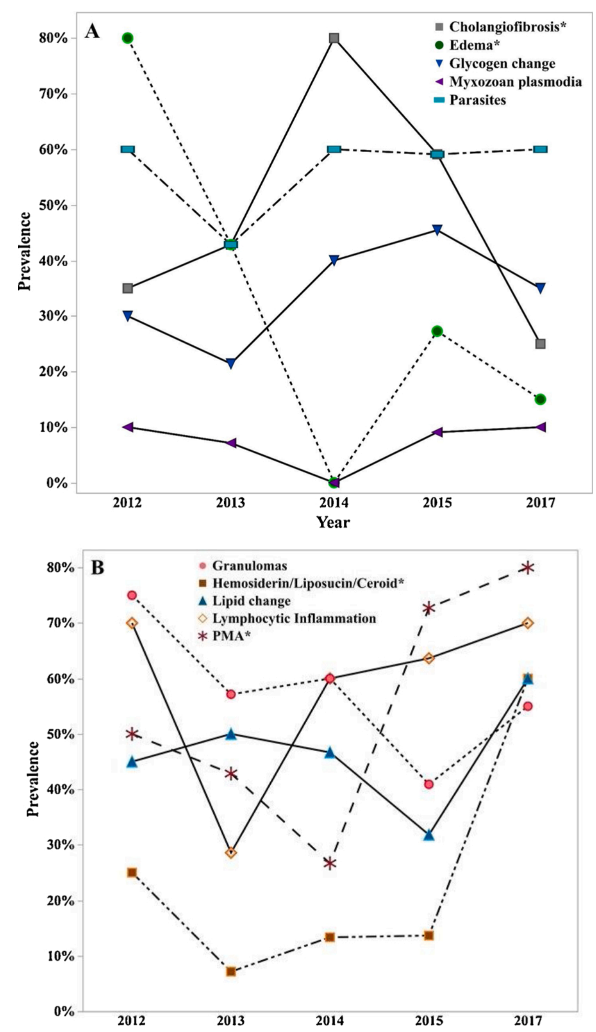

Fig. 5. Microscopic hepatic changes (MHCs) in Red Snapper collected in the north central region of the Gulf of Mexico that are demonstrating decreasing (A) and increasing (B) trends in prevalence over time (2011-2017). The MHC trends with an asterisk are significant.

observed in the PAH concentrations in hepatic tissues whereas biliary FAC levels declined after 2011, remaining relatively stable until 2015 which was followed by a significant increase in 2017. Additionally, 99 % of the Red Snapper evaluated had, on average, five coinciding MHCs and ranged between one to nine for an individual fish. The most significant changes observed through microscopic evaluation of the Red Snapper hepatic tissues included (in order of decreasing prevalence) lymphocytic inflammation, parasite burdens, PMAs, cholangiofibrosis and hepatic vacuolar changes. While some MHCs decreased over time in Red Snapper from the NC region, significant increasing trends in PMAs, hemosiderin, lipofuscin and ceroid laden macrophages and hepatocellular atrophy were observed. Hepatobiliary PAHs were associated with PMAs, parasitic infections and myxozoan plasmodia prevalence.

Considerable increases in mean biliary FACs were observed over time in Red Snapper from both the WFS (2013 to 2017: +625 %) and the NC region (2015 to 2017: +162 %). Patterns of increasing PAH exposures and concentrations have been observed in various matrices in the NC region following environmental disturbances (Diercks et al., 2018; Perez-Umphrey et al., 2018; Pulster et al., 2020a, d; Snyder, 2020; Struch et al., 2019; Turner et al., 2019). It is likely that the chronic inputs of petroleum hydrocarbons into the GoM are from many sources including riverine discharge, natural seeps, the resuspension of contaminated sediments (e.g. DWH), leaking infrastructure (e.g. Taylor MC20), and the thousands of accidental spills that occur annually in this region from active infrastructure, extraction and transportation activities (National Research Council, 2003; Kolian et al., 2015; Mason et al., 2019; Sun et al., 2018).

Distinct differences in the compositional profile of PAHs in Red Snapper liver tissues were also noted between 2011 and later years. Factor analysis revealed the distinction was mainly between LMW and alkylated PAHs (factor 1) and HWM compounds (factor 2), with the HMW PAHs more frequently detected in the 2011 liver samples compared to other years. The HMW PAHs are formed by the incomplete combustion of various fuels and organic matter which are mostly associated with anthropogenic sources. Thus, the in situ burning of surface oil following *DWH* is a plausible source for the HMW PAHs detected in the 2011 Red Snapper livers from the NC region. Alkylated PAHs and the LMW compounds were significant in the profiles for all other years, suggesting weathering, metabolic processes and increased bioavailability of LMW PAHs. This profile pattern dominated by LMW compounds in fish is consistent with other regional and global studies (El-Kady et al., 2018; Murawski et al., 2014; Pulster et al., 2020a, b; Snyder, 2020; Struch et al., 2019; Zhang et al., 2015). In the NC region, male Red Snapper had significantly higher hepatobiliary PAH concentrations compared to females. Higher contaminant concentrations in male fish has been previously attributed to increased biochemical responses during metabolism (e.g, mixed oxygenase function activity) and higher rates of swimming activity and energy expenditure (Gray et al., 1991; Madenjian et al., 2016; Stegeman et al., 1982). An additional explanation may be the documented maternal offloading of PAHs by spawning female fishes and has been suggested to occur in Red Snapper located in the NW GoM (Nicholson, 2019).

The weak yet significant associations found between hepatobiliary PAHs and biometrics, including some somatic indices, in this study suggests these variables have a negligible influence on hepatobiliary PAH concentrations. These results also suggest that PAH concentrations are likely not influencing the condition of Red Snapper, which is consistent with the conclusions reached for other species in the GoM (Pulster et al., 2020a, b; Snyder, 2020). Similar relationships were also observed between PAH metabolites and the condition factor of other species in areas with poor water quality, high levels of eutrophication, sedimentation and elevated PAH concentrations (Freire et al., 2020; Vives et al., 2004). Condition factors are commonly used as quantitative indicators of individual fish health and can be influenced by age, sex, season, stage of maturation, feeding status, type and quality of food consumed, amount of fat reserves, muscular development and the overall water quality, and environmental conditions. Although Red Snapper in the NC region were larger in length and weight than those collected in other regions, their condition factors were ≤ 39 % lower than most of the other regions. The declining condition factors, biometrics and somatic indices in Red Snapper from the NC region and along the WFS of the GoM may have been the result of annual variability in normal physiological parameters however they could also be a sub-lethal response to an acute stress (e.g., oil spill) or exposure to multiple stressors often experienced in this region (e.g., co-occurring xenobiotics, anoxic conditions, diet shifts, salinity and temperature changes). Declines in growth rates were observed in Red Snapper the first three years post-DWH (Herdter et al., 2017) and the condition factors of Golden Tilefish were still in decline as of 2017 (Snyder, 2020). These studies add supportive evidence that fishes in the NC region may be expressing sub-lethal responses to environmental stressors. Reduced growth rates or condition factors, are often interpreted as a depletion of energy reserves and commonly observed during periods of stress and therefore considered reliable indicators of stress in fish (Bonga, 1997; Geode and Barton, 1990; Schreck et al., 2016).

A single fish in the NC region near the Mississippi River Delta had a focal area of hepatocellular hyperplasia. Ito cell hyperplasia and rare FCAs were also observed in Red Snapper from this region which could potentially be a consequence of toxin exposure, although the prevalence was too low to detect correlations with PAHs levels. Exposure to carcinogenic or estrogenic compounds have been linked to altered hepatic foci, such as FCAs, and are considered to represent the first morphological stage of the neoplastic process in fish (Boorman et al., 1997). Age and sensitivity to chemical insult as well as species inclination for neoplastic transformation should be considered as factors in tumor development.

Compensatory adaptation could also play a role in the toxicity tolerance of the Red Snapper population in the GoM, an environment subjected to anthropogenic pollution and other human-related disturbances for at least a century. Convincing evidence is provided by thriving populations of Atlantic Killifish *(Fundulus heteroclitus)* in highly polluted estuaries along the US Atlantic coast (Whitehead et al., 2017). Rapidly evolved adaptation through the mutation of genes and the alteration of biotransformation pathways has led to the metabolic resistance, reduced chemical sensitivity and increased chemical tolerance in these populations. Although the short generation times of Atlantic Killifish (~4 years) allows for evolutionary change to parallel environmental change (Whitehead et al., 2017), compensatory adaption may be applicable to longer lived species like Red Snapper (~60 years) and should be further investigated since evolved adaption for chemical tolerance may be accompanied by physiological fitness costs. Therefore, the paucity of neoplasms observed in Red Snapper may be a consequence of adaptation to chronic toxicant exposure, age, concentration, frequency or duration of exposure.

More than half of the Red Snapper evaluated had PMAs (57 %), which can serve as an indicator of chronic stress, underlying disease, and/or the presence of parasites. The association between PMAs and stress or tissue damage could be confounded with infection (Steinel and Bolnick, 2017). The ubiquitously distribution of PMAs in vertebrates ensures they provide the first line of cell-mediated host defense against pathogens and can be functionally distinct depending on their precursory activation (Grayfer et al., 2018). Furthermore, PMAs are robust factories of cytokines, chemokines and lipid mediators, which act to potentiate and fine-tune inflammatory and adaptive immune responses (Hodgkinson et al., 2015). PMAs play a significant role in the activation of the immune response, antigen processing and immunological memory (Agius and Roberts, 2003). The key functions of PMAs in fish have been classified as (1) immune (humoral and inflammatory responses); (2) metabolic dumps for the storage and destruction of exogenous and endogenous substances; and (3) iron recycling (Steinel and Bolnick, 2017; Wolf and Wolfe, 2005; Wolke, 1992). Environmental and physiological changes can stimulate PMA response, underscoring the difficulty in interpreting PMA changes. Nonetheless, there are numerous reports demonstrating the increases in size and frequency of PMAs with pollutants and environmentally stressful conditions, indicating their utility as water quality indicators (Agius and Roberts, 2003).

In the present study, there was an increased prevalence of the hemosiderin, lipofuscin and ceroid laden macrophages in the hepatic tissues of Red Snapper in the NC GoM region which were associated with biliary FAC concentrations. Lipofuscin and/or ceroid accumulation has been observed in fish displaying a wide variety of pathological conditions, including nutritional deficiencies, bacterial and viral disease, and toxins (Agius and Roberts, 2003). Studies evaluating superoxide radicals produced by peritoneal macrophages from English sole (*Pleuronectes vetulus*) exposed to polycyclic aromatic compounds may provide support to the presence of oxidative stress and increase in ceroid/lipofuscin pigment in treated animals (Clemons et al., 1999).

Red Snapper in the GoM had a similar frequency of both glycogen- and lipid-type vacuolar change, complicating the interpretation. The loss or accumulation of hepatic glycogen and/or lipid can be a common morphological response to toxicity (Wolf and Wolfe, 2005), although these changes could also be a factor of normal physiological processes (e.g., spawning) or secondary to decreased body condition from starvation, stress, or more typically, concurrent with disease (Wolf and Wolfe, 2005). Both the accumulation or depletion of lipid and glycogen has been observed in fish either as a direct or secondary consequence following exposures to toxins, including PAHs at very low environmentally relevant concentrations (Marty et al., 2003; Miranda et al., 2008). Hepatic lipid storage in fish is an important energy reserve that can fluctuate seasonally as a response to spawning or periods of starvation. The high reproductive investment results in the mobilization of hepatic and muscle tissue lipid reserves that are transferred to the gonads for maturation and spawning. Reproductive assessment and energy status can be accurately represented by the gonadosomatic (GSI) and hepatosomatic (HSI) index, respectively. The majority of the Red Snapper in this study were collected in August, just following their peak spawning season (May-July) in the northern GoM. There was a significant positive association between the GSI and HSI for Red Snapper collected both Gulf-wide ($r = 0.484$, $p = 0.007$) and in the north central region ($r = 0.925$, $p = 0.001$), suggesting there does not appear to be a trade-off between energy reserves and gonad maturation and/or spawning. There were, however, significant temporal trends for both the HSI and the GSI for Red Snapper collected in the north central region suggesting changes in energy reserves over time. The elevated glycogen storage could have been replacement energy storage during periods of starvation and/or to compensate for the lipid remobilization during spawning while the elevated hepatic lipids may be the result of rebuilding the energy reserves post-spawning. However, further investigation is needed to determine the true cause of vacuolar change and the declining trends in somatic indices in Red Snapper.

While it is not uncommon to see low levels of parasites in wild caught fish, a pattern of infection was associated with biliary FAC concentrations in Red Snapper. The prevalence of myxozoan plasmodia in Red Snapper were also strongly associated with hepatic PAHs in this study. Nematodes specifically have been frequently described in a number of reef fishes from the GoM, including snappers, groupers and jacks (Montoya-Mendoza et al., 2017; Moravec et al., 2014). The prevalence of intermediate hosts, infection level of prey, and host susceptibility play a significant role in parasite infection rates. Fish experiencing decreased physiological condition or exposed to polluted waters tend to suffer from higher levels of parasitic infestations (Rohlenová et al., 2011). A recent study documented the effects of seasonal variability on host immunity and physiology as well as the different parasitic life-strategies that can influence the physiology of fish; activating various immunity pathways (Rohlenová et al., 2011). Some parasites (e.g. *Myxidium* sp.) are known to induce strong inflammatory responses responsible for tissue deterioration and parasite encystation that can interrupt normal organ function and reproductive potential in severe cases (Al-Jahdali and Hassanine, 2010). Additionally, a recent study provides compelling evidence for the strong associations between biliary neoplasia and myxozoan infections in White Perch (*Morone americana*) collected in Chesapeake Bay (Matsche et al., 2020).

Inflammation is a physiological process in response to tissue damage caused from infection, chemical irritation, and/or wounding (Philip et al., 2004). Additionally, chronic inflammation that is not caused by infection may also contribute to carcinogenesis (Philip et al., 2004). In Red Snapper, lymphocytic inflammation was observed in 58 % of the samples (Gulf-wide) and was commonly associated with the occurrence of parasitic infections. While both may be common, there is a vast amount of literature demonstrating the direct relationship between chronic inflammation and carcinogenesis (Lu et al., 2006; Philip et al., 2004).

In comparison to the NC benthic Golden Tilefish evaluated in a concurrent study (Snyder, 2020), Red Snapper had a lower prevalence of glycogen-type vacuolar change, hepatocellular atrophy and FCAs. Conversely, Red Snapper had a higher prevalence of lymphocytic inflammation, parasites, PMAs, granulomas, cholangiofibrosis, lipid-type vacuolar change, bile duct hyperplasia, hepatocellular necrosis, and Ito cell hyperplasia. These data support species-specific susceptibility to hepatic changes. A parallel study evaluating oxidative stress and immune system biomarkers in both Golden Tilefish and Red Snapper may provide some insight (Deak, 2020). Golden Tilefish from the NC region exhibited differential oxidative stress and immune system biomarker patterns as well as a possible indication of compensatory metabolism. These results combined with the chronic exposures to PAH contaminated sediments, suggest Tilefish may have increased resilience to environmental stressors and thus decreased susceptibility to disease and MHCs.

The biomarker patterns were more variable in Red Snapper with no indication that this species may be less resilient (Deak, 2020), however, there was an increased prevalence of several MHCs in Red Snapper compared to Golden Tilefish. It is possible the chronic exposure to PAHs in the NC region is causing a reduction in the oxidative stress response in Red Snapper through lower expression of the evaluated biomarkers. The reduced oxidative response combined with the increased and persistent PAH exposures in this region will result in cellular damage and a decreased metabolic capacity to react to toxins, which further increases the disease susceptibility in organisms (Adams and Sonne, 2013; Lushchak, 2011; Quintanilla-Mena et al., 2020). Additionally, hypoxia-induced DNA methylation and hormone regulation has been observed in Red Snapper (Rahman and Thomas, 2014). High levels of DNA hypomethylation have also been associated with chronic exposures to environmental contaminants, including hydrocarbons, and subsequently resulting in further genomic changes that have been linked with the development of disease and cancer (Mirbahai et al., 2011; Quintanilla-Mena et al., 2020).

The wide variety of MHCs observed in the liver tissues of Red Snapper have been described in other fishes, located in the GoM and globally, that were exposed to inorganic and organic pollutants, including PCBs, OCPs, PAHs and metals (Brown-Peterson et al., 2015; Freire et al., 2020; Gaber, 2013; Miranda et al., 2008). However, it is important to note that many of the microscopic changes have also been identified in other exposure and field studies as a result of age, sex, dietary status, or genetic predisposition. The observed prevalence of MHCs in Red Snapper could be a consequence of normal physiological changes exacerbated by chemical exposures or they could be secondary responses to other environmental stressors. Some of the observed MHCs were associated with hepatobiliary PAH concentrations but require further work to determine the role of PAHs and the induction of these MHCs in this species.

5. Conclusions

This study documents declining biometrics, somatic indices and condition factors combined with increasing levels of both PAH exposure and microscopic hepatic changes in Red Snapper from the NC GoM. Furthermore, 99 % of the Red Snapper evaluated Gulf-wide had one to nine microscopic hepatic changes with an average of five coinciding in an individual fish. While FAC exposures increased significantly in 2017, PAH liver burdens have remained relatively stable, indicating Red Snapper have been able to maintain metabolic clearance thus far. However, the continued increase in PAH exposures and the associated metabolism required, may have an energetic cost that is manifesting as one or more of the 15 MHCs observed in Red Snapper. The MHCs observed in Red Snapper can be characterized as inflammatory responses, metabolic responses, degenerative responses, nonneoplastic proliferative and neoplastic lesions that could be signaling potential disease progression. Many of the observed responses (i.e., declining growth rates and condition factors) and MHCs in Red Snapper are known secondary responses to stress in fish (Eissa and Wang, 2016). Considering there is a limited number of ways in which stressors can affect fish populations, the response of adult fish populations to pollution may be indistinguishable from the response to other stressors such as overfishing or eutrophication (Ryder and Edwards, 1985). An overabundance of co-occurring chemical substances in the aquatic environment, instead an overabundance of co-occurring chemical substances may have additive, synergistic, antagonistic or even potentiated effects. Pollution can elicit a broad range of physiological responses in the form of lethal, sub-lethal, metabolic or behavioral changes in fish. Subtle endogenous effects and health indices (like those described herein) measure the health of the ecosystem without necessarily being attributed to a particular chemical or specific concentration (Ryder and Edwards, 1985). Hepatic histopathology has been used as a valuable monitoring tool for decades with high ecological relevance, and has been proposed as the most reliable indicator of health impairment among aquatic species (Hinton et al., 2001; Yancheva et al., 2016). If we continue to ignore the secondary stress effects and responses observed in fish, then we will never fully understand the impacts and toxicity of chemicals (Bonga, 1997). Consequentially, the abovementioned range of non-neoplastic and neoplastic hepatic toxicities and declining health indices in Red Snapper serve as significant early warning signs and allow for regional comparisons of water quality, environmental health, and the detection of ecosystem trends, all of which can have population level implications.

Author contribution statement

EP assisted in sample acquisition, completed the comprehensive data analysis and wrote the manuscript with contribution from all coauthors. SF conducted all gross and histological evaluations. BC and JM assisted in sample acquisition and processing. SAM conceived and completed the study design and participated in all sample acquisitions and study oversight.

Declaration of Competing Interest

The authors declare no conflict of interest.

Acknowledgements

We thank the captain and crew of the R/V *Weatherbird II* and Florida Institute of Oceanography (FIO), S. Gilbert for logistical support and the “Mud & Blood” field teams from the USA and Mexico. We extend our gratitude to the G. Stuntz lab at Harte Research Institute (T. Topping and J. Wetz) and E. Peebles lab (SHELF Project) at the University of South Florida for additional sample collections. We appreciate the support of the Mexican and USA State Departments to allow international sampling efforts. Our appreciation is also extended to L. Brandenburg and M. Campbell for laboratory assistance. All sampling was conducted in accordance with Protocols IS00000515 and IS00003388 approved by the Institutional Animal Care and Use Committee at the University of South Florida. This research was supported by grants from The Gulf of Mexico Research Initiative, C-IMAGE I (# SA 12-10), C-IMAGE II (# SA 15-16), and C-IMAGE III (# SA 18-16). Data are publicly available through the Gulf of Mexico Research Initiative Information & Data Cooperative (GRIIDC) at https://data.gulfresearchinitiative.org (doi: 10.7266/n7-nmsy-tq94, 10.7266/RX6DR04 N, and 10.7266/N7 × 34W1J).

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.aquatox.2020.105714.

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