Report for the
Exxon Valdez Oil Spill Trustee Council

Studies on *Exxon Valdez* Lingering Oil: Review and Update on Recent Findings – February 2016

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Study History:

The EVOS Trustee Council (EVOSTC) has funded many studies related to the distribution, quantity, and weathering state of lingering, intertidal subsurface oil from the 1989 *Exxon Valdez* oil spill. These studies included: (1) locating remaining lingering oil, using field sampling and modeling; and (2) identifying factors that have slowed the natural removal of the oil. The emphasis of these studies has been the lingering subsurface oil in Prince William Sound; however, studies led by the U.S. Geological Survey also have monitored the lingering oil in the Gulf of Alaska, at National Park sites on the Kenai Peninsula and along the Alaska Peninsula.

In addition to studies of the oil itself, the EVOSTC has funded a large body of work evaluating exposure of marine life to lingering *Exxon Valdez* oil, as well as the effects of exposure on individuals, populations, and ecosystems. Longest-term and most comprehensive studies of lingering oil effects were conducted on sea otters and harlequin ducks. These species showed evidence of protracted exposure to lingering oil due to their occurrence in intertidal habitats and diets of benthic invertebrates, as well as population-level injury as a result of exposure.

The findings of these studies have been published in numerous reports and peer-reviewed publications. However, there is a need for a synthesis of the results to address key questions about what is currently known about the lingering oil from the *Exxon Valdez* oil spill. Thus the EVOSTC has requested this report to review past work and provide an update on the most recent findings.

Executive Summary:

At least 10,800,000 gallons of crude oil spilled from the *Exxon Valdez* into Prince William Sound in 1989, resulting in the stranding of oil on an estimated 2,100 kilometers of shoreline. Much of the stranded oil was removed by extensive cleanup efforts in 1989-1991 and natural removal processes. It was expected that remaining oil would be reduced to negligible amounts soon thereafter. However, observations indicated that oil, some of it only lightly weathered, remained in intertidal sediments of some beaches eight years after the spill, leading to concerns that lingering oil could continue to have harmful effects on fish and wildlife individuals and populations. This "lingering oil" and its effects have been the focus of extensive studies designed to: (1) evaluate the persistence, distribution, and state of lingering oil, (2) determine effects of lingering oil on biota, and (3) evaluate options for remediation of lingering oil. This report provides a synthesis of these studies and the most recent findings.

Oil remains on the shorelines of Prince William Sound and the Gulf of Alaska, but at a small number of sites. Spatial models were developed to predict where the oil is likely to occur, in addition to the known locations. Using a model accuracy of 80%, there could be any type of subsurface oil on 35 kilometers of shoreline; moderately and heavily oiled residues are estimated to occur on 9.4 kilometers, representing 0.45% of the original length of oiled shoreline. It is estimated that, based on data collected between 2001 and 2007, residual oil on the shoreline represented 0.25% of the total spill volume.

Much of the residual subsurface oil remains lightly weathered, indicating that it is sequestered in places where oxygen and nutrients are at levels too low to support microbial degradation. After more than 25 years of natural removal processes (sediment reworking on beaches, tidal flushing, and microbial degradation), it is expected that natural removal rates going forward will

be very low.

Recent monitoring of mussels and artificial membranes that absorb oil from the water column indicates that the residual oil is not being released in most areas (recent data only show very low amounts being released from a site in the Gulf of Alaska). In fact, levels of polynuclear aromatic hydrocarbons (PAHs; the components in oil that are most toxic) in mussels in Prince William Sound decreased to background levels around 2001.

Evidence of exposure to lingering oil was observed in a variety of marine species, although the duration of exposure varied widely. The species with the longest timeline indicating exposure via biomarker (cytochrome P4501A) induction, harlequin ducks, showed evidence of exposure through 2011, 22 years after the *Exxon Valdez* spill. More recent sampling of harlequin ducks in 2013 and 2014 showed no difference in biomarker induction between oiled and unoiled areas, suggesting that exposure had ceased. Consistent with findings above, these data indicate that lingering subsurface oil is no longer being released.

Effects of the *Exxon Valdez* oil spill were seen across many different taxa. A key finding of the large body of work funded by the EVOSTC was that there are many mechanisms by which the spill affected marine life, including direct toxic effects and an array of more subtle indirect effects. Another important result was the recognition that exposure to lingering oil had effects that spanned decades for some vulnerable species, particularly sea otters and harlequin ducks. These species feed on the bottom in the intertidal zone, where they could come in contact with subsurface oil. Sea otters and harlequin ducks showed population injury for two decades after the spill. However, the most recent studies have shown that sea otters and harlequin ducks have recovered, with population attributes returning to pre-spill conditions. All indicators suggest that lingering oil is no longer causing ecological damage.

Surveys of the subsistence use of fish and wildlife in the communities affected by the *Exxon Valdez* oil spill found that by 2014, subsistence harvests in Chenega Bay, Cordova, and Tatitlek were lower and less diverse than in most post-spill years. However, the reasons for these changes include a range of cultural, economic, and environmental explanations, some of which are linked to the oil spill, but many are not. A strong majority of the respondents expressed confidence that subsistence foods are safe to eat. However, many respondents stated that youth are not learning subsistence skills, elders are not engaged in transmitting essential knowledge and values, many natural resource populations have declined or are difficult to access, and the traditional way of life has not recovered from the effects of the spill.

Once it was understood that oxygen and nutrients were limiting the weathering rate of the subsurface oil in the gravel beaches, extensive field studies were conducted to develop effective ways to inject oxygen and nutrients into the oiled sediment layer. It was thought that this bioremediation method would be less intrusive than manual or mechanical removal. These studies showed that injection of oxygen and nutrients could speed the rate of oil degradation, but only under certain conditions. Thus, of the 63 sites identified as candidates for remediation, only 9 sites had conditions where bioremediation was likely to be effective. The other sites would require manual or mechanical removal of the oiled sediments below the surface.

The EVOSTC continues to monitor the lingering oil, to document its physical and chemical changes over time. However, the oil that remains in the subsurface sediments is expected to persist for decades. The evidence indicates that there are no longer any biological effects of the oil that is sequestered in the beaches, thus there is no ecological basis for active remediation. However, the EVOSTC continues to evaluate the costs, risks, and benefits of remediation options for the remaining oil to subsistence users, recreational users, and the public in general.

Key Words:

Lingering oil, subsurface oil, *Exxon Valdez* oil spill, Prince William Sound,

Citation:

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LINGERING OIL STUDIES:
REVIEW AND UPDATE ON RECENT FINDINGS – FEBRUARY 2016

INTRODUCTION AND SCOPE OF REPORT

At least 10,800,000 gallons of crude oil spilled from the *Exxon Valdez* into Prince William Sound in 1989 (Wolfe et al. 1994). In the days and weeks after the *Exxon Valdez* ran aground on Bligh Reef in northeastern Prince William Sound, spilled oil moved primarily south and west through Prince William Sound and then into the Gulf of Alaska (Galt et al. 1991). An estimated 40% of the spilled oil landed on beaches within Prince William Sound (Galt et al. 1991), affecting 783 kilometers (km) of shoreline (Short et al. 2004). The oil spread along the Kenai Peninsula and the Gulf of Alaska, affecting a total of 2,100 km of shoreline (Owens 1991). Based on shoreline cleanup assessment technique (SCAT) surveys, the extent and degree of oiling on shorelines decreased rapidly over the first few years after the spill (Neff et al. 1995). Given observed rates of depletion, it was expected that remaining oil would be reduced to negligible amounts soon thereafter (Neff et al. 1995). However, observations indicated that oil, some of it only lightly weathered, remained in intertidal sediments of some beaches eight years after the spill (Hayes and Michel 1999), leading to concerns that lingering oil could continue to have harmful effects on fish and wildlife individuals and populations.

Since that time, an extensive body of work has been conducted to: (1) evaluate the persistence, distribution, and state of lingering oil, (2) determine effects of lingering oil on biota, and (3) evaluate options for remediation of lingering oil. This report summarizes those studies and provides an update to a similar report delivered to the *Exxon Valdez* Oil Spill Trustee Council (EVOSTC) in 2010 (Michel and Esler 2010).

DISTRIBUTION, QUANTITY, AND STATE OF LINGERING *EXXON VALDEZ* OIL IN INTERTIDAL HABITATS

The EVOSTC has funded many studies related to the distribution, quantity, and weathering state of lingering subsurface oil from the 1989 *Exxon Valdez* oil spill. These studies included: (1) locating remaining lingering oil, using field sampling and modeling; and (2) identifying factors that have slowed the natural removal of the oil. The emphasis of these studies has been the lingering subsurface oil in Prince William Sound; however, studies led by the U.S. Geological Survey have monitored the lingering oil in the Gulf of Alaska, at National Park sites on the Kenai Peninsula (1 site) and along the Alaska Peninsula (5 sites).

In this section, we summarize studies conducted to date to answer the following questions:

- Where is the lingering oil?
- Why is the oil still there?
- How much oil lingers?
- How weathered is the lingering oil?
- Is the lingering oil bioavailable?
- What is the long-term fate of the lingering oil?

Where is the lingering oil?

In 2001, it was known that relatively unweathered oil remained at some locations that were heavily oiled initially (Hayes and Michel 1999). The extent of remaining oil was unknown, and this uncertainty raised concerns about the effects that lingering oil may have on humans and on fauna that may become exposed to the oil. Thus, in 2001, lingering oil studies in Prince William Sound began with a project by the NOAA Auke Bay Laboratory to address these concerns by providing a quantitative estimate of the amount of shoreline (length, area) that remained contaminated and the amount oil remaining. The NOAA project team randomly selected 91 beaches that had been described as heavily or moderately oiled in 1989-1991 based on Shoreline Cleanup Assessment Technique (SCAT) surveys. In 100-meter long segments at 124 locations, they randomly selected 96 locations in the intertidal zone to dig pits to describe the current oiling conditions. The project found subsurface oil at 42 of the 91 beaches. These data were used to estimate that there were 19.3 acres (the range due to uncertainty was 10.0-31.4 acres) of remaining subsurface lingering oil in Prince William Sound. The mass of remaining subsurface oil was estimated at 55,600 kilograms (kg) (range of 26.1-94.4 kg), which is about 0.2% of the spill volume (Short et al. 2004). In 2003, an additional 32 sites were surveyed to extend the pits into the lower intertidal zone (Short et al. 2006). Short et al. (2007) also estimated that the areal extent of oiled beaches in Prince William Sound had not changed significantly between 2001 and 2005, indicating that the rate of decline had slowed to ~4% per year.

The NOAA Auke Bay Laboratory studies did not map out specific locations where the subsurface oil was, and it was not possible to search every possible location. Also, there was interest in expanding this kind of assessment to the Gulf of Alaska. Therefore, a study (aka “finding the lingering oil”) was conducted to develop a model that could be used to map out the most likely locations of subsurface oil. To properly train the model, field data needed to be collected from a wider range of oiling conditions. Therefore, in 2007, data were collected from 108 segments in Prince William Sound and 32 segments in the Gulf of Alaska using similar methods as Short et al. (2004). Subsurface oil was observed in pits at 13 of the 140 (9%) beach segments investigated in 2007 and only in Prince William Sound. In 2008, 32 smaller segments were visited, as part of the model validation.

When all the data from the lingering oil surveys were combined (Figure 1 shows the location of all the segments in Prince William Sound), subsurface oil was observed on 88 of 307 (29%) distinct beach segments investigated between 2001 and 2007 and was found in 509 of a total of 13,734 (4%) pits.

The average thickness of subsurface oiled layers in Prince William Sound ranged from 4.1 to 20.7 centimeters (Figure 2). The heavily oiled residues were the thickest, followed by moderately oiled residues; these were also shallower compared to lightly oiled residues and oil film or sheen layers. Half of the oiled pits contained moderately oiled residues. Of pits with subsurface oil, 18% (94 pits) were in the upper intertidal zone, 75% (378 pits) were in the middle intertidal zone, and 7% (35 pits) were in the lower intertidal zone (Figure 3). Figure 4 is a photograph of the northern shoreline of Smith Island showing the approximate locations of these tidal zones.

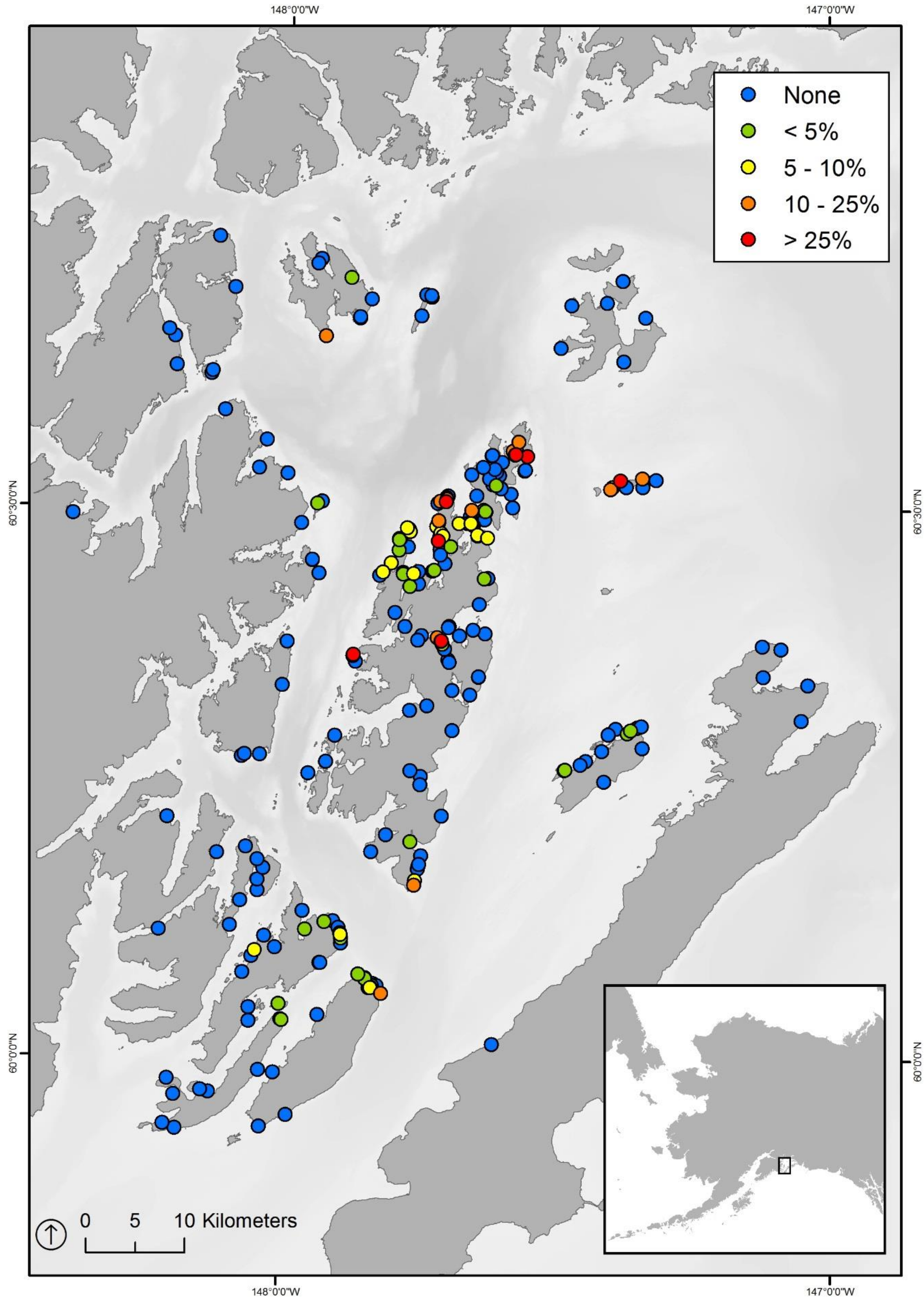


Figure 1. Location of shoreline segments symbolized by the percentage of pits with subsurface oiling in Prince William Sound for surveys conducted in 2001, 2003, and 2007.

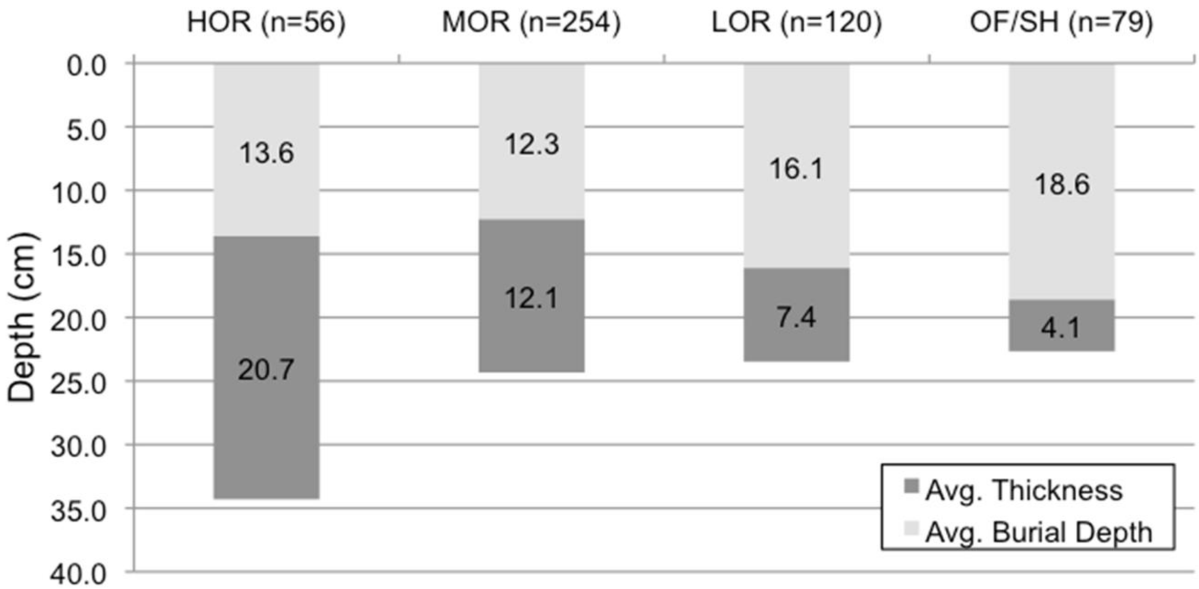


Figure 2. Average thickness of subsurface oiled layers and depth in centimeters below the surface categorized by oiling descriptor and based on data from pits in Prince William Sound surveyed in 2001, 2003, 2007, and 2008 (total of 509 pits). HOR = heavily oiled residue; MOR = moderately oiled residue; LOR = lightly oiled residue; OF = oil film; SH = oil sheen.

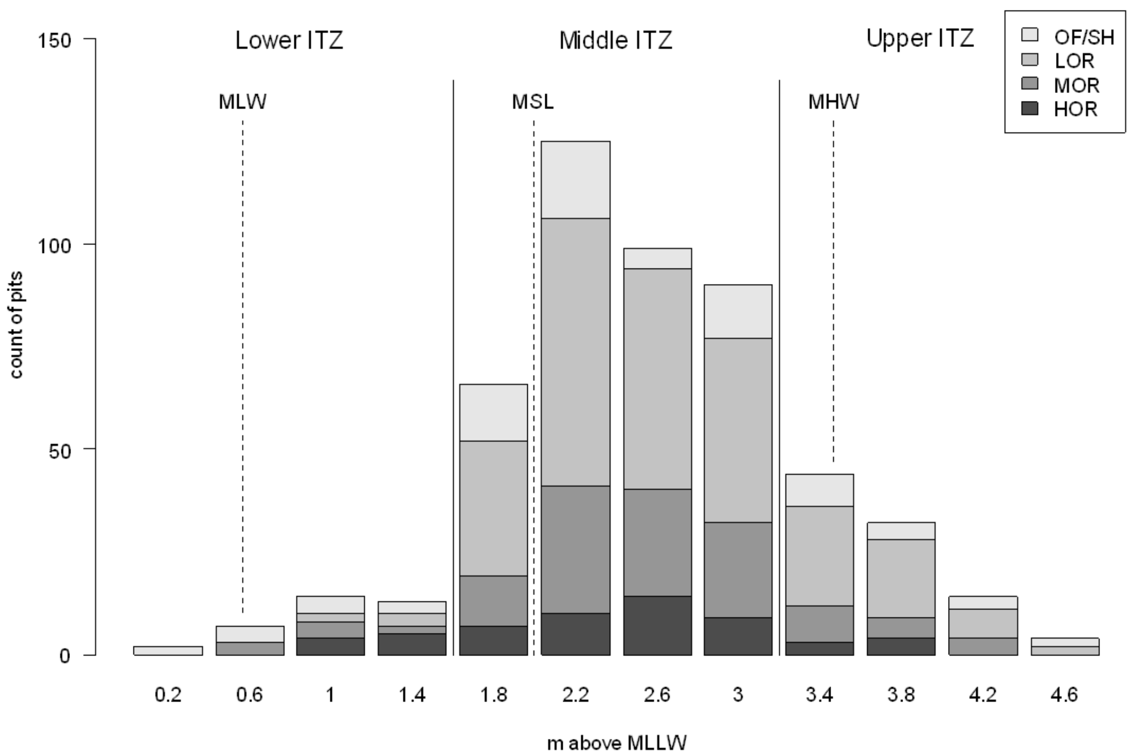


Figure 3. Counts of oiled pits by oiling descriptor and tidal elevation in meters above mean lower low water (MLLW) for pits in Prince William Sound surveyed in 2001, 2003, 2007, and 2008 (total of 509). Tidal range (0-4.8 m) is divided into thirds for the upper, middle, and lower intertidal zone (ITZ). Elevations of mean low water (MLW), mean sea level (MSL), and mean high water (MHW) are shown for reference. See Fig. 2 for definition of oiling categories.

Figure 4. Aerial photograph of Smith Island showing the general locations of upper, middle, and lower intertidal zones (ITZ).

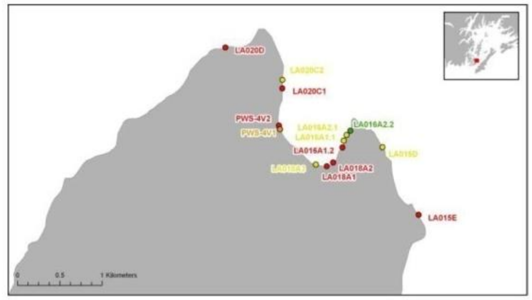
The upper intertidal zone is where waves tend to build and erode berms (this process usually removes the oil during the erosion part of the cycle), but it is also where the sediments are most permeable so the oil penetrated deeper, sometimes below the depths of normal reworking by waves as they break on the shoreline since the spill. The amount of oil in the lower intertidal is small but significant, because sea otters (Bodkin et al. 2012) and harlequin ducks (Esler et al. 2010) feed on the bottom in these areas.

Figure 5 shows a summary of the field survey data from a segment in Sleepy Bay on Latouche Island, which had three oiled pits out of 96 pits, with one MOR and two LOR. Figure 6 shows the field data summary for a segment on Eleanor Island with HOR and MOR in pits in a small tombolo behind a rock outcrop. A tombolo is the accumulation of sediments in the lee of a small islet connecting the islet to the mainland, and is one of the geomorphic features where the oil tends to persist.

LA020C1
Latouche Island
Prince William Sound
Subsurface/Surface Oil

1989	Heavy	Date:	7/8/01
1990:	Heavy	Length (m):	100
1991:	Moderate	ESI:	6A/2A

Pits: **96** Imp: **13** S Oil: **15** SS Oil **3**



green: no oil
yellow: surface oil
orange: subsurface oil
red: surface/subsurface oil

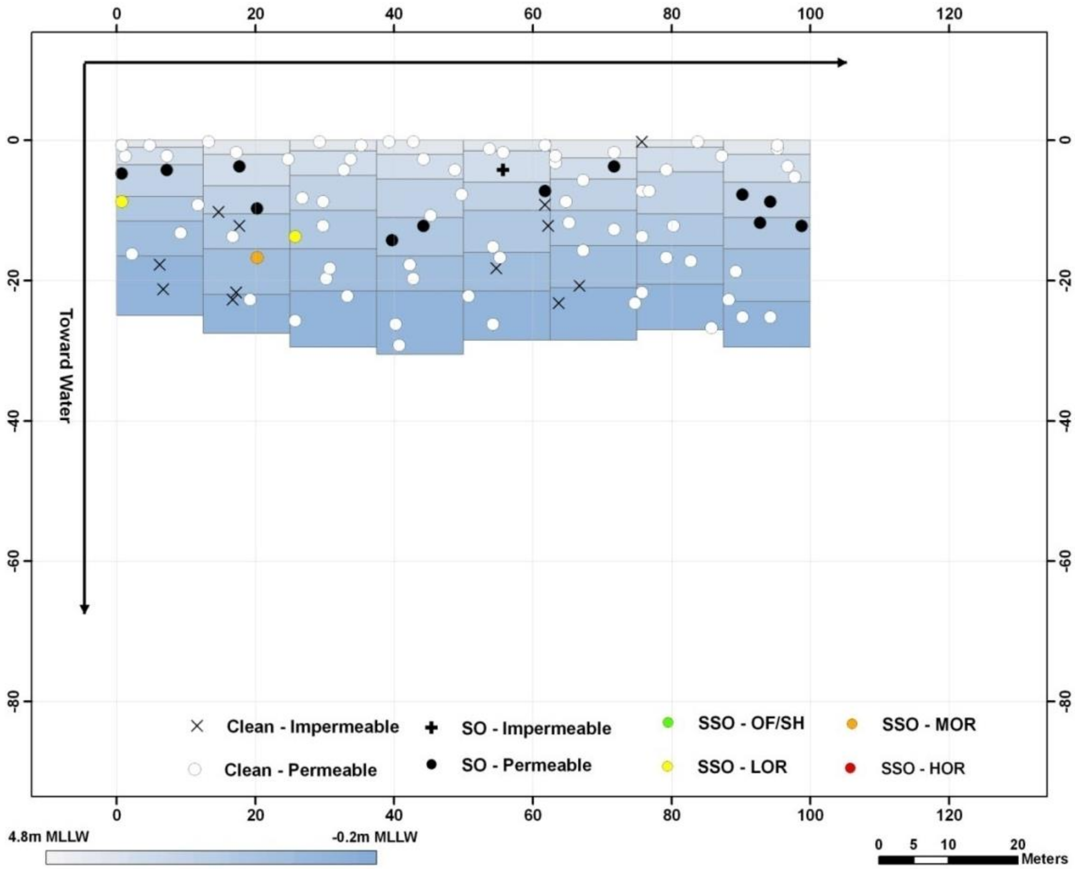
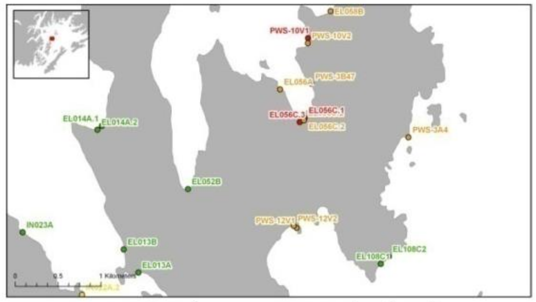


Figure 5. Site summary sheet for segment LA020C1 on Latouche Island, surveyed in 2001. The subsurface oil occurs in the area where the sediment accumulation is wider and thicker, bordered by shallow bedrock outcrops on either side.

PWS-3A4
Eleanor Island
Prince William Sound
Subsurface Oil

1989	Moderate	Date:	5/21/07
1990:	Heavy	Length (m):	87.5
1991:	Moderate	ESI:	6A/2A

Pits: **36** Imp: **10** S Oil: **0** SS Oil **11**



green: no oil   orange: subsurface oil
yellow: surface oil   red: surface/subsurface oil

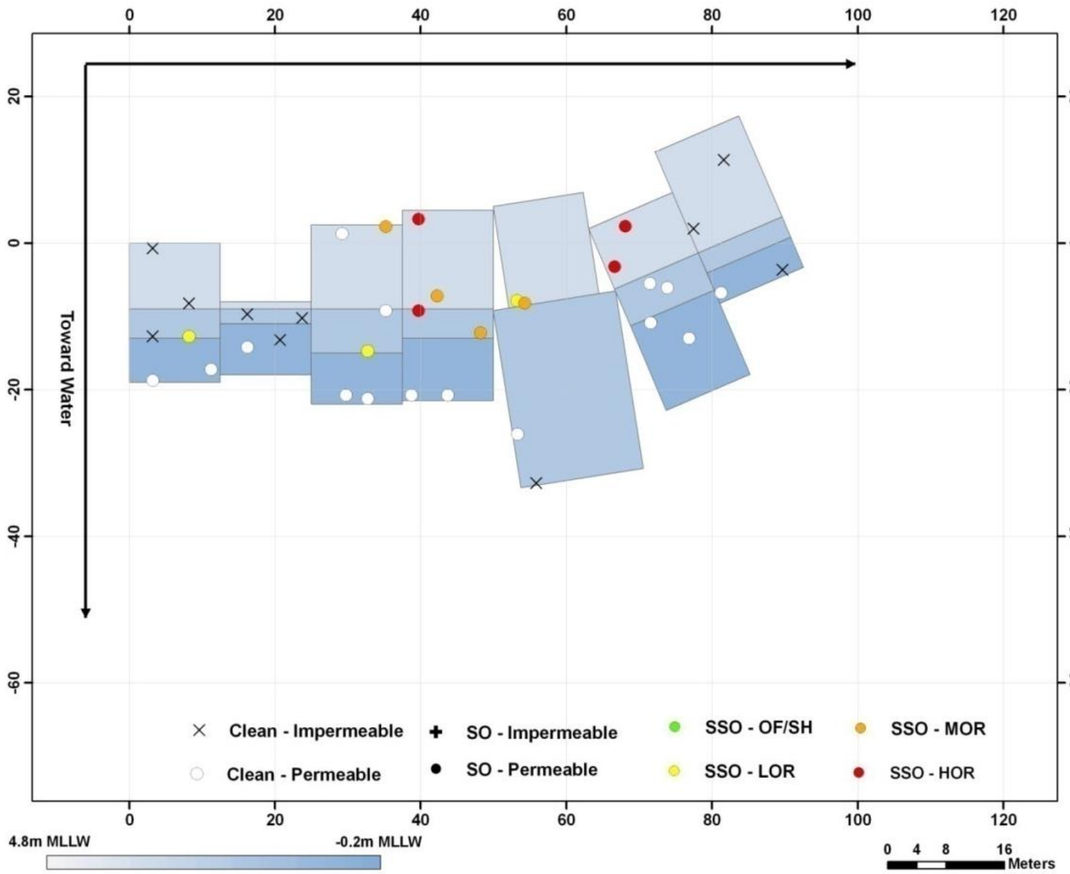


Figure 6. Summary sheet for segment PWS-3A4 on the northeast side of Eleanor Island surveyed in 2007. All of the pits in the small tombolo contained subsurface oil, which was mostly heavily to moderately oiled residues.

Why is the oil still there?

After review of all the beach segments studied in 2001-2008, it was clear that the presence and absence of lingering subsurface oil were being influenced by geomorphic and hydrologic factors such as those listed in Table 1. The initial degree of oiling is a very important influence on the presence of lingering oil, because the heavier the oiling, the deeper the oil would penetrate into the subsurface, which occurred only where the shoreline was permeable. Low exposure to wave action slowed the natural reworking of the oiled sediments. The importance of "armoring," which is where the finer gravels on the beach surface are transported away by waves, leaving a layer of coarse gravel on the surface that is very stable, was a new discovery following the spill (Hayes et al. 2010). Armored gravel beaches occur in Prince William Sound because of the wide range in the sizes of the gravel and in the variations in wave energy. These results both confirm and refine the findings of earlier investigations by Taylor and Reimer (2008) and others (Michel and Hayes 1991, Hayes and Michel 1999).

In addition, the ruggedness of the shoreline creates intertidal and nearshore bedrock outcrops that act as natural breakwaters, creating micro-sheltered habitats and bending the waves in ways that pile up sediment behind them. Accumulations of boulder-sized rubble along sheltered bays provide semi-permeable sediments with very limited wave energy. Li and Boufadel (2010) found that, in beaches with low freshwater seepage from the land, there were two layers in the beach: an upper layer that was permeable and a lower layer that was 100 times less permeable. The dissolved oxygen in the lower layer was low, which slowed microbial degradation of oil that penetrated into this lower layer.

Table 1. Geomorphic and hydrologic factors that tend to increase or decrease the likelihood of subsurface oil in Prince William Sound (modified from Nixon et al 2013)

Factors that increase the likelihood of subsurface oil	Factors that decrease the likelihood of subsurface oil
Heavy or moderate initial oiling	Impermeable bedrock
Low exposure to wave action	Platforms with a thin sediment veneer
Low topographic slope	Fine-grained, well-sorted gravel beaches with no armor
Armoring of gravel beaches	Low-permeability, raised bay-bottom beaches
Tombolos or natural breakwaters	Proximity to a stream outlet or strong shallow groundwater flow
Rubble accumulations	
Transitional edge effects (transitions between permeable and impermeable shoreline types)	

The challenge was to develop a series of models using parameters that would be surrogates for these factors and that could be mapped using available datasets. As described in Michel et al. (2010) and Nixon et al. (2013), the models they developed included a wide range of geomorphic and hydrographic variables as inputs, including:

- Distance from the spill source
- SCAT oiling history for 1989, 1990, and 1991
- Substrate permeability and distance to a transition between permeable and impermeable shoreline types (based on shoreline type)
- Shoreline convexity at different scales along shore (which influences the energy of waves that break on a shoreline)
- Intertidal topographic complexity (estimates the "bumpiness" of the intertidal zone, such as the presence of rock outcrops that create micro-sheltered areas)
- Intertidal slope

- Exposure index and maximum fetch (estimates the degree of exposure to wave energy)

Figure 7 shows model results for the northern part of Eleanor Island using different criteria. The top left figure shows the likelihood of any subsurface oil along the shoreline. Northwest Bay was very heavily oiled by the spill. There is lingering oil in the gravel beaches along the more sheltered bays, and along eastern shorelines in pocket beaches that are sheltered from wave action by bedrock outcrops and tombolos, which act as natural breakwaters.

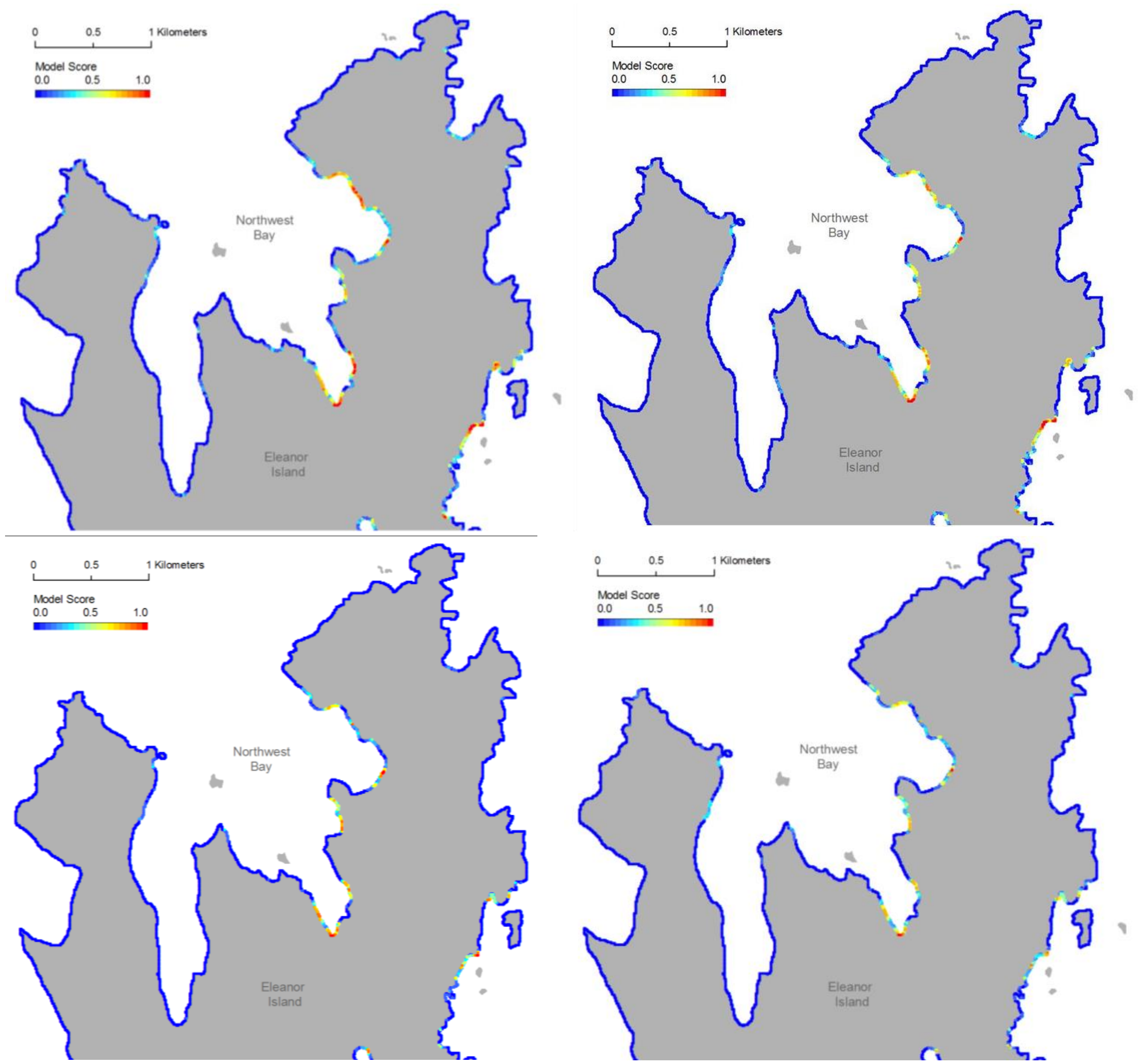


Figure 7. Maps of the model results for Eleanor Island for different criteria. Top Left: Any subsurface oil. Top Right: Lightly oiled residue (LOR) or higher. Bottom Left: Moderately oiled residue (MOR) or higher. Bottom Right = MOR or higher *plus* MOR or greater in 15% of the pits in the intertidal zone.

Note that the length of oiled shoreline decreases and gets patchier for the model output when mapping the higher oiling categories. The top right map shows shorelines where the model predicted likelihood of lightly oiled residues or greater (that is all areas with lightly, moderately, and heavily oiled residues) in the subsurface sediments. Note that some shorelines have changed color, going from the higher likelihood of reds and oranges, to lower likelihoods of yellow and green. Some shorelines change to blue, indicating that the model indicated no subsurface oil along those shorelines.

The bottom left map shows shorelines where the model predicted likelihood of moderately oiled residues (MOR) or greater in the subsurface sediments. There are fewer and shorter segments of shoreline for this model output. As discussed later, segments with sediments with MOR or greater was the first screening criteria for selecting candidates for remediation. The bottom right map shows areas with the likelihood of MOR or greater, *plus* MOR or greater in 15% or more of the pits in each section of shoreline. The model was run for MOR or greater and different % cover thresholds to identify areas that potentially have MOR that is less patchy, thus more likely to be candidates for remediation.

The model output in terms of length of shoreline and number of sites for any subsurface oil is shown in Table 2, for MOR or greater at different percent cover frequencies in Table 3, and any subsurface oil in three regions–Prince William Sound, Outer Kenai Peninsula, and Shelikof Strait in Table 4. Most of the subsurface oil as of 2001-2007 was located in Prince William Sound. As shown in Table 4, the length of shoreline with MOR or greater gets shorter as the amount of MOR per segment increases. That is, at 80% PPV, there are 6.42 km of shoreline with any MOR or greater, 5.09 km of shoreline with MOR or greater and 15% cover, 3.58 km of shoreline with MOR or greater and 30% cover, and 1.44 km of shoreline with MOR or greater and 50% cover.

Table 2. Known and model-estimated shoreline lengths and discrete number of sites by model score cutoff for any subsurface oil (SSO) and the three oiling descriptors LOR, MOR, and HOR. Site is defined as cluster(s) of shoreline locations within 100 meters of each other. PPV = Positive Predictive Value, an indicator of the expected accuracy of the model.

Cutoff	Any SSO		LOR or >		MOR or >		HOR or >	
	Length (km)	# Sites	Length (km)	# Sites	Length (km)	# Sites	Length (km)	# Sites
Known	2.85	67	2.56	57	1.45	37	0.48	17
90% PPV	19.36	167	11.59	113	3.57	64	2.21	47
80% PPV	35.04	276	23.04	194	6.42	105	2.94	49
70% PPV	49.57	321	32.21	235	11.32	147	5.86	88

Table 3. Known and model-estimated shoreline lengths and number of discrete sites by model score cutoff for any MOR, and three different frequencies of MOR occurrence. Site defined as cluster(s) of shoreline locations within 100 meters of each other. PPV = Positive Predictive Value, an indicator of the expected accuracy of the model.

Cutoff	MOR or >		MOR or > and > 15% Cover		MOR or > and > 30% Cover		MOR or > and > 50% Cover	
	Length (km)	# Sites	Length (km)	# Sites	Length (km)	# Sites	Length (km)	# Sites
Known	1.45	37	1.16	32	0.71	19	0.25	6
90% PPV	3.57	64	2.62	52	0.89	22	0.83	31
80% PPV	6.42	105	5.09	75	3.58	55	1.44	42
70% PPV	11.32	147	7.43	98	8.74	126	1.60	53

Table 4. Known and model-estimated shoreline lengths and discrete sites by region and model score cutoff for any subsurface oil for the entire study area, Prince William Sound (PWS), Outer Kenai Peninsula (KEN), and Shelikof Strait (SHL). Site defined as cluster(s) of shoreline locations within 100 m of each other. PPV = Positive Predictive Value, an indicator of the expected accuracy of the model.

Cutoff	Any SSO - Total		Any SSO - PWS		Any SSO - KEN		Any SSO - SHL	
	Length (km)	# Sites	Length (km)	# Sites	Length (km)	# Sites	Length (km)	# Sites
Known	2.85	67	2.53	60	0.02	2	0.30	5
90% PPV	19.36	167	16.13	138	1.42	14	1.81	15
80% PPV	35.04	276	29.31	227	2.54	24	3.19	25
70% PPV	49.57	321	41.07	261	3.61	30	4.90	30

To summarize, spatially explicit machine learning models were developed to identify potential areas where subsurface oil is still present on the shorelines of Prince William Sound and the Gulf of Alaska affected by the *Exxon Valdez* oil spill. The models were based on data collected at 314 shoreline segments that were surveyed between 2001 and 2008. These data allowed identification of geomorphologic and hydrologic factors that have contributed to the persistence of subsurface oil within Prince William Sound and the Gulf of Alaska two decades after the spill. The model used surrogates for these identified factors to make predictions about where oil subsurface oil is likely to be present at unsurveyed locations. The model results suggest there are a limited but significant number of as-yet uninvestigated locations in the study area that are likely to contain subsurface oil. Furthermore, the model results can also be used to prioritize shorelines for investigation with known uncertainty.

How much oil lingers?

Michel et al. (2010) estimated the total area and mass of oil by oiling categories for the same area that Short et al. (2004) used in their calculations. Table 5 shows that there is very good agreement between the two approaches. The *Exxon Valdez* spill released 10,800,000 gallons of

North Slope crude oil, which converts to 39,560 metric tons. Thus, the Michel et al. (2010) estimate of any subsurface oil in Prince William Sound as of the period of 2001-2007 represents 0.25% of the total spill, the estimate for LOR or greater represents 0.09% of the total spill, the estimate for MOR or greater represents 0.08% of the total spill, and the estimate for HOR or greater represents 0.07% of the total volume.

Table 5. Comparison of the estimates of the area (in hectares) and mass (in metric tons) of subsurface oil in Prince William Sound by any subsurface oil, LOR or greater, MOR or greater, and HOR or greater from Short et al. (2004) and the models of Michel et al. (2010).

Prince William Sound	Any SSO		LOR or >		MOR or >		HOR or >	
	Area (ha)	Mass (t)	Area (ha)	Mass (t)	Area (ha)	Mass (t)	Area (ha)	Mass (t)
Short et al. 2004	10.51	82.6	5.19	[illegible]	[illegible]	27.0	1.17	24.5
Michel et al. 2010	12.36	97.2	6.11	36.6	2.89	31.74	2.89	28.78

Carls et al. (2016) re-sampled six segments in Prince William Sound in 2015, using the same methods as Short et al. (2004) in 2001. They dug 400 pits in these six segments and found that there has been essentially no change in the percent of pits that were oiled, and thus no significant changes in the estimated total area with subsurface oiling at each site. The authors also reported no significant differences in average subsurface oil mass per unit area (kg/m^2) by subsurface oil descriptor (LOR, MOR, or HOR) from those reported by Short et al. (2004). Though this investigation was more limited in spatial extent and scope, these results imply that there is little ongoing change in the amount of subsurface oil in Prince William Sound at present in terms of either mass or areal extent.

Outside Prince William Sound, the lingering oil is different in terms of its state and where it has persisted. Irvine et al. (2014) have been studying the emulsified oil (mousse) under very large and stable boulder armor on very exposed beaches at five sites along Shelikof Strait and one low-energy site in Kenai Fjords for 23 years. As of 2012, they found that the emulsified oil at four sites has persisted in a slightly weathered state–it has essentially the same composition as an 11-day oil sample collected in Prince William Sound. Only the oil at McArthur Pass on the Kenai Peninsula showed evidence of biodegradation. The lack of weathering for 23 years is likely due to the formation of the stable emulsion (which makes the oil much less available to microbes).

Irvine et al. (2014) also found that the *surface* oil extent continued to decline and by 2012 was at very low levels at all sites, whereas *subsurface* oil continued relatively unchanged at four of the six sites. At Kashvik Bay, the site most distant from the spill, no oil was detected in 2012, and very little oil remained at either the surface or in the subsurface at the second-most distant site (Cape Gull). They used new chemical techniques to fingerprint the oil collected in 2012 and found that, at two sites, the oil could not be matched to the oil from the *Exxon Valdez*, indicating contamination from another source of oil. In addition, in 2011 and 2012, to determine if petroleum hydrocarbons were being released from the sites into the water column, Irvine et al. (2014) put out polyethylene membrane devices (PEMDs; material that sorbs oil from the water column) right above the oil patches and sampled mussels from the lower intertidal zone near the oil patches at two sites. No oil from the spill was detected in the mussels; however, oil was still detectable in about half of the PEMDs, indicating very low levels of dissolved oil were still being released from the surface oil at these two sites.

How weathered is the lingering oil?

The lingering oil varies widely in terms of its degree of weathering. In general, surface oil is more weathered than the subsurface oil (Short et al. 2004).

The degree of weathering of the subsurface oil over the last decade (2005-2014) varies from extremely weathered to no different than the 11-day floating *Exxon Valdez* oil, which has been the standard oil to compare the rate of weathering once the oil stranded on the shoreline (Short et al. 2007, Venosa et al. 2010). The weathering rate of the PAHs in oil buried in Prince William Sound has been greater than that in the more heavily emulsified oil in Gulf of Alaska beaches (Carls et al. 2016a). Even the biomarkers, those compounds in the oil that are most resistant to weathering, have been weathering, and the rate is also generally slower in the Gulf of Alaska (Carls and Fugate 2016, Carls et al. 2016b).

Is the oil bioavailable?

From 1992 to 1995, oil from the spill was detectable in mussels from oiled beaches in Prince William Sound (Carls et al. 2001). However, by 2002, Boehm et al. (2004) and Page et al. (2005) showed that oil concentrations in mussels from oiled sites in Prince William Sound were at or near reference-site levels. As part of the Long-term Environmental Monitoring Program (LTEMP) in Prince William Sound, Payne et al. (2008) showed steady decreases in the total polycyclic aromatic hydrocarbons (PAHs) from the *Exxon Valdez* oil in mussels from 1993 to 2006 from four sites in Prince William Sound, at which time only the mussels from Disk Island showed possible evidence of *Exxon Valdez* oil. Payne et al. (2015) reported that LTEMP mussel samples did not contain any lingering *Exxon Valdez* oil for samples collected in 2009 and 2013. Payne et al. (2013) concluded that the total PAHs in the buried oil were highly sequestered and did not appear to be bioavailable unless disturbed, such as by digging into the oiled layer. In Prince William Sound, the remaining oil is held in the beach sediments by capillary forces (Li and Boufadel 2010), so it can’t leach out by tidal flushing. Along Shelikof Strait, small amounts of the surface oil protected by very large boulders do leach out at very low levels, as demonstrated by the uptake of oil on the PEMDs deployed there in both 2011 and 2012 (Irvine et al. 2014).

What is the long-term fate of the lingering oil?

There are only two natural pathways for further removal of the subsurface lingering oil. The first one is by *physical disturbance*, such as during a storm that generates large waves that erode the beach sediments down to the oiled layer in Prince William Sound or dislodges boulders on the beaches along Shelikof Strait. However, the sediments have been subject to weathering and storms for over twenty-six years (as of January 2016) so it is not possible to predict when this will happen. The shorelines in Prince William Sound have wide ranges of orientation and lots of deep, sheltered bays. No one storm would be able to erode all the oil from the shoreline there; it would take multiple storms with different wave angles to accomplish that. This is confirmed by the lack of substantial change in the amounts of lingering subsurface oil between 2001 and 2015 at sites in Prince William Sound. Along Shelikof Strait, the surface oil has been almost completely removed by the small movements of the boulders over time, though the subsurface oil has not. If the oiled sediments were eroded during a storm, the turbulence is expected to break the oil into small droplets that would be dispersed into the water column and quickly diluted. Also, the oil is mostly in small patches, so there would only be a small amount of oil released per length of shoreline in most areas.

Physical disturbance could also occur by animals that excavate the sediments during foraging, such as sea otters, or humans who dig for shellfish or for other reasons. As discussed in the next section, there is no longer any evidence of oil exposure to sea otters, suggesting that they are not encountering oil when foraging. Also, the magnitude of human excavation is small relative to that of foraging otters, so humans are unlikely to encounter remaining oil unless they are specifically searching for it.

The second removal process is by *microbial degradation*. Venosa et al. (2010) showed that the PAHs in the oil could be further degraded in the presence of oxygen and nutrients. However, Li and Boufadel (2010) and Sharifi et al. (2010) showed that the dissolved oxygen levels in the beach groundwater in the oiled areas was very low, and the degradation rate in the absence of dissolved oxygen is very slow, orders of magnitude slower than that in the presence of dissolved oxygen. Therefore, it could be decades more before the oil is completely removed by natural processes.

EFFECTS OF LINGERING OIL ON MARINE LIFE AND SUBSISTENCE HARVEST

Recognizing that lightly weathered oil persisting in subsurface sediments of some intertidal areas could negatively affect marine life, the EVOSTC supported a large body of research evaluating (1) the degree of exposure to oil across a range of fish and wildlife species and (2) effects of oil on individuals and populations.

Following the *Exxon Valdez* oil spill, numerous studies indicated that marine invertebrates, fish, and wildlife continued to be exposed to oil well beyond the first weeks and months after the spill. For lingering oil to have effects on individuals and populations, animals must be exposed to that oil at levels that have meaningful biological consequences. This requires evaluation of the timeline and degree of exposure, as well as the timeline and degree of effects; however, these should not be expected to be entirely concordant. There may be consequences of chronic exposure to lingering oil that persist even after exposure is eliminated or reduced below toxic effects. For example, individuals exposed to oil in the past may be more likely to suffer deleterious health effects beyond the period of exposure. Similarly, depressed numbers of animals due to population-level consequences of exposure will not immediately rebound following cessation of exposure; demographic lags may prolong full population recovery (Matkin et al. 2008). Conversely, evidence of exposure cannot be assumed to indicate deleterious effects, as exposure may occur, and be detected, below levels that cause meaningful damage to individuals or populations (Lee and Anderson 2005). Summaries of exposure and effects are found below.

In this section, we address the following questions:

- What species were exposed to lingering oil and how long did exposure occur?
- What were effects of exposure to oil and how long did they persist?

What species were exposed to lingering oil and how long did exposure occur?

<u>Exposure of Marine Invertebrates to Lingering Oil</u>

Invertebrates (mostly shellfish) in intertidal and shallow subtidal areas of Prince William Sound where oil stranded were clearly exposed to high concentrations of PAHs at the time of the spill (Houghton et al. 1996, Jewett et al. 1999). As filter-feeders that can accumulate PAHs, and as key prey for vertebrate predators that showed delayed recovery (including sea otters and harlequin ducks; see below), evaluations of persistence of lingering oil focused on clams and mussels.

Hydrocarbon burdens in littleneck clams were elevated in oiled areas in 1991, and were different from reference samples through 1997 (Shigenaka et al. 2008). By 2002, littleneck clams did not show elevated PAHs (Thomas et al. 2007). Similarly, tissue hydrocarbon levels in littleneck clams were very low in 2007, across all oiling history categories, indicating that exposure to lingering oil had ceased by that time (Shigenaka et al. 2008).

PAHs were elevated in mussels in some areas in the first years following the *Exxon Valdez* oil spill (Boehm et al. 1996, Carls et al. 2001). By 2002, Thomas et al. (2007) found low levels of *Exxon Valdez* oil in mussels from known oiled sites. Boehm et al. (2004) and Page et al. (2005) also concluded that hydrocarbons in mussels were at or near background by 2002.

Exposure of Fish and Wildlife to Lingering Oil

In the decades following the *Exxon Valdez* oil spill, numerous studies indicated that fish and wildlife continued to be exposed to oil well beyond the acute phase (first weeks and months) following the spill (Bowyer et al. 2003, Golet et al. 2002, Jewett et al. 2002; Trust et al. 2000). Differences in indicators of oil exposure between areas of Prince William Sound that received *Exxon Valdez* oil and those that were not oiled were largest and most persistent for animals that occurred in intertidal habitats, particularly those that consumed benthic invertebrates that live on or in the sediment, such as harlequin ducks (Esler et al. 2010; see below). Recent work has shown that species previously exhibiting protracted exposure to lingering oil are no longer being exposed (Esler et al. 2015), suggesting that lingering *Exxon Valdez* oil is no longer bioavailable to vertebrate consumers.

Many studies of oil exposure in vertebrates after the *Exxon Valdez* oil spill were based on indicators of cytochrome P4501A (CYP1A) induction, which is elevated when animals are exposed to one of a limited number of compounds, including polycyclic aromatic hydrocarbons (PAHs) like those found in crude oil. CYP1A induction in response to PAHs has been demonstrated in large numbers of lab and field studies (Goksøyr 1995, Peakall et al. 1989, Ben-David et al. 2001, Miles et al. 2007), and is widely considered to be a reliable indicator of oil exposure. Elevated CYP1A induction reflects recent oil exposure (over a scale of weeks), not historical exposure (i.e., exposure in previous years).

Because other compounds can induce CYP1A, particularly polychlorinated biphenyls (PCB), studies (Ricca et al. 2010) evaluated whether indications of elevated induction of CYP1A in oiled areas that were interpreted as exposure to *Exxon Valdez* oil could have been confounded by differential PCB exposure between oiled and unoiled areas. Ricca et al. (2010) found that concentrations of PCBs generally were low in harlequin ducks and sea otters from Prince William Sound. Further, the patterns of PCB composition in sea otters and harlequin ducks did not suggest recent exposure to persistent PCBs but rather exposure to distantly derived, weathered PCBs. Finally, PCB concentrations were similar between oiled and unoiled areas of Prince William Sound, or higher in unoiled areas for some groupings of PCBs in sea otters. Therefore, PCBs likely did not explain observed patterns of elevated CYP1A induction in areas oiled during the *Exxon Valdez* spill in a number of vertebrate species. Combined with considerations of sources of hydrocarbons on beaches (Short et al. 2004), it is unlikely that there are any compounds confounding interpretation of CYP1A induction as an indicator of exposure to lingering *Exxon Valdez* oil.

For pink salmon, natural history and vulnerability to spilled oil are linked. Pink salmon return in the fall to their natal streams and spawn in gravel, particularly in the intertidal portions of stream mouths. Embryos develop in the gravel until fry emerge the following spring. Unfortunately, pink salmon spawning areas place developing embryos in the habitat where oil was stranded and subsequently persisted. In addition to the potential for oil-exposed embryos, it was possible that out-migrating fry would receive exposure as they dispersed and foraged along contaminated shorelines.

At the time of the spill, an estimated 31% of the pink salmon spawning streams in the southwest portion of Prince William Sound were oiled to some extent. Evidence of oil exposure included PAH concentrations in tissue, induction of the biomarker enzyme CYP1A, (Carls et al. 1996, Willette 1996, Wiedmer et al. 1996), and oil globules observed in stomachs and intestines (Sturdevant et al. 1996).

Two intertidal fish, crescent gunnels and masked greenlings, showed evidence of elevated biomarkers in oiled areas of Prince William Sound, relative to unoiled areas, in 1998 and 1999 (Jewett et al. 2002). More recent samples, collected in 2008, indicated elevated CYP1A induction in crescent gunnels from heavily oiled areas relative to those from unoiled areas, by a factor of 2.8 times. However, gunnels from moderately oiled areas, and greenlings from all oiled areas, showed no elevation of biomarkers, suggesting that they are no longer exposed to lingering *Exxon Valdez* oil.

Studies of harlequin ducks have generated the most complete data series evaluating CYP1A induction since the *Exxon Valdez* oil spill, measured as 7-ethoxyresorufin-O-deethylase (EROD) activity in liver samples taken as nonlethal biopsies from captured birds. In 1998, 9 years after the spill, harlequin ducks from oiled areas of Prince William Sound had indicators of CYP1A that averaged nearly 3 times higher than those from unoiled areas (Trust et al. 2000). Similar patterns were observed through 2009 (Esler et al. 2010), which was interpreted as evidence that some harlequin ducks continued to be exposed to lingering *Exxon Valdez* oil for up to 20 years post-spill.

Recent data indicate reduction of exposure to lingering *Exxon Valdez* oil. In 2011, indices of CYP1A activity of harlequin ducks from oiled areas were lower than in previous years, although still statistically different from those in unoiled areas (Figure). In 2013 and 2014, average EROD activity did not differ between harlequin ducks from oiled and unoiled areas of Prince William Sound. We interpret these data to indicate that exposure to lingering oil had largely ceased by 2013, 24 years after the EVOS.

Another intertidal-dwelling, invertebrate-consuming sea duck, Barrow’s goldeneye, also showed higher average hepatic EROD activity in oiled areas of Prince William Sound than in unoiled areas during 1996/97 (Trust et al. 2000) and 2005 (Esler et al. 2011). The most recent data, from 2009, indicated that average CYP1A induction was similar between oiled and unoiled areas (Esler et al. 2011), suggest that Barrow’s goldeneyes exposure to *Exxon Valdez* oil had ceased by 2009.

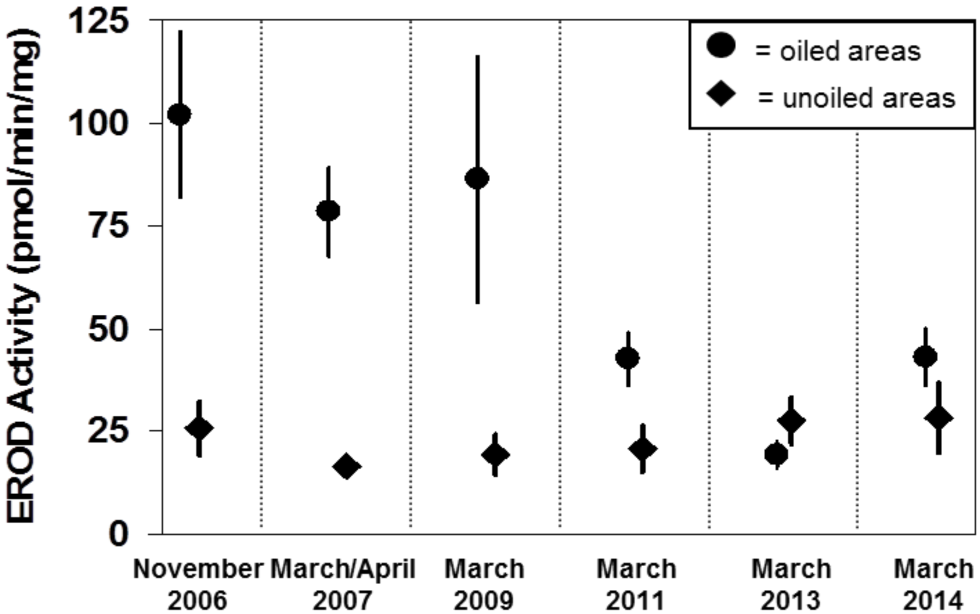


Figure 8. Average (± standard error) hepatic 7-ethoxyresorufin-O-deethylase (EROD) activity (pmol/min/mg protein) of harlequin ducks captured in Prince William Sound, Alaska in March 2013 and 2014 (n = 50 each year), contrasted with results from previous years (Esler et al. 2010).

CYP1A induction of another bird, the black oystercatcher, was measured in 2004 as hepatic EROD activity. Average induction was slightly (1.5 times) but statistically significantly higher in oystercatchers captured on oiled areas, relative to those from unoiled areas, suggesting continued exposure to oil at that time. Like sea ducks, oystercatchers occur in the intertidal zone and consume benthic invertebrates, which may increase their risk of exposure to lingering oil. Oystercatchers have not been sampled since 2004 to determine whether indicators of exposure have declined.

Pigeon guillemots, a sea bird that forages in shallow waters for both fish and invertebrates, showed elevated average CYP1A response in oiled areas in 1999 in adults, but not nestlings in 1998 (Golet et al. 2002). Data collected during 2004 from adult guillemots showed that average indicators of CYP1A induction were similar between oiled and unoiled areas, indicating that exposure to oil had ceased by that time.

Valid, noninvasive methods of determination of CYP1A induction in sea otters have not been developed, so other methods have been applied. A technique for measuring differential gene expression in response to hydrocarbon exposure was developed based on laboratory studies with mink (Bowen et al. 2007). The array of genes that was evaluated includes aryl hydrocarbon receptor (AhR), which is a direct but short-lived measure of hydrocarbon exposure, as well as other genes that vary in expression following hydrocarbon exposure. For sea otters captured in 2008 in oiled areas of Prince William Sound, there was no evidence of differential AhR expression relative to unoiled reference areas (Miles et al. 2012); however, AhR is ephemeral, with differential expression likely detectable for only days following exposure. When considered collectively in a cluster analysis, gene assays indicated that sea otters from oiled areas of Prince William Sound displayed patterns consistent with (but not exclusive to) the kinds of responses that would be expected as a result of the physiological cascade that occurs following exposure to hydrocarbons, particularly inflammation and immune suppression.

As another measure of potential for oil exposure in sea otters, Bodkin et al. (2012) estimated the average number of times a sea otter would encounter oil annually at heavily oiled northern Knight Island. Sea otters excavate sediments when they forage; therefore, they could be exposed either through consumption of prey that have assimilated hydrocarbons (Fukuyama et al. 2000) or by disturbing oiled sediments and releasing lingering oil, which could then adhere to their fur and subsequently be ingested upon grooming. Bodkin et al. (2012) evaluated the degree of spatial and temporal overlap of foraging otters and lingering oil, to determine whether these were plausible pathways of exposure.

Using sea otters with abdominally implanted time and depth recorders, Bodkin et al. (2012) found that, of more than a million foraging dives, most (82%) were subtidal, and thus not a risk for encountering lingering oil (Figure 9). However, all individuals (n = 19) foraged in intertidal zones at least some of the time, averaging between 8 and 91 intertidal foraging dives per day. Within the intertidal zone, foraging dives occurred most frequently at lower elevations, where lingering oil was less common (Figure 2), but an average of 3 to 38 dives per day were at elevations >1.8 m above MLLW, where most lingering oil persisted. Based on these foraging data and the distribution of lingering oil during 2001 and 2003, Bodkin et al. (2012) estimated that sea otters would encounter subsurface lingering oil an average of 10 times each year, ranging from 2 to 24 times, depending on individual foraging routines (Figure 10).

In sum, the body of evidence suggests that many intertidal or intertidally-foraging species were exposed to lingering *Exxon Valdez* oil for years to decades post-spill. Most recent data show that even the species exhibiting the longest timelines of exposure are no longer exhibiting evidence of exposure (e.g., harlequin ducks), indicating that lingering oil is no longer bioavailable in significant amounts. Although the timeline of exposure was much longer than anticipated, evidence suggests that direct ecological effects from exposure have ceased. These findings not only indicate the expected pattern of declining exposure over time, they also support the conclusion that elevated levels observed earlier were related to exposure to *Exxon Valdez* oil when it was more abundant, rather than some other contaminant.

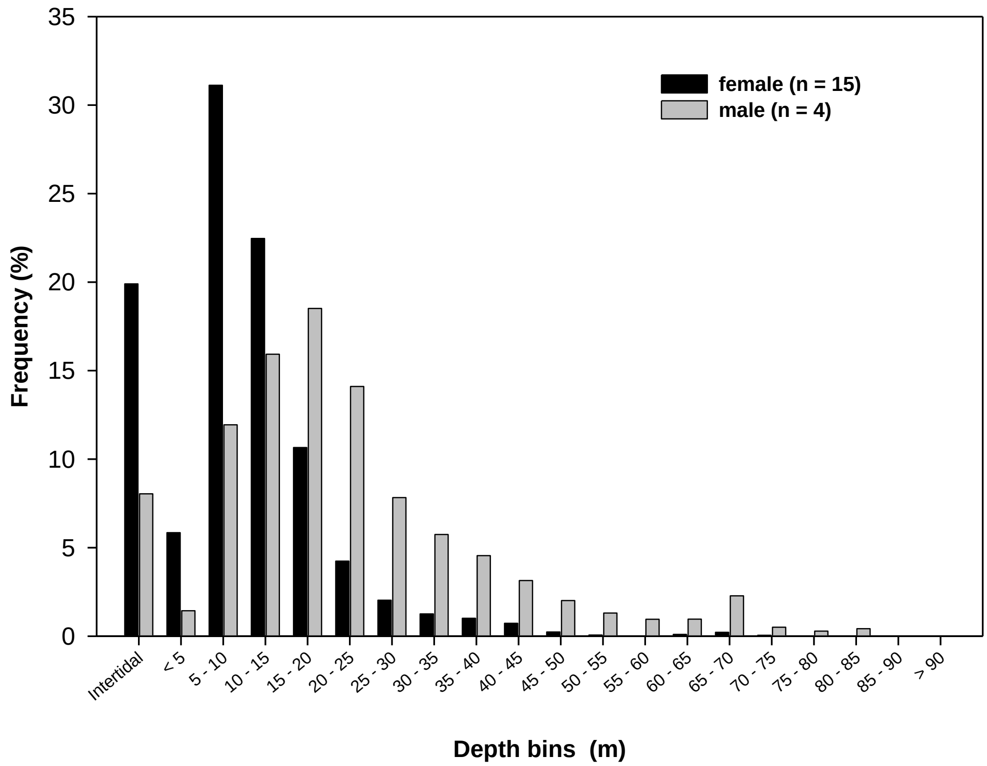


Figure 9. Depths of sea otter foraging dives in Prince William Sound, Alaska, based on animals outfitted with time and depth recorders. From Bodkin et al. (2012).

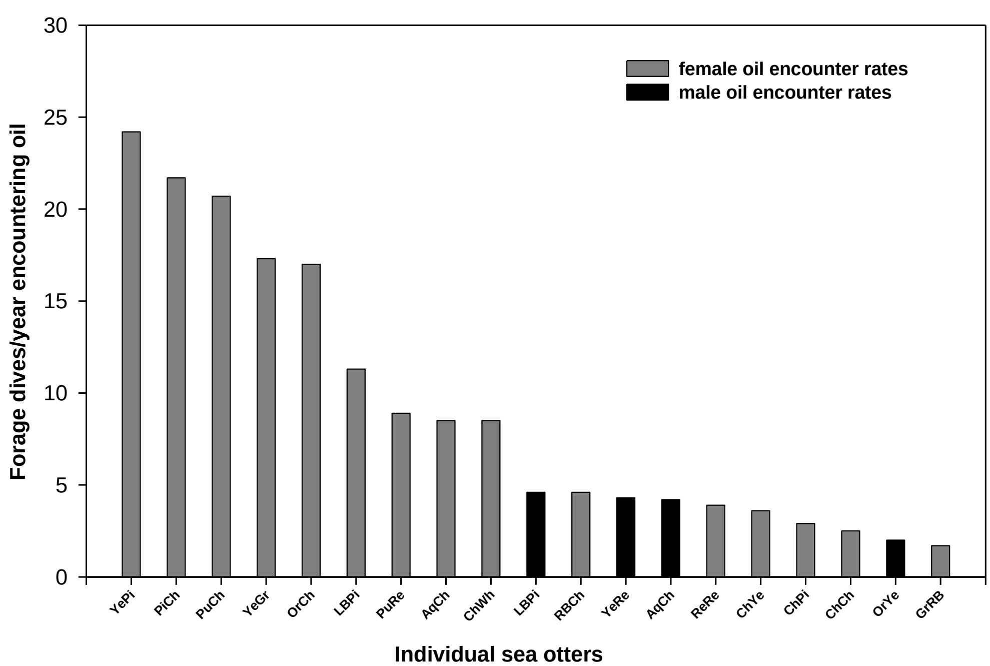


Figure 10. Estimated number of foraging dives per year in which individual otters (n = 19) would encounter lingering, subsurface oil in intertidal zones of Prince William Sound, Alaska. From Bodkin et al. (2012).

What were effects of exposure to oil and how long did they persist?

Effects of Lingering Oil on Marine Invertebrates

Marine invertebrates in oiled areas of Prince William Sound were directly affected by the *Exxon Valdez* oil, with evidence of extensive acute mortality across a range of taxa (Highsmith et al. 1996, Houghton et al. 1996, Jewett et al. 1999).

NOAA monitoring showed consistent recovery of intertidal invertebrates in oiled areas between 1990 and 2000, with most recovery occurring within a few years post-spill (Coats et al. 1999). Similarly, by 1995, most differences detected between oiled and reference subtidal areas had ceased (Jewett et al. 1999).

Fukuyama et al. (2014) reviewed recovery status of infaunal communities. Most species had recovered within a few years of the spill. As an exception, littleneck clam abundance was depressed in oil-affected areas for longer than most taxa, but steadily increased until comparable to reference sites by 2000. Lack of recovery of littleneck clams at oiled beaches that were treated likely reflected removal of interstitial sediments during clean-up (Lees and Driskell 2007, Shigenaka et al. 2008). However, resampling in 2007 revealed that littleneck clams had declined across PWS, irrespective of oiling history, indicating that lingering oil was not a causative factor (Shigenaka et al. 2008).

Survival and growth of littleneck clams was depressed several years after the spill, in association with elevated hydrocarbon burdens (Fukuyama et al. 2000). This was demonstrated by a translocation experiment, in which clams from unoiled areas were moved to oiled areas and accumulated hydrocarbons and associated deleterious effects. Hydrocarbon burdens were reduced in clams that were moved from oiled to unoiled sites, and this was associated with improved growth (Fukuyama et al. 2000).

Thomas et al. (1999a) found that mussels from oiled beaches in 1996, with confirmed elevated hydrocarbon concentrations, had lower survival that reference mussels. However, Thomas et al. (1999b) did not find differences in byssal thread production, condition index, clearance rate, or glycogen content between mussels from oiled sites and reference sites.

Thomas et al (2007) found low levels of DNA damage in mussels and clams in known oiled areas in 2002, 13 years after the *Exxon Valdez* oil spill. There is no evidence that injuries to marine invertebrates caused by lingering oil have persisted beyond that time.

Effects of Lingering Oil on Fish

Among fish, studies of effects of lingering oil were most detailed for pink salmon. Acute effects on growth and survival of pink salmon fry were detected in 1989 (Wertheimer and Celewycz 1996, Willette 1996). By 1990, fry grew comparably in oiled and unoiled reference portions of Prince William Sound with no evidence of increased CYP1A enzyme induction or tissue hydrocarbons, suggesting that lingering oil was not having an effect.

However, sampling through 1993 continued to find elevated embryo mortalities, although the differences in mortalities with reference streams declined over time (Bue et al. 1998). Murphy et al. (1999) confirmed that lingering oil adjacent to streams was associated with elevated embryo mortalities. Further, embryo mortality was consistent with interstitial drainage of oil-contaminated water into spawning gravels from surrounding stream banks (Carls et al. 2003).

A series of controlled laboratory exposure tests were conducted to determine if chronic low-level exposure to polycyclic aromatic hydrocarbons could duplicate the field observation of elevated pink salmon embryo mortalities. Long-term exposure of embryos to PAH concentrations of 20-50 part per billion resulted in increased deformities, slower development, histopathological damage, and lower survival (Marty et al. 1997, Heintz et al., 1999, 2000). Chemical analyses for hydrocarbons in tissues and induction of CYP1A in embryonic tissues indicated that PAHs were permeating the outer egg membranes, resulting in lower growth and survival (Carls et al. 2003).

The sensitivity of embryos to chronic exposure to vary low PAH concentrations was demonstrated through delayed impacts on growth and marine survival (Heintz et al. 1999, 2000). In these experiments, pink salmon embryos were exposed to water contaminated with four different concentrations of PAH, and then moved to salt water pens for growth tests, or tagged and released to the environment to assess adult returns 1.3 years later. A delayed effect on growth was measured in juvenile salmon that survived embryonic exposure to a concentration of 18 ppb. Marine survival of salmon fry was tested using coded-wire tags to indicate the exposure dose during the embryonic life stage. Over a year later, the returning adults bearing the tags were decoded and counted to determine survival rates by dose (included controls treated and tagged similarly). Marine survival of pink salmon that had been exposed as embryos to 5 ppb in 1995 was reduced 16%. Exposure to 19 ppb resulted in a 36% reduction in marine survival, indicating a dose-response relationship. The controlled laboratory exposure tests, followed by further environmental challenges (migration out and back, growth, predation) are unprecedented and demonstrate that low level exposures (ppb) at the embryonic life stage can affect fitness (growth), and have an impact at the population level (adult returns).

Collectively, the measure of elevated embryo mortality in the five years following the spill, identification of the exposure mechanism from contaminated beaches to spawning gravels, and measured effects on fitness following embryonic exposures provides compelling evidence of chronic impacts on pink salmon from lingering oil. Given that lingering oil is sequestered and unavailable, these types of effects likely have not occurred in recent years.

Effects of Lingering Oil on Wildlife

Initial consequences of the *Exxon Valdez* oil spill for wildlife were immediate and obvious. Mortalities due to oil in the weeks following the spill were estimated to be in the hundreds of thousands of marine birds (Piatt et al. 1990), several thousand sea otters (Garrott et al. 1993, Ballachey et al. 1994), significant proportions of resident (33%) and transient (41%) pods of killer whales (Matkin et al. 2008), and varying numbers of a wide assortment of other wildlife species. These levels of mortality are consistent with expectations, given the amount of oil spilled, the size of the oil-affected area, the abundance of wildlife in the area, and the known toxic and thermoregulatory consequences of exposure to oil, particularly in cold-water environments.

Other effects of oil spills on wildlife, including chronic or indirect effects related to lingering oil, were not fully understood, recognized, or anticipated at the time of the *Exxon Valdez* oil spill (Peterson et al. 2003, Rice 2009). However, a considerable body of research funded by the EVOSTC has addressed wildlife recovery from the spill. This has allowed for an unprecedented and thorough understanding of the timelines and mechanisms of population recovery following catastrophic spills. Below, we review the timelines and processes of recovery of wildlife from the *Exxon Valdez* oil spill. We also consider factors that resulted in variation in recovery times across species, and present recent data for two species that showed protracted recovery related to exposure to lingering oil, the sea otter (*Enhydra lutris*) and harlequin duck (*Histrionicus histrionicus*).

Varying Mechanisms of Effect

Wildlife mortality has been documented in association with many large oil spills (e.g., Flint et al. 1999, Goldsworthy et al. 2000, Munilla et al. 2011). Much of this mortality occurred in the days and weeks following these events, when freshly spilled oil is readily encountered by wildlife, known as the acute phase. Effects expressed beyond the acute phase, known as chronic effects, can extend for months, years, or decades and, in the case of chronic effects due to exposure to lingering oil, may exceed the magnitude of acute mortalities (Iverson and Esler 2010, Monson et al. 2011).

Chronic effects of oil spills on wildlife can be manifested in a number of ways, including direct and delayed toxic effects, demographic lags, and indirect effects. Direct chronic effects occur due to toxic or thermoregulatory effects of exposure to lingering oil. Chronic effects also result from demographic lags, i.e., the time it takes for populations to return to conditions that would have existed in the absence of the spill, after direct oil effects have ceased. In other words, there are constraints on how fast populations can increase in abundance and these can delay recovery. The importance of demographic lags depends on a number of species-specific factors, including maximum reproductive potential, rates of dispersal, population structure, and other factors influencing density dependence. However, it is clear that immediate recovery did not occur once the direct survival effects of the EVOS ended for a number of taxa, and demographic lags were undoubtedly involved to some degree (Matkin et al. 2008, Iverson and Esler 2010, Monson et al. 2011). Finally, chronic effects can result from indirect effects that can take a number of forms. Generally, indirect effects refer to oil spill-induced changes to the environment that, in turn, have deleterious consequences for wildlife. These could result from changes in prey availability or predator abundance, for example, or from other cascading effects that occur upon disruption of complex food webs (Peterson et al. 2003). Indirect effects related to EVOS-induced changes to prey availability have been implicated as a contributing factor constraining recovery for some taxa (Golet et al. 2002) but rejected for others (Dean et al. 2002, Esler et al. 2002).

Review of Recovery Timelines Across Wildlife Species

In this section, we introduce a suite of species that exemplify variation in injury and recovery timelines (Figure 11) and highlight the role of lingering oil. Different wildlife species have different vulnerabilities to the suite of potential effects of oil spills described above. Those vulnerabilities are influenced by life history characteristics, such as generation times, reproductive potential, and natural survival rates, along with natural history characteristics, such as habitat use, diet, and foraging behavior. Given the diversity of wildlife species occurring in marine habitats of the northern Gulf of Alaska, it is not surprising that effects of the EVOS varied, including the relative importance of lingering oil.

Glaucous-winged gulls are representative of species for which relatively small numbers of acute mortalities were detected (Piatt et al. 1990). Densities may have been depressed during the year of the spill, but no chronic injury or lack of recovery was evident (Day et al. 1997, Irons et al. 2000, Cushing et al. 2012). Lingering oil did not play any role.

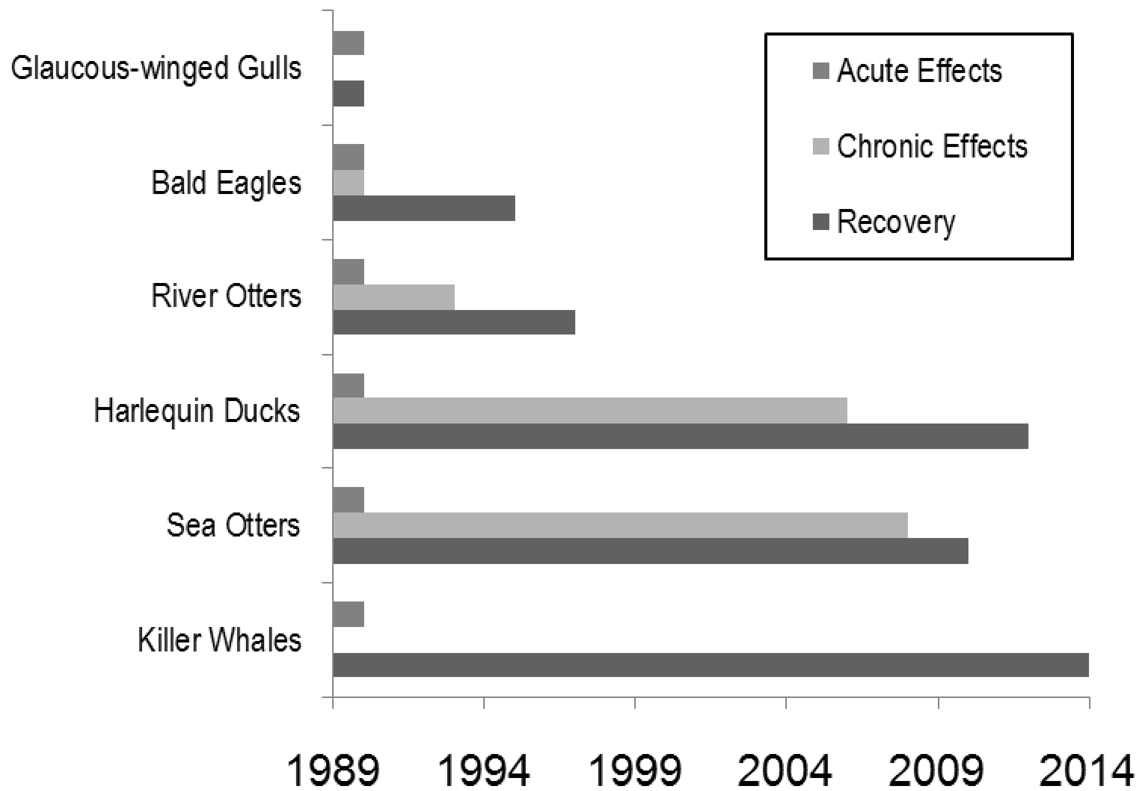


Figure 11. Examples of timelines of injury and recovery of selected species following the 1989 *Exxon Valdez* oil spill in Prince William Sound, Alaska.

Bald eagles experienced roughly 5% acute mortality in Prince William Sound and significantly reduced reproductive performance in oiled areas during 1989, the year of the spill (Bowman et al. 1997). However, no differences in survival or reproduction were observed in subsequent years (Bowman et al. 1995), and bald eagles were considered to have recovered by 1995 based on return to pre-spill numbers. There was no evidence that lingering oil was a constraint on eagle recovery.

River otters in oiled areas expressed values for a variety of biomarkers indicative of poor health during the years immediately following the EVOS (1989-1992), presumably as a result of direct chronic exposure to oil (Bowyer et al. 2003). Habitat use, diet, and body mass also differed between river otters living in oiled and unoiled areas during that period. By the mid-1990s, many of these attributes had improved, and Bowyer et al. (2003) concluded that river otters had recovered by 1997. Lingering oil likely played a role in constraining river otter recovery, but for less than a decade post-spill.

Harlequin ducks have been intensively studied since the spill. Several lines of evidence indicate direct chronic injury and protracted recovery related to exposure to lingering oil. Densities were lower in oiled areas than expected through 1997, after accounting for habitat attributes (Esler et al. 2000a). Also, adult female survival in oiled areas was depressed up to a decade following the spill (Esler et al. 2000b), coincident with biomarker evidence of exposure to lingering oil (Trust et al. 2000). By 2003, survival rates had returned to normal, despite continued oil exposure (Esler and Iverson 2010). Demographic data were assembled in a population model, and the best estimate of time until full recovery was 24 years post-spill, or 2013, due in part to demographic lags (Iverson and Esler 2010). Habitat utilization models for harlequin ducks using survey data from 2000 and 2008, environmental covariates, and more detailed modeled estimates of amounts of nearby lingering shoreline oiling (Nixon et al., in prep) indicated limited evidence that harlequin ducks were underutilizing suitable habitat in 2000, but that by 2008, little evidence of this effect remained. However, biomarker data indicated that harlequin ducks continued to be exposed to oil through 2011 (Esler et al. 2015). Lingering oil played an important role in the protracted recovery of harlequin ducks.

Sea otters are another species for which exposure to lingering oil led to direct chronic effects. Population models were developed to determine the timeline and spatial extent of mortalities related to chronic effects of the *Exxon Valdez* spill, using age distributions of living and dead otters and spatially-explicit population trend data (Monson et al. 2000, 2011). These models indicated that mortality rates were higher in areas affected by lingering oil until at least 2005, at which point survival effects began to dissipate.

Sea otter abundance data show the timeline of recovery from exposure to lingering oil. Since 1993, sea otter abundance has been quantified based on aerial survey methods (Bodkin and Udevitz 1999) throughout western Prince William Sound, including the northern Knight Island archipelago, where heavy oiling resulting in strong acute and chronic effects on otters and, thus, where recovery was most delayed. Pre-spill numbers at northern Knight Island were estimated as the number of living animals observed in, as well as the number of carcasses recovered from, the northern Knight Island survey area (Dean et al. 2000). Sea otter abundance in western Prince William Sound has been increasing since shortly after the oil spill (Ballachey et al. 2014), likely reflecting recovery from mortalities as a result of the oil spill; however, comparable estimates from before the spill are not available at this spatial scale.

Habitat utilization modeling for sea otters using survey data from 1999 through 2013, environmental covariates, and more detailed estimates of amounts of nearby ongoing lingering shoreline oiling (Nixon et al., in prep) indicated evidence that sea otters were underutilizing otherwise suitable habitat, and that these patterns of underuse were related to ongoing presence of lingering heavier subsurface shoreline oiling in the lower intertidal. These effects were found in all years, even as overall abundances increased though 2013, though the magnitude of these effects have been generally declining over time. However, it is likely that these effects are driven by lingering demographic effects and dispersal lag in only a subset of originally heavily oiled areas across northern Knight Island.

Across the wider northern Knight Island area, sea otter abundance was below estimated pre-spill abundance through 2009, 20 years after the *Exxon Valdez* spill (Figure). However, in the 3 most recent years of surveys (2011, 2012, and 2013), abundance was similar to the pre-spill estimate (Figure). Given that northern Knight Island likely represents a “worst-case scenario” in terms of sea otter recovery, these recent data are an encouraging sign that sea otter status in Prince William Sound met the recovery criteria set by the EVOSTC, and in 2014 sea otters were declared recovered (*Exxon Valdez* Oil Spill Trustee Council 2014).

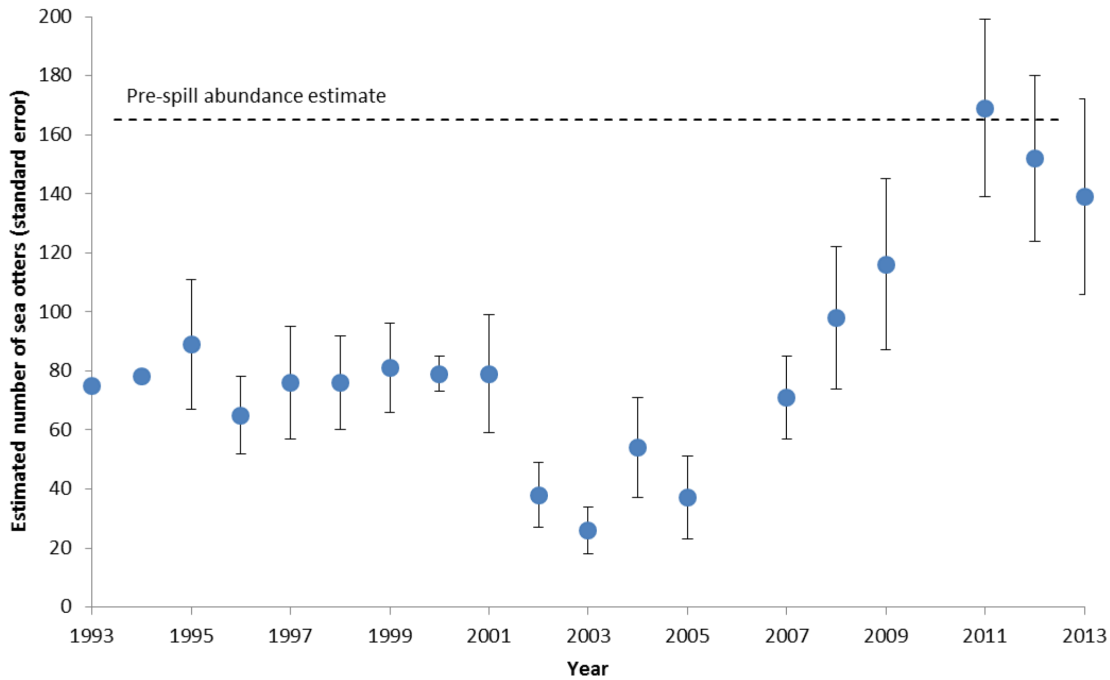


Figure 12. Estimated numbers of sea otters (± standard error) at heavily oiled northern Knight Island, Prince William Sound, Alaska, 1993–2013 relative to a pre-spill abundance estimate (from Ballachey et al. 2014).

The age distribution of sea otter carcasses recovered from beaches in western Prince William Sound each spring has proven to be another important gauge of population status (Monson et al. 2000, 2011). Discovered carcasses are assumed to be representative of mortality patterns in the population. Teeth are extracted from carcasses to assign age, based on cementum layers. Under normal conditions, mortalities in sea otters, like most other long-lived mammals, are concentrated in the youngest (0-1 years) and oldest (> 8 years) age classes. This pattern is evident in data from western Prince William Sound collected prior to the EVOS (Figure). However, during the year of the spill (1989) and the subsequent 20 years, a different pattern of mortality was evident, with higher proportions of prime-age (2-8 years) otters dying. This difference was interpreted as evidence of elevated mortality related to effects of exposure to lingering oil (Monson et al. 2000, 2011). However, in recent years (2010 to 2013), mortality patterns were similar to those expected under normal conditions and observed pre-spill (Figure). These data suggest that between 20 and 25 years after the spill direct chronic or delayed toxic effects of exposure to lingering *Exxon Valdez* oil were no longer causing sea otter mortality.

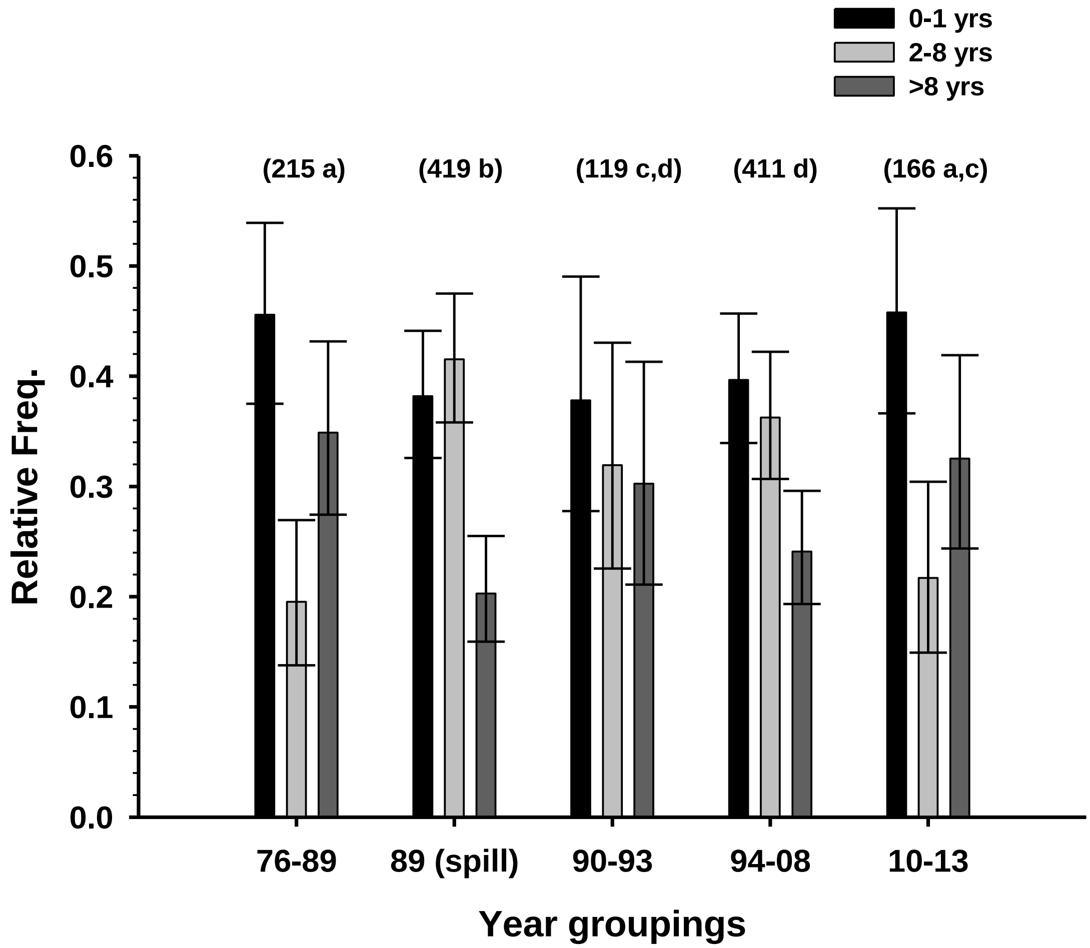


Figure 13. Relative age distributions of sea otter carcasses collected on western Prince William Sound beaches from 1976 to 2013. Total numbers of carcasses collected are in parentheses above each grouping and distributions with the same letter do not differ significantly from each other (from Ballachey et al. 2014).

Killer whales suffered acute mortalities in both a resident and transient pod that occur in Prince William Sound (Matkin et al. 2008). Neither of these pods has recovered to pre-spill numbers, although it is exceedingly unlikely that exposure to lingering oil is leading to chronic direct effects. Killer whale recovery is constrained by demographic factors associated with life history characteristics and small population size. Timeline to recovery for this species is unknown, and it is possible that the transient pod will never recover (Matkin et al. 2008).

Factors Related to Recovery Timelines

As indicated above, mechanisms leading to oil spill injury and timelines to population recovery vary widely among wildlife species. In the acute phase of oil spills, wildlife that spend much of their time on the ocean surface are particularly vulnerable to direct spill effects (Piatt et al. 1990, Goldsworthy et al. 2000), as that is where much of the oil occurs. For example, killer whales were observed surfacing in oil slicks (Matkin et al. 2008). In the chronic phase, much of the bioavailable oil is found in intertidal habitats, so wildlife that use those habitats are more likely to be exposed to lingering oil and, therefore, to be subject to direct chronic effects of exposure. In the case of the *Exxon Valdez* spill, wildlife utilizing intertidal habitats showed chronic exposure, chronic direct effects of oil, or both. These included river otters (Bowyer et al. 2003), harlequin ducks (Esler et al. 2002, 2010), and sea otters (Bodkin et al. 2002, Dean et al. 2002, Monson et al. 2011, Bodkin et al. 2012), mentioned above, as well as pigeon guillemots (*Cepphus columba*; Golet et al. 2002), black oystercatchers (*Haematopus bachmani*; Andres 1999), and Barrow's goldeneyes (*Bucephala islandica*; Esler et al. 2011).

In addition to habitat use, diet also can influence vulnerability to lingering oil exposure and thus likelihood of injury and delayed population recovery. In the case of the *Exxon Valdez* spill, wildlife that consumed benthic invertebrates were more likely to be exposed to oil and subject to chronic direct effects of lingering oil (Peterson and Holland-Bartels 2002). There may be multiple reasons for this. First, wildlife consuming benthic invertebrates are foraging on, and in some cases digging into, sediments that may contain lingering oil (Bodkin et al. 2012). Second, invertebrate prey, particularly filter feeders, may accumulate hydrocarbons (Fukuyama et al. 2000, Thomas et al. 2007) that, once consumed, may lead to detrimental effects on the wildlife consumers. Species consuming vertebrate prey (e.g., fish), such as river otters, may be less vulnerable, as vertebrate prey are not likely to accumulate hydrocarbons because they possess physiologic mechanisms capable of metabolizing and eliminating hydrocarbons.

Wildlife species also vary in their thermal or metabolic sensitivity to oil exposure. For example, during the acute phase of the spill, sea otters were considered to be more vulnerable than other marine mammals to effects of external oiling, due to their reliance on their fur for insulation and lack of a substantial blubber layer (Ballachey et al. 1994). Birds in cold water environments are known to be highly susceptible to hypothermia when their insulation is compromised due to feather oiling (Jenssen and Ekker 1991). External oiling also is possible during the chronic phase of oil spills, if lingering oil sequestered in the environment is released through disturbance of sediments by storms or foraging animals (Bodkin et al. 2012). Wildlife species also may be metabolically sensitive to effects of oil exposure due to naturally high metabolic rates (e.g., sea otters) or little ability to accommodate additional energetic costs due to oil exposure (e.g., harlequin ducks, Esler et al. 2002).

Conclusions

The large amount of research on wildlife following the *Exxon Valdez* oil spill, including that conducted as part of Gulf Watch Alaska in recent years, led to unprecedented documentation of varying timelines of recovery and the opportunity to evaluate underlying constraints to recovery. This body of work resulted in some unexpected findings, including durations of recovery that were measured in decades for some species. Also, chronic exposure to lingering oil persisting in intertidal sediments had stronger effects than anticipated, including effects that outweighed acute mortality in some species. Recent data indicate that two species that had protracted recovery due to long-term exposure to lingering oil, sea otters and harlequin ducks, have recently met the recovery criteria of the EVOSTC, and both were declared recovered in 2014 (*Exxon Valdez* Oil Spill Trustee Council 2014).

Effects of Lingering Oil on Subsistence Use

The EVOSTC conducted two surveys in 2003 and 2014 to determine the status of subsistence use of fish and wildlife in *Exxon Valdez*-affected area communities. The results of the 2003 survey of fourteen villages in Prince William Sound, lower Cook Inlet, the Kodiak Island Borough, and a portion of the Alaska Peninsula included:

- Harvests of key and culturally significant resources such as clams and harbor seals were lower in 2003 in most Prince William Sound, Cook Inlet, and Kodiak Island study communities compared to pre-spill levels.
- Almost half the interviewed households (46.5 percent) said that their overall subsistence uses were lower in 2003 than before the spill.
- Almost all the interviewed households (83.1 percent) said that their use of at least one kind of subsistence resource was lower in 2003 than before the spill, and 39.0 percent cited oil spill-related reasons for this decline.
- Many harvesters reported investing more harvest effort in 2003 than in earlier years, due to reduced resource populations but also due to competition with other users.
- Most survey respondents (78 percent) reported that in their view, injured subsistence resources have not recovered to pre-spill levels.
- Many survey respondents (47.2 percent) reported that youth are not learning enough about subsistence skills, primarily because of disinterest.
- Most survey respondents (72 percent) said that the traditional way of life has not recovered from the effects of the spill.

Based on the 2014 survey of the communities in Prince William Sound of Cordova, Tatitlek, and Chenega Bay, Fall and Zimpleman (2016) reported that most respondents expressed confidence in the safety of using subsistence foods, and this level of confidence has increased since 2003. Few respondents pointed to contamination from the *Exxon Valdez* oil spill as a source of concern about food safety. However, they also reported that a small but notable portions of respondents expressed concerns about food safety, especially related to Pacific herring and clams. Some key respondents wondered if lingering *Exxon Valdez* oil-contamination concerns were not voiced due to a strong preference for eating traditional foods (such as clams). *Exxon Valdez* oil contamination was commonly cited as a cause of food safety issues among those who did express a concern.

Fall and Zimpleman (2016) concluded that subsistence harvests remain an important source of food in the study communities, include a wide range of species, are frequently shared, and provide a context for expressing and sharing the skills and values intimately linked to centuries-old traditions and future cultural survival. However, the study also documented relatively low harvests compared to other post-spill years. Subsistence uses were also less diverse in 2014 than in any study year except for the first two years after spill. Many respondents stated that youth are not learning subsistence skills, elders are not engaged in transmitting essential knowledge and values, many natural resource populations have declined or are difficult to access, and the traditional way of life has not recovered from the effects of the *Exxon Valdez* oil spill.

Fall and Zimpleman (2016) suggested potential actions to include local communities in restoration efforts as well as strengthen communities for their future. These recommendations included support for cultural camps and other ways to engage elders with youth, programs to assist community residents to participate in fishing, hunting, and gathering activities, and long-term monitoring of natural resource populations as well as the affected human populations.

REMEDIATION OPTIONS FOR LINGERING OIL

In this section, we address the following questions:

- What are feasible methods for remediation of lingering oil?
- How much would remediation cost?

Restoration of sites with subsurface oil can be accomplished using a variety of methods to speed the degradation and/or removal of the oil. Using the NOAA (2010) guidance for response options for gravel beaches, the following methods were identified as having an environmental impact score of A or B (causing some impact, versus a score of C or D): monitored natural attenuation, manual oil removal, sediment reworking/tilling, and bioremediation The other methods are considered to be very intrusive (e.g., mechanical oil removal) or not effective on oil that is only in the subsurface (e.g., flooding or flushing methods do not reach the oiled layers). Sediment reworking (also known as berm relocation) and tilling were conducted successfully on nearly 30 exposed gravel beaches in 1989-1991 (Owens et al. 1991; Hayes and Michel 1999), mostly to treat oil that had penetrated into the sediments in the upper intertidal and supratidal zones. As shown in Figure 3, most of the subsurface oil is in the middle intertidal zone, where sediment reworking would be less effective and more disruptive. Also, most of the sites with subsurface oil have only intermittent exposure to wave energy (thus the rate of sediment reworking would be slow), and the excavated sediments would have to be placed in the lower intertidal (where attached biota would be severely impacted by crushing and smothering). Therefore, only three options were considered further for restoration of shoreline habitats with persistent subsurface oil: Monitored natural attenuation, manual removal, and bioremediation.

The EVOSTC has conducted extensive studies to determine feasibility of methods for remediation of the lingering oil, particularly in Prince William Sound. This work started in 2006, with the original evaluation of oil remediation technologies and treatment locations in Prince William Sound (Michel et al. 2006), which was used as the basis for the re-opener claim. There also were studies to determine the limiting factors, i.e., evaluating why was the oil persisted with so little degradation. These studies determined that the two-layer flow in some Prince William Sound beaches resulted in very low dissolved oxygen in the beach groundwater (Li and Boufadel 2010). Venosa et al. (2010) demonstrated that lingering oil would degrade with exposure to sufficient oxygen and nutrients. As a result, several years of field studies were conducted to determine methods for injecting both oxygen and nutrients into beaches to speed the rate of microbial degradation.

Results of these studies showed that injection of nutrients and oxygen increased the rate of oil degradation, but only in some beaches. For injection to work, sediments have to be thick enough so that oxygen and nutrients can be effectively applied to the oiled sediments without quickly washing out. Many beaches have only a thin layer of sediments over the bedrock, and thus injection methods won’t work.

It is important to note that black residues would remain after all PAHs were degraded using nutrient and oxygen injection methods, because oil from the *Exxon Valdez* spill contains compounds that are very resistant to degradation, such as asphaltenes and high molecular weight resins. For example, Atlas and Bragg (2009) noted that a few samples collected from pits in Prince William Sound in 2007-2008 that were described as heavily oiled residue (HOR) in fact contained less than 0.5 ppm PAHs and were composed of 85-90% resins and asphaltenes. This persistence of highly weathered oil residues after bioremediation treatments is one of the limitations of that treatment option. These residues have low chemical toxicity because they are not soluble or bio-available, however, if disturbed, they could continue to pose some risk of physical toxicity from fouling as long as they remained liquid or sticky.

Boufadel et al. (2015) used data on lingering oil locations (known and modeled), degree of oiling, and patch sizes to evaluate feasible remediation options and identify specific sites for possible treatment. They also developed initial cost estimates.

They started with an initial list of 100 sites with at least one pit with MOR or greater oiling based on actual field data or the model predictions (the report by Boufadel et al. 2015 includes detailed maps showing these sites). Each site was inspected individually and removed from the list if it met the following criteria for the 54 known sites:

- The oiled layer thickness was 5 cm or less, because the volume of oil present was not likely to be enough to warrant disturbances during removal actions. Five sites fell within this category.
- The site was composed of steep, rocky rubble or very large boulders, because the volume of oil present was not likely to be enough to warrant disturbances during removal actions, which would require extensive movement of the large rocky rubble to access small patches of oil underneath. Two sites fell within this category.
- The oil thickness was greater than 5 cm, but additional pit data that indicated that it was an isolated occurrence and available chemistry data indicated the total PAH concentrations were below 44 parts per million (ppm), which is the Effects Range Median–a commonly used sediment screening level (Long et al. 1995). One site fell within this category, with a total PAH of 30 ppm in a sample collected in 2007.
- The oil thickness was greater than 5 cm, but the oiled sediments included a shallow peat layer, because the volume of oil present was not likely to be enough to warrant disturbances to the shallow peat during removal actions. One site fell within this category.
- The pit with MOR was in the upper intertidal zone or under very large boulders and isolated from any other oiled pits that had LOR or OF. Three sites fell within this category.
- The sites were on Point Helen, which is difficult to access and where the surface sediments consist of very large boulders, making the oil likely not readily bio-available. The pits with MOR were widely scattered, indicating that the patches were likely small. Two sites fell within this category.

For the 46 modeled sites, different screening criteria were used to remove sites, because there were no field data available to use for screening:

- The site was adjacent to a known site that was removed from further consideration. One site fell into this category.
- The site was adjacent to a known site, but the adjacent site did not have the same geomorphic characteristics of the field-surveyed site. Most often, this was because the adjacent site did not have nearshore breakwaters or tombolos. Four sites fell into this category.
- The modeled (estimated) area of MOR (or greater) oiled sediments was equal to or less than 20 m^2. This area was selected as a threshold because most of the known candidate sites that were retained had more than 20 m^2 of estimated MOR or greater. Only five sites exceeded this threshold.

- The site was in a highly sensitive area where removal actions would likely disturb sensitive biological or known archaeological resources; this guideline was applied only to one site on Seal Island.

After screening with these criteria, the list of candidate sites for restoration consisted of 63 sites: 40 known sites, 18 model-predicted sites that were adjacent to known sites, and 5 model-predicted sites that were not adjacent to known sites. After careful evaluation of all remediation options, three restoration methods were selected as most feasible:

- Monitored Natural Attenuation: Relies on monitoring the sites every five years to evaluate the effectiveness of natural removal over time. This method would not contribute to oil removal; it would just document the oil amount and state over time.
- Manual Technique: Requires manual excavation to 1) remove liquid oil using sorbents and solidifiers, 2) treat oiled sediments prior to re-placement on the beach, and 3) dispose of oiled sediments that could not be treated on site. This method could release some oil during treatment, as any disturbance to the sediments does generate sheens and oil droplets as the tide floods the disturbed area. It would include methods to contain and recover any released oil, though such methods are not expected to be 100% successful.
- Bioremediation Technique: requires delivery of chemical amendments (oxygen and nutrients) to enhance the natural biodegradation of oil, and in particular the biodegradation of the PAH.

In principle, Monitored Natural Attenuation or Manual Technique could be applied at all sites. However, use of the Bioremediation Technique is more limited, as it requires beaches with sufficient sediment material to ensure delivery of the oxygen and nutrients to the oiled layers.

All data for each site were reviewed to evaluate the site for feasibility of using the Bioremediation Technique, based on what was learned during the field trials at four sites. If bioremediation was not feasible, it was assigned to the Manual Technique. The Bioremediation Technique was determined to be feasible at 9 sites, and Manual Technique was determined to be feasible at the remaining 54 sites. The preliminary cost estimates for the remediation of the 63 sites are shown in Table 6.

Table 6. Summary of costs for various restoration techniques.

Restoration Technique	Cost USD
Monitored Natural Attenuation at 20 sites	$2,347,000
Manual Technique at all 63 sites	$13,470,383
Manual Technique at 54 sites + Bioremediation Technique (BT) at nine sites	$17,593,109

Boufadel et al. (2015) evaluated the restoration methods for oiled shorelines by considering only the technical feasibility and predicted disruption of each method, the estimated cost, and the achievable endpoints. There are other factors that the State and Federal trustees should take into consideration to decide which, if any, of the sites should be restored and which methods should be used. Because there is no evidence of ecological impacts from the remaining oil, these factors include benefits to subsistence users, recreational users, and the public in general likely to be gained by restoration, to be balanced against potential ecological costs from disturbances to wildlife and disruption of habitats during implementation. The valuation of these factors is critical to the final determination of where and how best to achieve restoration of the lingering oil.

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